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Long-term Exposure of European Population Subgroups to PM2.5 and NO_2

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Long-term Exposure of European Population Subgroups to PM2.5 and NO_2

Von der Fakultät Energie-, Verfahrens- und Biotechnik der Universität Stuttgart
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Naixin Li
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List of Acronyms and Abbreviation

Abbreviation	Description
AER	Air exchange rate
AHU	Air handling unit
AirBase	European air quality database
APEX	Air Pollutant Exposure Model
ASHRAE	American Society of Heating, Refrigerating and Air-Conditioning Engineers
BUA	Bronchodilator usage adults
BUC	Bronchodilator usage children
CB	Chronic bronchitis
CE	Capture efficiency
CEN	European Committee for Standardization
CI	Confidence interval
CLC	CORINE Land Cover
CO	Carbon monoxide
CO_2	Carbon dioxide
CH_4	Methane
CHA	Cardiac hospital admissions
CHAD	Consolidated Human Activity Database
CRF	Concentration response function
CTM	Chemical transport model
EDGAR-HYDE	Emission Database for Global Atmospheric Research - Hundred Year Database for Integrated Environmental Assessment
EE	Eastern Europe
EEA	European Environment Agency
EHIS	European Health Interview Survey
EIONET	European Environment Information and Observation Network

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Abbreviation	**Description**
EMEP/MSC-W	European Monitoring and Evaluation Programme Meteorological Synthesizing Centre - West
ERF	Exposure response function
ETS	Environmental tobacco smoke
EU	European Union
EU-SILC	European Union Statistics on Income and Living Conditions
EUROSTAT	European Union Statistical Office
EXPOLIS	Air Pollution Exposure Distributions within Adult Urban Populations in Europe
GSD	Geometric standard deviation
HAF	Harmonised aggregate file
HEALS	Health and Environment-wide Associations based on Large population Surveys
HEF	Harmonised episode file
HEIMTSA	Health and Environment Integrated Methodology and Toolbox for Scenario Assessment
HEPA	High-efficiency particulate absorber
HRAPIE	Health risks of air pollution in Europe
HSF	Harmonised simple file
HVAC	Heating, ventilation and air conditioning
IM	Infant mortality
IncMSE	Increase in mean squared error
INTARESE	Integrated Assessment of Health Risks from Environmental Stressors in Europe
I/O ratio	Indoor/Outdoor ratio
IPCC	Intergovernmental Panel on Climate Change
ISCED	International Standard Classification of Education
LAU	Local Administrative Unit
LRSA	Lower respiratory symptoms adults
LRSC	Lower respiratory symptoms children
ME	Micro-environment

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Abbreviation	Description
MERV	Minimum efficiency reporting value
MRAD	Minor restricted activity days
MTUS	Multinational Time Use Study
NE	Northern Europe
N_2O	Nitrous oxide
NH_3	Ammonia
NH_4NO_3	Ammonium nitrate
NIST	National Institute of Standards and Technology
NMVOC	Non-methane volatile organic compound
NO	Nitrogen oxide
NO_2	Nitrogen dioxide
NUTS	Nomenclature of Territorial Units for Statistics
NWE	Northwestern Europe
O_3	Ozone
OFFICAIR	On the reduction of health effects from combined exposure to indoor air pollutants in modern offices
PBSC	Prevalence of bronchitic symptoms in asthmatic children
PM	Particulate matter
PM2.5	Particulate matter with an aerodynamic diameter smaller than 2.5 µm
PM10	Particulate matter with an aerodynamic diameter smaller than 10 µm
PTEAM	Particle Total Exposure Assessment Methodology
RAD	Restricted activity days
RHA	Respiratory hospital admissions
SE	Southern Europe
SES	Socio-economic status
SHEDS-PM	Stochastic Human Exposure and Dose Simulation model for Particulate Matter
SI	Street increment

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Abbreviation	**Description**
SINPHONIE	Schools Indoor Pollution and Health: Observatory Network in Europe
SO_2	Sulphur dioxide
TIMES PanEU	Pan-Europäische TIMES Energiesystemmodell
TRIM	Total Risk Integrated Methodology
UN	United Nations
UNECE	United Nations Economic Commission for Europe
UOWM	University of Western Macedonia
US	United States
US EPA	United States Environmental Protection Agency
VOC	Volatile organic compound
WHO	World Health Organization
WLD	Work loss days
YLD	Years lost due to disability
YOLL	Years of life lost

Abstract

Numerous epidemiological studies have demonstrated the damaging influence of air pollutants on human health. However, the environmental health studies up to now use urban background concentrations in the ambient air to estimate the health risks, while the inhalation of toxic substances, i.e. the concentration of pollutants, where the exposed person breathes, is the relevant indicator for estimating the health impacts. The main objective of this thesis is to assess the long-term exposure to fine particles and nitrogen dioxide for different European subgroups that are characterised by certain features including age, gender, region and socio-economic status. The exposure simulation is realised by developing a probabilistic model that incorporates an air quality model for estimating the ambient pollutant concentration, a mass-balance model for assessing the concentration of indoor micro-environments and a life course trajectory model for predicting retrospectively the transition between socio-economic states. The results of the exposure modelling are subsequently incorporated with exposure response functions (ERFs), aggregation factors and monetary values to assess health impacts and damage costs.

The geometric mean annual exposure to PM2.5 and NO_2 for Europeans in 2015 was 17.0 and 14.0 $\mu g\ m^{-3}$ respectively with a geometric standard deviation of 1.7 and 1.6. The total DALYs (disability adjusted life years) caused by the exposure to both pollutants for the European population in 2015 were 1.22×10^{7} (95% CI: 3.61×10^{6}-2.95×10^{7}) and the damage costs were 1.01×10^{12} (95% CI: 3.46×10^{11}-2.37×10^{12}) €. The average exposure over lifetime of an 80-year-old European to PM2.5 and NO_2 reached 23.86 (95% CI: 2.95-81.86) and 13.49 (95% CI: 1.36-43.84) $\mu g\ m^{-3}$, which resulted in years of life lost (YOLL) of 3.53×10^{-2} (95% CI: 4.79×10^{-3}-1.17×10^{-1}) and 4.76×10^{-3} (95% CI: 0.00-3.03×10^{-2}) per year of exposure respectively. The modelling results indicate that for European countries the outdoor air concentration is a key factor for the total exposure, especially for NO_2. Similar to the trend of the ambient pollutant concentration, the average exposure for both pollutants in Europe reached a peak between the 1980s and 1990s and showed a notable decrease afterwards due to the introduction of emission reduction policies and smoking bans. For PM2.5 the most important indoor sources include smoking and cooking, while for NO_2 cooking and wood burning are the significant contributory factors. Considerable variance of the exposure levels has been observed among different European countries for both pollutants. The results also reveal that the exposure and its source distribution are dependent on gender, smoking habit and socio-economic status including household income level, employment status, education level and civil status.

Kurzfassung

In zahlreichen epidemiologischen Studien wurde der schädlichen Einfluss von Luftschadstoffen auf die menschliche Gesundheit nachgewiesen. Alle diese Studien verwenden städtische Hintergrundkonzentrationen in der Außenluft, um die Gesundheitsrisiken abzuschätzen. Jedoch ist der relevante Indikator für die Einschätzung der gesundheitlichen Auswirkungen offensichtlich die Schadstoffkonzentration der eingeatmeten Luft. Das Hauptziel dieser Arbeit ist es, Werte für diesen Indikator, also die langfristige Exposition mit Feinstaub und Stickstoffdioxid für verschiedene europäische Bevölkerungsgruppen, die durch bestimmte Merkmale wie Alter, Geschlecht, Region sowie sozioökonomischen Status charakterisiert sind, abzuschätzen. Die Berechnung der Exposition erfolgt mit Hilfe eines probabilistischen Modellsystems. Dieses besteht aus einem atmosphärischen Modell zur Abschätzung der Schadstoffkonzentration in der Außenluft, einem Mass-Balance-Modell zur Beurteilung der Konzentration in Innenräumen unter Berücksichtigen von Emissionsquellen in Innenräumen und einem Modell zur Abbildung des Lebensverlaufs der analysierten Personen hinsichtlich ihres sozioökonomischen Status. Die Ergebnisse der Expositionsmodellierung werden anschließend mit Expositions-Wirkungs-Beziehungen und spezifischen monetären Werten multiplizert, um die Auswirkungen auf die Gesundheit und die verursachten Schadenskosten abzuschätzen.

Der geometrische Mittelwert der Exposition von PM2,5 bzw. NO_2 für Europäer im Jahr 2015 betrug 17,0 bzw. 14,0 µg m^{-3} mit einer geometrischen Standardabweichung von 1,7 und 1,6. Die gesamten Gesundheitsschäden, die durch die PM2,5- und NO_2-Exposition in der europäischen Bevölkerung im Jahr 2015 verursacht wurden, betrugen 1,22 $\times 10^{7}$ (95% KI: 3,61 $\times 10^{6}$-2,95 $\times 10^{7}$) DALYs (disability adjusted life years), die daraus resultierenden Schadenskosten betrugen 1,01 $\times 10^{12}$ (95% KI: 3,46 $\times 10^{11}$-2,37 $\times 10^{12}$) €. Die durchschnittliche Exposition über die Lebensdauer eines jetzt 80-jährigen Europäers gegenüber PM2,5 bzw. NO_2 betrug 23,86 (95% KI: 2,95-81,86) bzw. 13,49 (95% KI: 1,36-43,84) µg m^{-3}. Diese Exposition verursachte pro Jahr einen durchschnittlichen Verlust an Lebenszeit von 3,53 $\times 10^{-2}$ (95% KI: 4,79 $\times 10^{-3}$-1,17 $\times 10^{-1}$) und 4,76 $\times 10^{-3}$ (95% KI: 0,00-3,03 $\times 10^{-2}$). Die Modellierungsergebnisse zeigen, dass die Schadstoffe in der Außenluft für die europäischen Länder einen großen Teil der Gesamtexposition ausmachen, insbesondere bei NO_2. Ebenso wie der Trend der Schadstoffkonzentration in der Außenluft erreichte die durchschnittliche Exposition der Bevölkerung gegenüber beiden Schadstoffen in Europa zwischen den 1980er und 1990er Jahren ihren Höhepunkt und nahm danach infolge der Einführung von Emissionsminderungsmaßnahmen und Rauchverboten deutlich ab. Bei PM2,5 gehören Rauchen und Kochen, genauer das Braten zu den wichtigsten Innenraumquellen, wohingegen bei NO_2 das Kochen mit Gasherden die wichtigste Innenraumquelle ist. Bei beiden Schadstoffen sind die Expositionen je nach Land sehr unterschiedlich. Die Ergebnisse zeigen auch, dass die Exposition vom Geschlecht, den Rauchgewohnheiten und Aspekten des sozioökonomischen Status, z. B. dem Haushaltseinkommen, dem Beschäftigungsstatus, dem Bildungsniveau und dem Familienstand abhängen.

1 Introduction

1.1 Background and problem statement

Air quality has drawn worldwide attention since the last few decades. Despite the huge efforts that have been made, air pollution is still resulting in severe health impacts in Europe. Various epidemiological studies have proven the correlation between adverse health outcomes and air pollutants, in which the largest part was owing to particulate matter (PM) and nitrogen dioxide (NO_2) (WHO 2013a; WHO 2013b):

- Fine particles

  According to WHO (2005b), PM is defined as "an air pollutant consisting of a mixture of particles that can be solid, liquid or both, are suspended in the air and represent a complex mixture of organic and inorganic substances". Referring to its source, PM is generally classified into primary and secondary particles. Primary particles are emitted directly into the atmosphere from either natural sources such as volcano, forest fire and sea spray, or anthropogenic processes such as fossil fuel and wood combustion, industrial activities and abrasion process, e.g. tyre and brake wear (Reddington et al. 2011; Penkała et al. 2018). Secondary particles appear in the air due to chemical reactions of precursors such as sulphur dioxide (SO_2), nitrogen oxides (NO_x), ammonia (NH_3) and non-methane volatile organic compounds (NMVOCs) (Hodan and Barnard 2004).

  PM varies in size, in which the ones with an aerodynamic diameter smaller than 2.5 micrometres are categorised as fine particles (PM2.5) (Atkinson et al. 2015). Compared to the coarse fraction of PM, PM2.5 is deemed as more threatening because it can penetrate deeply into the lung, corrode the alveolar wall and damage the respiratory system and other parts of the body (Pražnikar and Pražnikar 2012; Xing et al. 2016; Li et al. 2018).

  PM2.5 has been revealed to be strongly associated with the incidence of cardiovascular and pulmonary diseases and an increase of the morality rate. The correlation is even more evident in the vulnerable groups such as infants, children, pregnant women, the elderly and people with heart or lung diseases (Schwartz et al. 1996; Boezen et al. 1999; Samoli et al. 2005; Ostro et al. 2006; Huynh et al. 2006; Orru et al. 2011; de Oliveira et al. 2012).

- Nitrogen dioxide

  As one of the precursors of particles, nitrogen dioxide (NO_2) is a gaseous pollutant that is mainly yielded through combustion processes (Hao et al. 1987; Linak et al. 1990). A large share of the ambient NO_2 stems from motor vehicles, especially in urban areas (WHO 2003). Besides, NO_2 is the precursor involved in the atmospheric reaction that leads to the formation of ozone (O_3) and ammonium nitrate

(NH_4NO_3). Compared to PM2.5, less studies have investigated the health impacts of NO_2. New evidences have shown that NO_2 is correlated with respiratory difficulties, lung function decrements, or even low birth weight (Ha et al. 2001; Cibella et al. 2015). According to the studies of Peters et al. (1999) and Shima and Adachi (2000), children are more susceptible to NO_2 than adults with respect to asthma. However, the complex interrelationships between NO_2, PM and ozone makes it difficult to identify whether the impacts are originated from NO_2 itself, or the mixture of these pollutants.

Traditional epidemiological studies have provided valuable insights for understanding the impacts of PM2.5 and NO_2 on human health. However, these studies have only focused on pollutant concentrations, which were mostly taken from ambient rural or urban monitoring stations. The main weakness of this method is its underlying assumption that basically the outdoor concentrations in urban background stations determine the health effects. This assumption goes against the fact that during a day people are moving between different micro-environments. The utilization of ambient concentration data as proxy ignores the influence of personal activity pattern, building characteristics and indoor pollutant sources since in reality people spend the dominant portion of their time indoors. According to Crist et al. (2008) and Amato et al. (2014), the personal exposure is significantly correlated with the indoor pollutant concentration. Consequently the ambient concentration is inadequate to reflect the real pollutant exposure of an individual. Thus, to estimate the health impacts, the exposure to pollutants has to be known; i.e. a model to simulate the exposure is necessary.

Moreover, most of the published studies are constrained to the analysis of concentration of a specific short time period. Especially for earlier years (before 1975), few studies on the health impacts of PM2.5 and NO_2 can be accessed. However, the health status of persons, especially with respect to the chronic disease caused by environmental stressors, is influenced by the long-term exposure over a whole life time (Kloog et al. 2013; Wang et al. 2017; Yitshak-Sade et al. 2018). Thus, to assume that the exposure during the epi-study is the same as in earlier life periods is apparently not convincing.

Last but not least, the limited temporal and spatial coverage of monitoring data has restricted the epidemiological method to be carried out only to a small number of areas. Hence, to develop a methodology that can be applied to the whole European population for modelling the long-term exposure to PM2.5 and NO_2 would definitely bring new visions for health impacts studies and optimal policy-making.

1.2 Socio-economic status and exposure

In the last decades, the implementation of a series of regulations has improved the overall air quality in Europe. However, the improvement is not equally applied to all the population since a disproportionate share of the burden of environmental exposure falls on certain vulnerable groups (American Lung Association 2001). These groups are usually from low socio-economic classes and argued to be associated with serious environmental inequalities (Lee 2002; Kim et al. 2007; Laurent et al. 2007; Ou et al. 2008).

The associations between the socio-economic status (SES) and the exposure to both PM2.5 and NO_2 have been extensively analysed worldwide. According to Evans and Kantrowitz (2002), Kohlhuber et al. (2006) and Branis and Linhartova (2012), the exposure was significantly influenced by SES variables such as income level, employment status and education level. However, some of the findings challenged the hypothesis that the people from a lower SES class suffered from the worse environmental conditions. Byun et al. (2010) observed higher PM10 levels in children's bedrooms for families with higher monthly expenses in South Korea. Charafeddine and Boden (2008) studied the effect of income inequality and the results did not follow the assumption that people living in higher income inequality areas were more vulnerable to the impact of air pollution. Due to the inconsistency of the findings, the influence of socio-economic status on exposure needs to be further studied.

It should be noticed that most of the studies involved have aimed at the relationships between the ambient pollutant concentrations, socio-economic factors and health outcomes. As pointed out in Section 1.1, the outdoor pollutant concentration is unable to reflect the real exposure an individual burdens. Thus, it is necessary to examine the association between the SES variables and the actual exposure rather than just the outdoor concentration.

1.3 Scope and objectives

Based on the context discussed above, the main objective of this thesis is to develop and apply a methodology to simulate the lifelong exposure to PM2.5 and NO_2 for European population subgroups characterised by age, gender, region, socio-economic status and behavioural habits. The framework should be applicable for the EU27 countries[1] plus Norway and Switzerland (EU27+2). The following aspects are included:

- to incorporate the emission data and air quality models that enable the calculation of the ambient concentration for recent and previous years.
- to simulate the concentrations of pollutants for different micro-environments, especially the indoor environments.
- to investigate the contribution of different emission sources, including indoor sources of pollution.
- to estimate the annual average exposure of population subgroups with certain features.

[1]The EU27 countries in this thesis refer to the European Union in the period between 2007 and 2013, before Croatia joined, when it had 27 countries. Detailed information can be found in Table 3-4.

- to analyse the influence of socio-economic status, home town and behavioural habits on exposure.
- to link the exposure model with a life course trajectory model to assess the lifelong exposure of the individuals.
- to apply the results of the exposure modelling to exposure response functions (ERFs), aggregation factors and monetary values to assess health impacts and damage costs.

1.4 Outline of the thesis

This thesis comprises six chapters, with the background information, problem statement and objectives of the research in Chapter 1 as the starting point.

In Chapter 2 the state of the art of the exposure modelling and the improvements made in this thesis are explained.

Chapter 3 is devoted to the description of the general methodological framework implemented and enhanced within this thesis.

In Chapter 4 the comprehensive work of data collection is conducted. This is followed by the presentation of the results and the uncertainty estimation in Chapter 5.

Finally, the conclusions of this thesis along with an outlook on future research are compiled in Chapter 6.

2 State of the art of the exposure models

2.1 Existing exposure models

As stated in Chapter 1, the pollutant concentration measured at the monitoring station is not able to reflect the real personal exposure. Multiple studies have conducted fieldwork with monitoring meters that are portable, or installed in each micro-environment, for a certain period of time to measure the average personal exposure (Clayton et al. 1993; Jantunen et al. 1998; Meng et al. 2009; Wallace et al. 2011). Burke et al. (2001) have pointed out that 1) the field studies were usually expensive to operate, especially when they were applied to a large number of participants, 2) the measurement study could be troublesome for some vulnerable subgroups, such as children and people with disease and 3) the results of the fieldwork were only representative of specific groups of people and not applicable to the overall population.

The disadvantages of the fieldwork have imposed the development of models for estimating the individual exposure. In the following sections some existing models are introduced.

2.1.1 APEX

The Air Pollutant Exposure Model (APEX) is part of the overall Total Risk Integrated Methodology (TRIM) model framework of the United States Environmental Protection Agency (US EPA 2017). As a probabilistic model, it is able to simulate the movement of individuals through time and space, as well as their exposure to a given pollutant in different micro-environments (Johnson et al. 2018). The APEX model starts with characterising the study areas and individuals. Daily activity patterns for these individuals in the study area are obtained from Consolidated Human Activity Database (CHAD) (McCurdy et al. 2000). The APEX enables the modelling for concentration in multiple micro-environments including residences, schools, offices and vehicles. The hourly concentrations in the micro-environments are simulated probabilistically based on the user-defined descriptions, and subsequently aggregated at daily, monthly and annual average level (US EPA 2008).

APEX is able to assess the human exposure to various air pollutants including NO_2, O_3, carbon monoxide (CO), SO_2 and PM2.5 (US EPA 2009; 2010; Johnson et al. 2018). Nevertheless the model is principally applicable for US population due to the data limitations (e.g. APEX utilizes the time activity data from the American surveys).

2.1.2 SHEDS-PM

US EPA has developed a series of Stochastic Human Exposure and Dose Simulation models, in which the SHEDS-PM was built up for predicting the distribution of exposure to PM2.5. Similar to APEX, the SHEDS-PM is capable of assessing the concentration in micro-environments. The concentration of residential buildings is simulated with a mass-balance equation that synthesises data for ambient outdoor concentration, physical factors (e.g. air exchange rate, penetration) and emission strengths for indoor sources (e.g. smoking, cooking), while the concentration in non-residential micro-environments (vehicles, offices, schools, stores and restaurants/bars) is calculated with equations developed from regression analysis (Burke 2005). Activity patterns for population being modelled are also taken from CHAD.

Except for the study conducted in Philadelphia from Burke et al. (2001), the model has also been applied in the research of Georgopoulos et al. (2005), Cao and Frey (2011) and Liu and Frey (2011). Unfortunately this model is only suitable for population in the US with respect to the input data.

2.1.3 EXPOLIS

A stochastic exposure modelling framework for PM2.5 has been developed as a part of the EXPOLIS (Air Pollution Exposure Distributions within Adult Urban Populations in Europe) study. The framework employs the micro-environment approach to simulate the exposure of different sub-populations. The model starts with defining the relevant micro-environments and requires data of the concentration distributions of each micro-environment, as well as the time-activity patterns. For indoor micro-environments, the framework uses ambient concentrations, effective penetration factors and contributions of indoor sources to assess the concentrations. A log-normal distribution is assumed for the micro-environment concentrations and the contribution of indoor sources, while a beta distribution for the time spent in a micro-environment and the penetration factor (Kruize et al. 2003).

Although the measurement data for NO_2 were available, only the framework for PM2.5 has been established. Besides, the model is highly dependent on measurement data and constrained to the four fieldwork cities, i.e. Athens, Basel, Helsinki and Prague.

2.1.4 Models for NO_2

Compared to PM2.5, few models have been built for the exposure assessment of NO_2:

- Sexton et al. (1983) have developed a simple deterministic model that relates the NO_2 exposure to background ambient concentration, indoor values and human activities. Data for the ambient and indoor parameters originate from the monitoring programmes of six American cities. This study has proven the influence of outdoor air, cooking fuel, air exchange rate, as well as indoor sources on NO_2 exposure.

- Dimitroulopoulou et al. (2001) have established a dynamic multi-compartment computer model to calculate the exposures to NO_2 of a typical homemaker, school-attending child and office worker. The modelling approach applied combines a physical model to simulate the hourly indoor pollutant concentration of different micro-environments, and an exposure model to predict the personal exposure by combining the simulated concentrations in a series of micro-environments.

- Fabian et al. (2012) simulated the concentrations in a typical multi-family building of US with the CONTAM model. The CONTAM is a multi-zone indoor air quality and ventilation analysis computer program which can be applied to the simulation of airflows, contaminant concentrations and personal exposure (Dols and Polidoro 2015). The input parameters for the study of Fabian et al. (2012) include stove use, presence of exhaust fans, information for smoking and building leakiness.

These studies provide instructive insights for individual exposure modelling of NO_2. However, they are either too old, or applicable to only specific regions and population subgroups.

2.2 Improvements in this thesis

Even though several models have been developed for estimating the exposure of population subgroups, no existing models can be directly applied in this thesis. Most of the models introduced can only be used for very specific cases (certain pollutant, micro-environment, or pollutant source). Additionally, many of them are not suitable for the assessment in Europe. A few models for Europe are available, however, they are only capable of simulating for short time periods.

Thus, a new model has been developed in this thesis, which includes the following improvements and innovations compared to the existing models:

- The personal exposure models are established for both PM2.5 and NO_2 and can be applied for all EU27+2 countries.
- The personal exposure is simulated with a probabilistic model on the basis of a micro-environment approach.
- All relevant emission sources, including indoor sources and their impact on exposure, are taken into account.
- The exposure is estimated for population subgroups that are characterised by age, gender, behavioural habit and socio-economic status.
- The exposure model is for the first time synthesised with a sequence analysis model to simulate the lifelong exposure and health impacts of population subgroups and individuals.
- The results of the exposure model are combined with exposure response functions and an assessment model to evaluate health impacts and damage costs at European level.

3 Methods

3.1 Overall framework

The exposure of an individual was expressed by Watson et al. (1988) as "a description of the duration of the contact and the relevant pollutant concentration". In this thesis, the indirect approach, or the micro-environment approach (Duan 1982; Letz et al. 1984; Ryan et al. 1986) is used and the exposure of an individual can be expressed as Equation 3-1:

$$EXP = \frac{\sum_{i=1}^{n} C_i T_i}{T} \tag{3-1}$$

where:

EXP = the mean daily personal exposure [µg m^{-3}]

C_i = concentration in micro-environment i [µg m^{-3}]

T_i = time spent in micro-environment i [h]

T = total time of a day [24h]

The overall framework of methodology applied in this thesis is displayed in Figure 3-1. A comprehensive probabilistic model to estimate the temporal course of the external exposure for population subgroups that are characterised by certain features (e.g. age, gender, socio-economic status) has been developed. This model partly follows the integrated assessment approach, which was defined by Friedrich (2016) as "a multidisciplinary process of synthesising knowledge across scientific disciplines with the purpose of providing all relevant information to decision makers to help to make decisions". This approach has been widely applied in the field of evaluation of the air pollution control policies.

The main steps of the methodology framework are listed as follows:

- Ambient concentration modelling

  As the basis of the framework, the ambient concentration fields are simulated with the help of a chemical transport model (CTM), an interpolation model and multiplicative bias adjustment (see Section 3.2). The concentration fields from 1930 to 2015 are collected or generated by the author in this thesis.

- Exposure modelling

  For each individual characterised by certain features, multiple diaries for activity pattern are given by the data from the Multinational Time Use Study (MTUS) (see Section 3.4). For each diary, a set of realisations was compiled for the parameters of the models that are responsible for simulating the pollutant concentrations in micro-environments. The model parameters are represented by certain probability distributions instead of single values (see Chapter 4). The results of the exposure

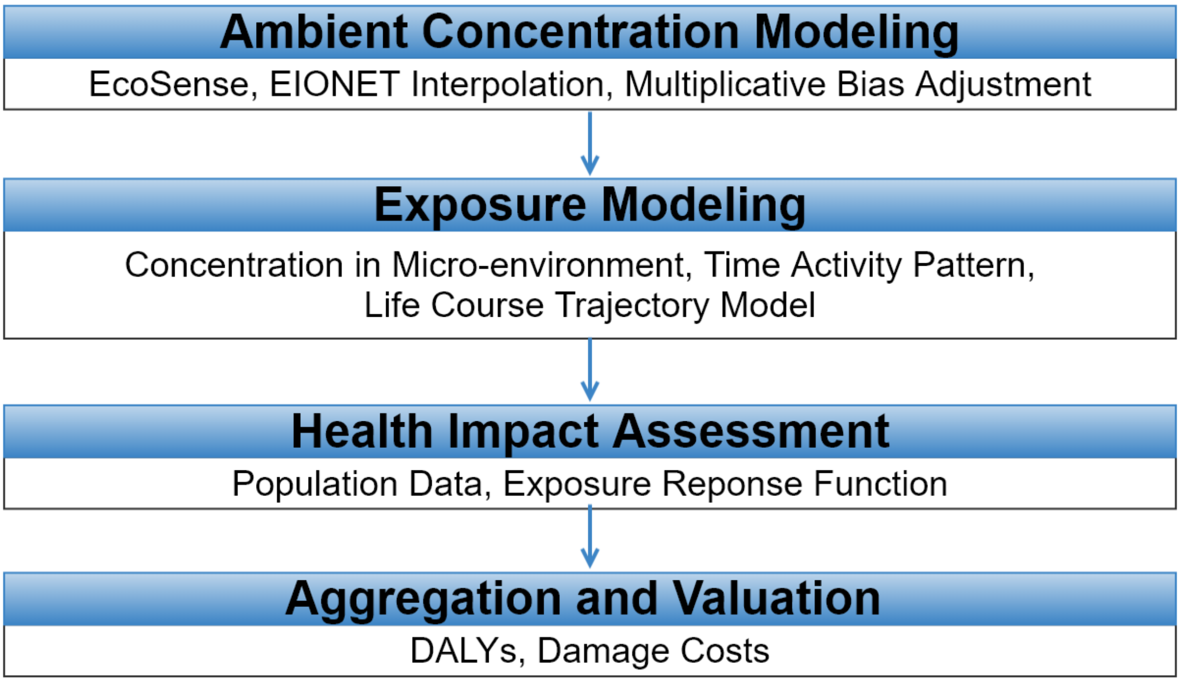

Figure 3-1: Overall framework of the methodology.

modelling are subsequently combined with a life course trajectory model to predict the long-term exposure to both pollutants (see Section 3.5).

- Health impact assessment

  The results of the exposure modelling, which are in the form of probability distributions, are used with the exposure response functions (ERFs) (see Section 3.6.1) and population data (see Section 3.6.3) to assess the health impacts at different endpoints.

- Aggregation and valuation

  To aggregate the different health endpoints and make them comparable, the health impacts are converted into DALYs (disability adjusted life years) as a measure of the burden of disease. Alternatively, each endpoint is designated with a monetary value and transformed into damage costs (see Section 3.6.2).

3.2 Ambient concentration fields

The monitoring data from measurement stations are considered the most reliable source of the air quality information. However, the available observing stations are limited considering its spatial and temporal coverage. Especially for PM2.5, there are hardly measurement data accessible before 2000. To simulate the long-term Europe-wide concentration fields, the following methods are applied in this thesis.

3.2.1 EMEP/MSC-W chemical transport model

The ambient concentration fields are the basic input data for the exposure modelling. Since the eldest subjects of the population are over 80 years old, it is necessary to simulate the ambient concentration fields back to the 1930s. The chemical transport model acts as a powerful tool to study the mechanism of chemical distribution in three-dimensional scale (Akiyoshi et al. 2001; Cope et al. 2009; Chemel et al. 2014; Dore et al. 2015). The CTM developed by European Monitoring and Evaluation Programme Meteorological Synthesizing Centre - West (EMEP/MSC-W) is one of the leading tools in Europe regarding the air pollution policy assessment (Simpson et al. 2012; Fagerli et al. 2015; Fagerli et al. 2018). The model is widely engaged in the simulation of concentration fields for air pollutants including PM2.5 and NO_2.

EMEP MSC-W provides the modelled air concentration fields from 1980 to 2015 for European countries[2]. However, the performance of EMEP/MSC-W model has been evaluated and over/under-predictions of concentrations have been discovered (Fagerli et al. 2012). Especially for PM2.5, an overall trend of underestimation has been reported (Bessagnet et al. 2014; Schaap et al. 2015). To modify the modelling results, the EIONET interpolation method and multiplicative bias adjustment are employed by the author in this thesis.

3.2.2 EIONET interpolation

An interpolation methodology was developed by Horálek et al. (2007) to adjust the EMEP /MSC-W modelling results. This method is a combination of the linear regression model between AirBase[3] measurement data and supplementary data and the interpolation of the regression model residuals (see Figure 3-2). The supplementary data include EMEP/MSC-W model results, meteorological data, altitude and population density. The maps are generated for urban and rural stations separately and subsequently merged into one based on the population density. For NO_2, it is worth mentioning that the interpolated maps ignore the dense-traffic stations where the concentration levels usually exceed the regulated limits. Therefore, the improved method from Horálek et al. (2017) is adopted, which uses the measurement data of traffic stations, the CORINE Land Cover (CLC) data (EEA 2016) and the OpenStreetMap data (OpenStreetMap 2017) to generate the traffic grid map additionally. The traffic grid map is merged together with the urban and rural maps based on the population density and the buffers around the streets or roads.

EIONET provides the interpolated European air quality maps for several years[4] (see Table 3-1 for data availability). The interpolation method is considered as an effective tool to modify the results of CTMs. However, one obvious drawback to this method is its inapplicability when no sufficient monitoring data are available, which is particularly an issue for PM2.5 (only five stations accessible in 1998). Hence, the author only applies the

[2] https://www.emep.int/mscw/
[3] https://www.eea.europa.eu/data-and-maps/data/airbase-the-european-air-quality-database-6
[4] https://acm.eionet.europa.eu/databases/

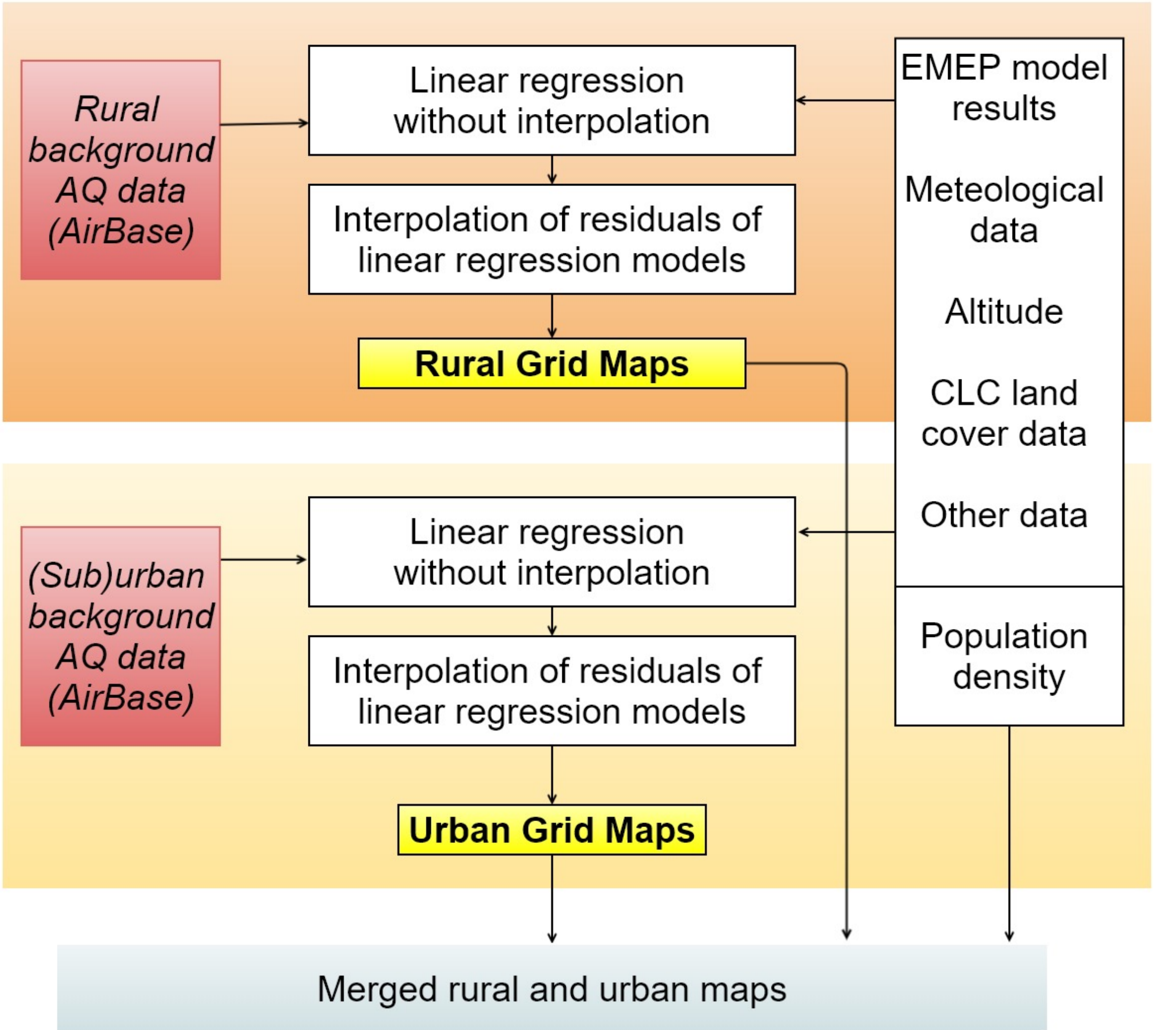

Figure 3-2: Flow chart of the interpolation methodology using the monitoring data, the supplementary data (EMEP/MSC-W model data, meteorological data, altitude and population density), the linear regression and the interpolation techniques. The maps are developed for the urban and rural stations respectively and merged into one final map with the population density (Horálek et al. 2007). For NO_2 a traffic grid map is generated additionally incorporating the land cover and traffic data. The traffic map is combined with the above-mentioned two maps based on the population density and the buffers around the roads (Horálek et al. 2017).

interpolation method to fill the data gaps for PM2.5 after 2005 and for NO_2 after 2000.

3.2.3 Multiplicative bias adjustment

For years between 1980 and 2000s when no interpolated concentration maps are available, multiplicative bias adjustment factors are employed to modify the EMEP/MSC-W model

Table 3-1: Data applied for simulating the PM2.5 and NO_2 concentration fields. Three main types of data are involved: concentration measurement data, concentration modelling data and emission data.

Type	Pollutant	Source	Resolution	Years data available
Concentration fields	PM2.5	EIONET	10km×10km	2005, 2007, 2008, 2010-2012
		EMEP/MSC-W	50km×50km	1980, 1985, 1990, 1995-2015
Measurement data		AirBase	-	1998-2012
Emission		EDGAR-HYDE	1°×1°	1890-1990
Concentration fields	NO_2	EIONET	10km×10km	2007, 2011
		EMEP/MSC-W	50km×50km	1980, 1985, 1990, 1995-2015
Measurement data		AirBase	-	1973-2012
Emission		EDGAR-HYDE	1°×1°	1890-1990

results. Multiplicative bias correction is one of the widely applied methods to enhance the CTM performance (McKeen et al. 2005; Monteiro et al. 2013), which can be expressed by Equation 3-2 as:

$$C_{corrected} = R_{bias} \times C_{model} \tag{3-2}$$

where:

$C_{corrected}$ = corrected model concentration [µg m^{-3}]
R_{bias} = multiplicative bias adjustment factor
C_{model} = raw model concentration [µg m^{-3}]

In this thesis, the multiplicative bias adjustment factors are developed by the author with the EIONET concentration fields generated in Section 3.2.2. The accessible EIONET concentration maps (including the ones simulated by the author) are compared with the EMEP/MSC-W data for the corresponding years to generate the ratios at the EMEP grids level[5]. The ratios are calculated for each available year and averaged to yield the multiplicative bias adjustment factors. These factors are subsequently applied to modify the EMEP/MSC-W concentration fields for years when no interpolated maps can be simulated.

[5]https://www.eea.europa.eu/data-and-maps/data/emep-grids-reprojected-by-eea

3.2.4 EDGAR-HYDE and EcoSense

Unfortunately, for the years before 1980 no simulation data from EMEP/MSC-W are directly available. To fill the data gaps, emission data from EDGAR-HYDE (Emission Database for Global Atmospheric Research - Hundred Year Database for Integrated Environmental Assessment) (van Aardenne et al. 2001) are utilised as input data to simulate the concentration grids. EDGAR-HYDE contains anthropogenic emissions of CO_2, CH_4, N_2O, CO, NO_x, NMVOC, SO_2 and NH_3 for the period from 1890 to 1990 with a temporal resolution of 10 years. Figure 3-3 shows the development of the total emission for the above-mentioned pollutants in European countries from 1930 to 1980. The sectors covered include "Fossil combustion", "Fossil production", "Biofuel combustion", "Industrial processes", "Agriculture land", "Animals", "Savannah burning", "Deforestation", "Agriculture waster burning" and "Landfills". The emission data are subsequently implemented in the EcoSense model as the input data to simulate the concentration fields. EcoSense is an integrated atmospheric dispersion and exposure assessment model which can be used for assessing the external costs related to the exposure to airborne pollutants (University of Stuttgart 2018). As a starting point, the EcoSense model uses the source-receptor matrices from EMEP/MSC-W to calculate the background concentrations. This part of the model is applied in this thesis and the concentration fields simulated are further modified with the multiplicative bias adjustment factors generated by the author (see Section 3.2.3).

However, it must be stressed that the EDGAR-HYDE database contains no emission data for particles which are required for the operation of EcoSense model. To solve this problem, the author uses the emission data of EDGAR (Emissions Database for Global Atmospheric Research) to calculate the average ratio between PM2.5, PM10 and NO_x. The EDGAR v4.3.2 assembles the gaseous and particulate air pollutant emissions from 1970 to 2012 (Crippa et al. 2018). The average ratios between 1970 and 1980 are generated and utilised to transform the NO_x emissions of EDGAR-HYDE into PM2.5 and PM10 emissions. Moreover, it must be noted that large uncertainties are existing in the activities related to World War II and thus the emission data before 1945 from EDGAR-HYDE might not be properly estimated.

Since the population data are only available for each LAU2 region rather than EMEP grid cells (see more details in Section 3.6.3), the concentration field for each year is aggregated at the LAU2-regional level and assumed to follow the normal distribution.

3.3 Simulation of concentration in micro-environments

As stated in Section 3.1, the micro-environment approach is applied in this thesis for simulating the individual exposure. A micro-environment can be described as a space with homogeneous pollutant concentration (Duan 1982). In this thesis, the micro-environments are classified as:

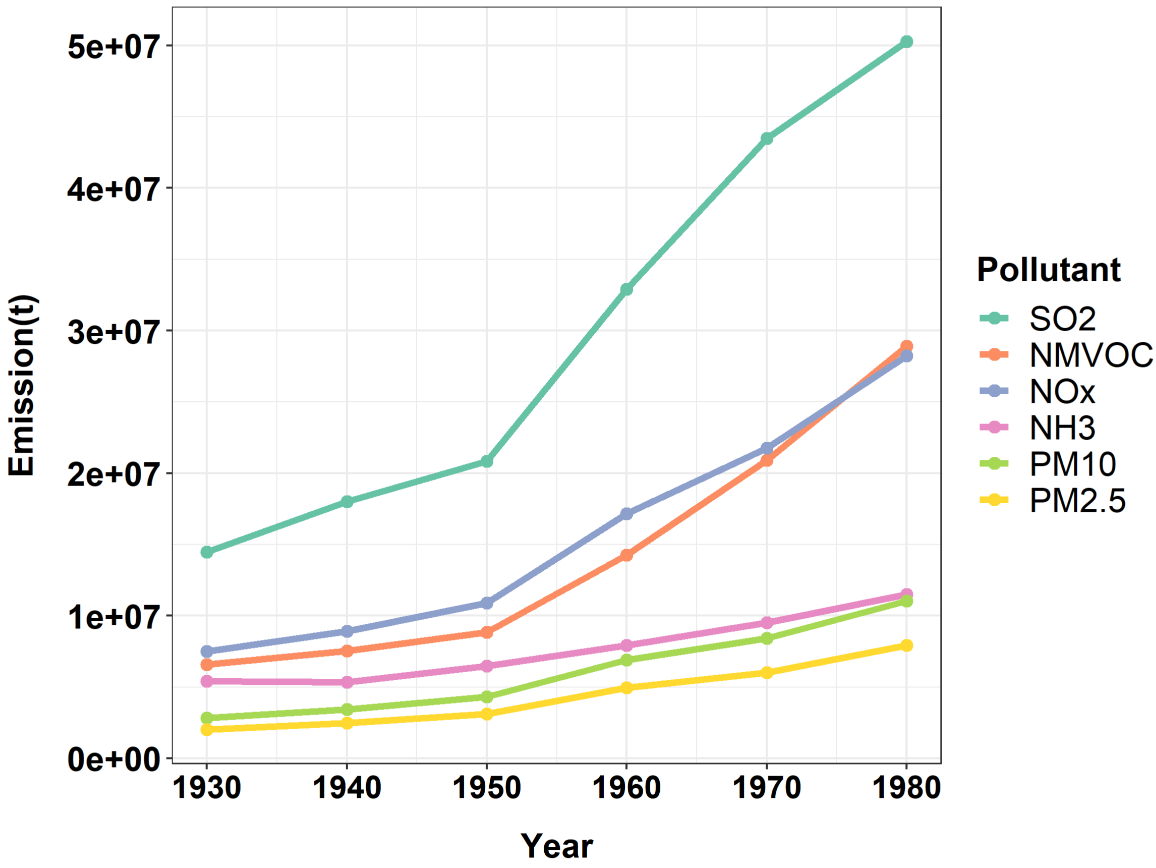

Figure 3-3: Total emissions for European countries from 1930 to 1980 according to the data from EDGAR-HYDE.

- home indoor
- work indoor
- school indoor
- other indoor
- outdoor
- travel/commute

3.3.1 Outdoor

For activities taken place outdoors, the ambient concentration is used. The process of generating the concentration fields for PM2.5 and NO_2 is described in detail in Section 3.2.

3.3.2 Travel/commute

To simulate the pollutant concentration in transport, a method from Gens (2012) using a traffic factor is applied (Equation 3-3).

$$C_{trans} = F_{ME} \times C_{out} \tag{3-3}$$

where:

C_{trans} = concentration in transport [μg m^{-3}]

C_{out} = ambient pollutant concentration [μg m^{-3}]

F_{ME} = traffic micro-environment factor

The values employed for the ME factors are described in detail in Section 4.5.

3.3.3 Indoor

It is evident that the indoor air quality is a dominant health concern due to the fact that: 1) indoor air is a mixture of the ambient air with the input of emissions from the additional indoor sources and 2) people spend most of their time indoors (Wallace 1996; Long et al. 2001; Hänninen et al. 2004). To simulate the indoor pollutant concentration, it is assumed that the air indoors is under steady state and a mass-balance model is applied (Dockery and Spengler 1981; Koutrakis et al. 1992; Wallace and Williams 2005) (see Equation 3-4).

$$C_{in} = \frac{C_{out} p \times AER + \frac{\sum_{i=1}^{n} E_i}{V}}{AER + k} \tag{3-4}$$

where:

C_{in} = indoor pollutant concentration [μg m^{-3}]

C_{out} = ambient pollutant concentration [μg m^{-3}]

p = penetration factor

AER = air exchange rate [h^{-1}]

k = decay rate [h^{-1}]

E_i = emission rate of source i [μg h^{-1}]

V = room volume [m^3]

Nowadays, the air handling unit (AHU) is becoming popular in commercial buildings as well as apartment buildings. As a part of the heating, ventilation and air conditioning (HVAC) system, the AHU is a device used to regulate and circulate air. It usually includes elements such as fan or blower, heating coil and filters (ASHRAE 2008). For indoor micro-environments equipped with an AHU, the mass-balance model was modified by Thornburg et al. (2001) as Equation 3-5:

$$C_{in} = \frac{C_{out}p \times AER + \frac{\sum_{i=1}^{n} E_i}{V}}{AER + k + \eta ND} \tag{3-5}$$

where:

η = filter efficiency

N = recirculated air exchange rate [h^{-1}]

D = duty cycle of the AHU

Equation 3-4 and 3-5 can be further simplified as Equation 3-6 (Hänninen et al. 2004):

$$C_{in} = F_{inf}C_{out} + C_{ig} \tag{3-6}$$

where:

C_{in} = indoor concentration [μg m^{-3}]

C_{out} = outdoor concentration [μg m^{-3}]

F_{inf} = infiltration factor

C_{ig} = concentration generated from indoor sources [μg m^{-3}]

As displayed in Equation 3-6, the overall concentration indoors comprises of two parts: the pollutant concentration that infiltrated from outdoors and the concentration formed by the indoor sources.

3.3.3.1 Infiltration

It has been proven by numerous studies that pollution infiltrated from outdoors plays a critical, even dominant role in the composition of the indoor concentration for both PM2.5 and NO_2 (Lee et al. 1998; Levy 1998; Abt et al. 2000; Burke et al. 2001; Fabian et al. 2012). The parameters for simulating the infiltration factor and their definitions are given in Table 3-2.

3.3.3.2 Street increment

Higher pollution concentration levels have been frequently monitored in urban street canyons (Vardoulakis et al. 2003; Rakowska et al. 2014; Zhong et al. 2016). Since street network is an essential component that affects the air quality in urban areas, numerous models have been developed to study the street increment (SI) (den Boeft et al. 1996; Berkowicz et al. 2008; Mensink and Cosemans 2008; Fallah-Shorshani et al. 2017). In this thesis, the SI data generated by Gens (2012) for PM2.5 are applied. The spatially resolved data were simulated with the hybrid dispersion modelling of Ortiz (2012), which used the average annual daily traffic, emission factor, wind speed, tree factor and road width as

Table 3-2: List of parameters for the infiltration factor and their definitions.

Parameter	Definition	Source
Air exchange rate	Measurement of how often per hour is the indoor air exchanged with outdoor air	Bouhamra et al. (1998)
Decay rate	Loss rate of air pollutants due to all processes	Gens (2012)
Filter efficiency	Removal efficiency of the AHU filter	Thornburg et al. (2004)
Recirculated air exchange rate	Measurement of how much air is removed from a space and reused in a given time period	ASHRAE (2007)
Duty cycle of AHU system	Fraction of time that the AHU fan is operating	Thornburg et al. (2001)

the input data. Similar to the concentration fields, the SI field is available in the format of EMEP grids. For NO_2 the SI data are not additionally developed since traffic-related information is included through the process of generating concentration fields (see Section 3.2.2).

According to the suggestion of Gens (2012), it is implied in this thesis that around 70% of the people in urban areas are affected by the street increment concentration. The remaining 30% of the population are supposed to live comparably remote from the busy streets and therefore the background concentration is used. Besides, the SI grid map generated is applied to all years due to the limitation of data.

3.3.3.3 Indoor sources

Indoor sources contribute considerably to the indoor concentration, especially with respect to PM2.5 (Spengler et al. 1985; Wigzell et al. 2000). Smoking has been revealed by numerous studies as one of the most important indoor sources (Koistinen et al. 2001; Liu et al. 2004; Lai et al. 2006; Loffredo et al. 2016). The emission from cigarettes is calculated by Equation 3-7:

$$E_{cig,i} = \frac{S_{cig} N_{cig,i}}{T_i} \tag{3-7}$$

Where:

$E_{cig,i}$ = emission rate for cigarettes in micro-environment i [µg h^{-1}]

S_{cig} = source strength per cigarette [µg]

$N_{cig,i}$ = number of cigarettes smoked in the micro-environment i per day

T_i = total time in micro-environment i during a day [h]

Studies have also reported cooking as another significant source for PM2.5 (Oezkaynak et al. 1996; Dennekamp et al. 2001; Wallace et al. 2003; Gerharz et al. 2009; Abdullahi et al. 2013). The emission from cooking can be calculated as:

$$E_{cooking,i} = \frac{S_{cooking} t_{cooking,i} (1 - CE)}{T_i} \tag{3-8}$$

Where:

$E_{cooking,i}$ = emission rate for cooking in micro-environment i [μg h^{-1}]

$S_{cooking}$ = source strength of cooking [μg min^{-1}]

$t_{cooking,i}$ = time of cooking activities in micro-environment i per day [min]

CE = capture efficiency of kitchen hood

T_i = total time in micro-environment i during a day [h]

Other indoor sources for PM2.5, including wood burning (Boleij and Brunekreef 1989; McDonald et al. 2000; Mandin et al. 2009), candles/incense (Sørensen et al. 2005; Stabile et al. 2012; Hu et al. 2012) and other activities (Long et al. 2000; Abt et al. 2000; Ferro et al. 2004) have also been identified and can be calculated by Equation 3-9, Equation 3-10 and Equation 3-11 respectively:

$$E_{wood,i} = \frac{S_{wood} t_{wood,i} H_{demand} V_i (1 - R_{removal})}{T_i} \tag{3-9}$$

Where:

$E_{wood,i}$ = emission rate for wood burning in micro-environment i [μg h^{-1}]

S_{wood} = source strength of wood burning [μg (kJ)$^{-1}$]

$t_{wood,i}$ = time of burning wood in micro-environment i per day [h]

H_{demand} = heat demand [kJ m^{-3}h^{-1}]

V_i = room volume of micro-environment i [m^3]

$R_{removal}$ = removal ratio of chimney

T_i = total time in micro-environment i during a day [h]

$$E_{candle,i} = \frac{S_{candle} t_{candle,i}}{T_i} \tag{3-10}$$

Where:

$E_{candle,i}$ = emission rate for candle in micro-environment i [μg h^{-1}]

S_{candle} = source strength of candle [μg min^{-1}]

$t_{candle,i}$ = time of burning candles in micro-environment i per day [min]

T_i = total time in micro-environment i during a day [h]

$$E_{other,i} = \frac{\sum_{j=1}^{n} S_{other,j} t_{other,i,j}}{T_i} \tag{3-11}$$

Where:

$E_{other,i}$ = emission rate for other activities in micro-environment i [μg h^{-1}]

$S_{other,j}$ = source strength of activity j [μg min^{-1}]

$t_{other,i,j}$ = time spending on activity j in micro-environment i per day [min]

T_i = total time in micro-environment i during a day [h]

Similar to PM2.5, indoor sources are the crucial factors affecting the individual exposure to NO_2 (Spengler et al. 1983; Levy 1998; Kornartit et al. 2010). Cooking, tobacco smoking and wood burning are included in this thesis according to the studies of Shima and Adachi (1998), Cyrys et al. (2000), García-Algar et al. (2003).

3.4 Time-activity patterns

3.4.1 Introduction of MTUS

As expressed by Equation 3-1, the simulation of exposure for an individual requires the duration of time that the person stays in each micro-environment. Data for the time-activity patterns are derived from the Multinational Time Use Study (MTUS), which is a concept initially proposed by Professor Jonathan Gershuny in the 1970s, aiming at creating a multi-national harmonised set of time use surveys (Fisher et al. 2012). Figure 3-4 shows the time-activity patterns for Spanish males and females between 13 to 75 years of age as an example.

The MTUS includes three data formats: harmonised simple file (HSF), harmonised aggregate file (HAF) and harmonised episode file (HEF):

- HSF

  Each row in the HSF file represents a diary, i.e. a 24-hour observation of the time-activity profile (Fisher and Gershuny 2016). The activities are classified into simplified 25 categories (see Table A-1 in the Appendix). Disadvantages of HSF are the limited coverage of demographic and socio-economic variables, as well as missing micro-environment information.

- HAF

  HAF has a similar data structure as HSF. However, the activities are further classified into 69 categories (see Table A-1). Besides, more detailed information on socio-economic status can be attained from the database.

- HEF

HEF maintains the same activity category as HAF but obtains a different data structure. Each row in the file records the time a diarist spends on a certain activity in the corresponding location, which means, precise information about the micro-environment is given.

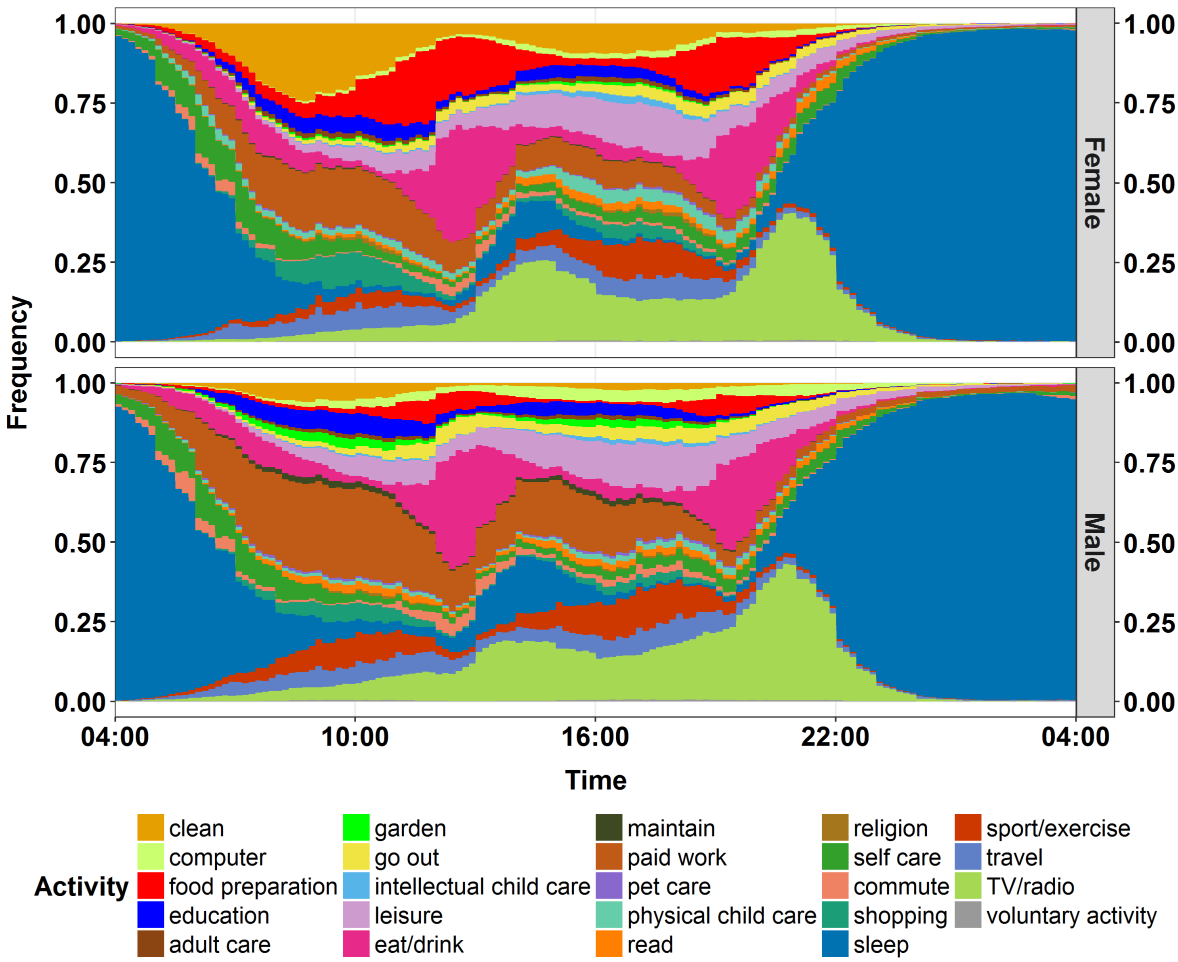

Figure 3-4: Daily activity profiles of females and males in Spain for 2010. The activities are aggregated into 25 categories by Fisher and Gershuny (2016) for simplicity (detailed information for categorization is given in Table A-1, category "missing data" is not displayed in the figure). The profiles (diaries) start at 04:00 and end at 04:00 the next day. The plot illustrates the frequency of a certain activity the females or males take part in for each minute of a day.

3.4.2 Data fusion: HEF and HAF

The data availability of MTUS for European countries is displayed in Table 3-3. Since no information is available for children under three years of age in Europe, diaries from US survey are adopted as substitution.

Even though the HSF data has the highest temporal and spatial coverage among the three data formats, its unavailability of location information hinders the modelling framework which relies on the micro-environment approach. To solve this problem, the author merges the HEF and HAF data based on the identifier variables, including household identifier, person/diarist identifier and diary identifier. The fused data profit from the structure of HEF with detailed micro-environment information. Simultaneously, more accurate socio-economic information is inherited from HAF.

Thus, the fused HEF and HAF are employed in this thesis for time periods when three formats of data can be accessed at the same time. For countries that only HSF data are available (especially for Denmark, Norway and Slovenia), the micro-environment information is assumed by the author with the existing HEF data. Details are given in Section 3.4.3.

Table 3-3: Data availability of the Multinational Time Use Study (MTUS) for several time periods. [- = no data available, ○= only HSF, ●=HSF, HAF and HEF].

Country	before 1975	1975-1979	1980-1984	1985-1989	1990-1994	1995-1999	2000-2004	2005-2010
Austria	-	-	-	-	●	-	-	-
Denmark	○	-	-	○	-	-	○	-
France	○	-	-	-	-	●	-	-
Germany	-	-	-	-	●	-	○	-
Italy	○	-	-	●	-	-	-	-
Netherlands	-	●	●	●	●	●	●	●
Norway	○	-	○	-	○	-	○	-
Slovenia/former Yugoslavia	○	-	-	-	-	-	○	-
Spain	-	-	-	-	●	●	●	●
United Kingdom	○	●	-	●	-	●	●	●
United States	-	-	-	-	●	-	-	-

3.4.3 Definition of micro-environments in MTUS

For the fused data, location codes are inherited from HEF and comprised of two variables, i.e. “eloc” and “inout”. The variable “eloc” describes in which location an activity occurs, while “inout” determines whether this activity takes place indoors, outdoors, or in transportation. These two variables are combined and reclassified to match the micro-environment categories described in Section 3.3 (see Figure 3-5).

For countries where only HSF can be accessed, the data gap for micro-environmental information must be filled. The author summarises the available HEF data and conducts the location an activity is most likely to take place. The “hypothesised location” is assigned to each activity in HSF and detailed information can be found in Table A-1.

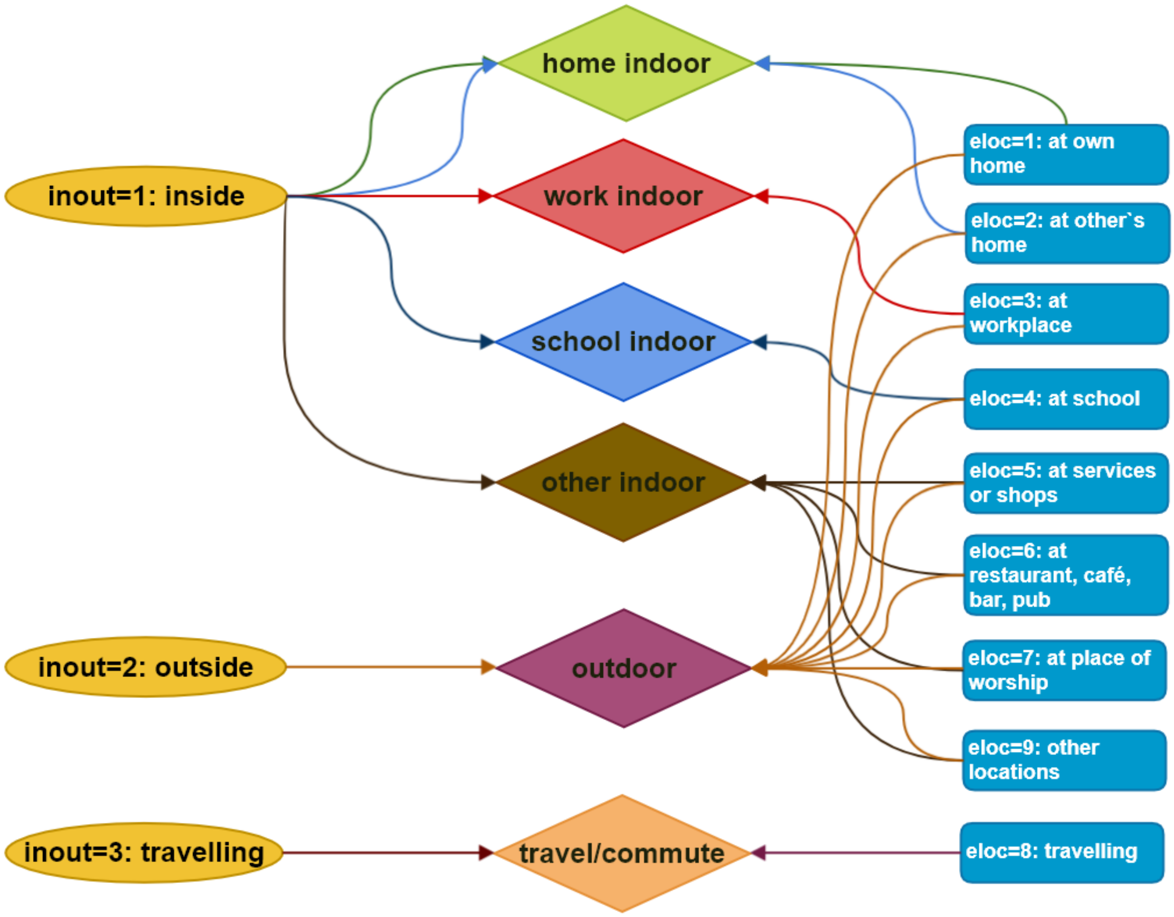

Figure 3-5: Definition and reclassification of location variables in the MTUS data. The variable “eloc” describes in which location the activity occurs (nine categories on the right side in the blue rounded rectangles) and “inout” (three categories on the left side in the yellow ellipses) determines whether the location is indoors, outdoors, or in transportation. The rhombuses in the middle represent the micro-environment categories for the exposure modelling.

3.4.4 Determination of time-activity patterns

As described above, the MTUS database also contains socio-demographic information, including gender, age, income for each respondent. The diaries of activity pattern in MTUS are assigned to subjects of population data with the same socio-demographic characteristics. As displayed by Table 3-3, the MTUS data cover only a limited number of countries. This leads to the problem that for many individuals no diaries can be matched. For these individuals, it is necessary to neglect some factors that are least influential for their time-activity patterns.

In order to solve this problem, a random forest model is built with the randomForest package of the R programming language (Liaw and Wiener 2014) to determine the order of the importance for the socio-demographic variables. Random forest is an algorithm for regression and classification that is built upon a multitude of simple decision trees to get more accurate prediction results (Ho 1995; Breiman 2001). The model uses the socio-demographic factors as independent variables, while the time spent "at home", "at work", "outside" and "travel/commute" are utilised as the dependent variables since people spend most of their time in these four micro-environments. Figure 3-6 shows the percentage increase in mean squared error (%IncMSE) of the predictor variables for this model, which explains 50% of the total variance. %IncMSE is an informative measure to identify the importance of independent variables: the higher the %IncMSE value is, the more influential the variable is. As displayed in the figure, "Country" and "Year of the survey" are the least important variables influencing the individual diaries. The similarity of time-activity patterns among different countries are confirmed by Hänninen et al. (2005) and Torfs et al. (2007) as well. In contrast, "Age", "Civil status" and "Employment status" are the key factors that significantly affect the time-activity pattern. For countries where no MTUS data are available, the author follows the method of Gens (2012) to adopt the data from the same geographical region as substitution and determine the diaries based on other socio-demographic variables. The EU27+2 countries are categorised into Southern, Eastern, Northwestern and Northern regions. The details can be found in Table 3-4.

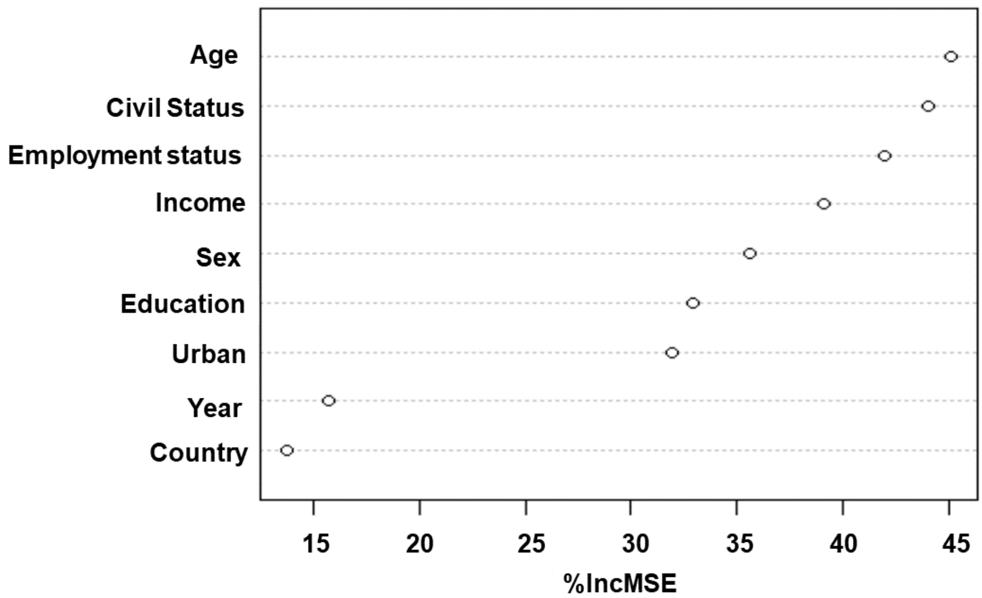

Figure 3-6: Percentage increase in mean squared error (%IncMSE) of the independent variables for the random forest model. The model explains 50% of the variance. As displayed in this figure, "Country" and "Year of the survey" are the least influencing factors, while "Age", "Civil status" and "Employment status" are the most important variables for the time-activity patterns.

Table 3-4: Classification of geographical regions for the EU27+2 countries.

Region	Country	Abbreviation
Eastern Europe	Bulgaria, Czech Republic, Estonia, Hungary, Latvia, Lithuania, Poland, Romania, Slovakia, Slovenia	EE
Northern Europe	Denmark, Finland, Norway, Sweden	NE
Northwestern Europe	Austria, Belgium, France, Germany, Ireland, Luxembourg, Netherlands, Switzerland, United Kingdom	NWE
Southern Europe	Cyprus, Greece, Italy, Malta, Portugal, Spain	SE

3.5 Life course trajectory model

With the methods introduced in the above sections, it is possible to evaluate the exposure to PM2.5 and NO_2 for a past year for an individual with certain features. With respect to lifelong exposure, however, it is not appropriate to sum up the exposure of each year since the socio-economic status of a person changes from birth to death. The most obvious examples are employment status and education level. This leads to a question: how can the exposure of each year in the past be incorporated properly into the lifelong exposure?

To address this issue, the life course trajectory model developed within the frame of HEALS project (Schieberle et al. 2017) is employed. This model was built with the functionalities provided by the TraMineR package for R programming language. TraMineR was developed by Gabadinho et al. (2009) with the aim of "knowledge discovery from event or state sequences describing life courses". The life course trajectory model in HEALS was established relying on the EU-SILC longitudinal data (see Section 3.6.3.1) for employment status and education level (details see Table A-2). With this model, it is possible to identify the trajectory patterns for all states of economic status and the transitions between the states retrospectively based on the status of a given year.

Figure 3-7 shows the life course trajectory of a German, male, full-time employee, aged 40 in 2010 as an example. The model predicts the probability of the economic status for each year of an individual's lifespan. As described above, this model is linked with the approach for exposure assessment to simulate the lifelong exposure.

This method for sequence analysis has been employed widely in social science (Widmer and Ritschard 2009; Svensson et al. 2015), however, this is the first time it has been applied in the field of environmental science.

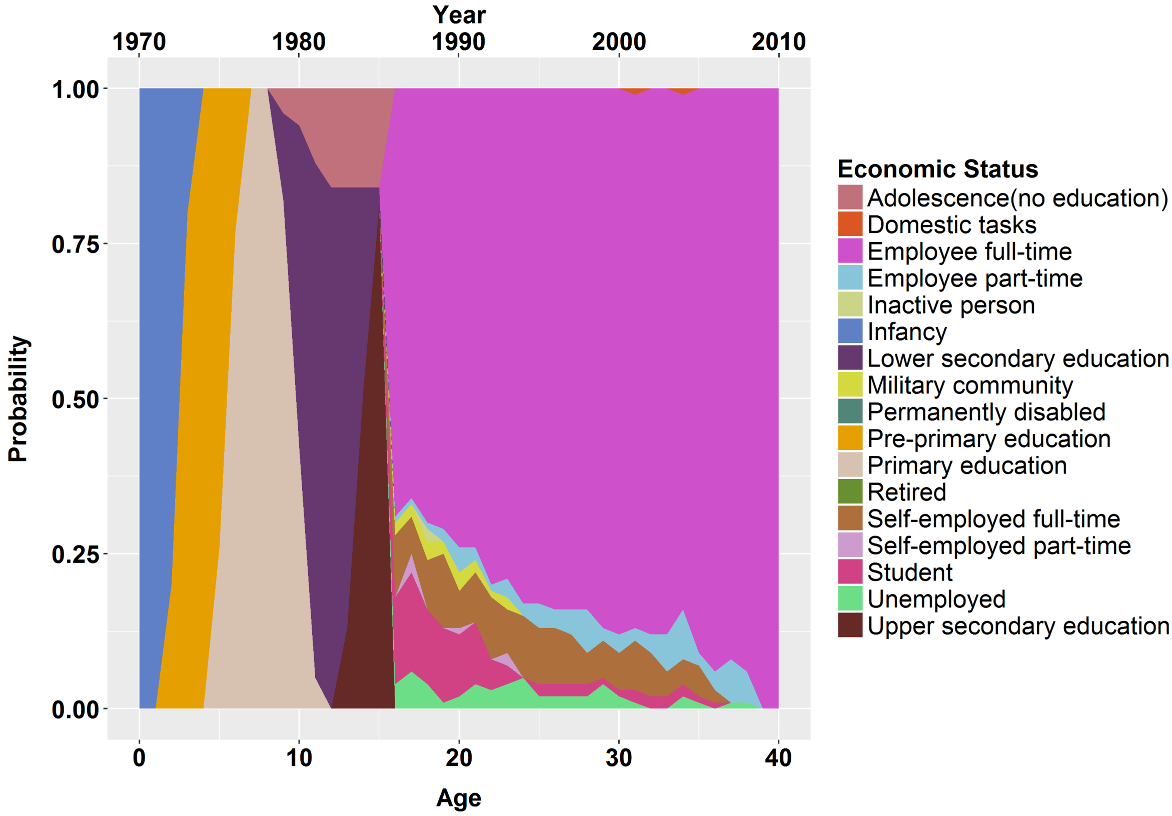

Figure 3-7: Life course trajectory of a German, male, employee full-time, age of 40 in 2010.

3.6 Health impact assessment, aggregation and valuation

3.6.1 Exposure response function

The most widely-applied method for estimating health impacts is relying on concentration response functions (CRFs, e.g. Anderson et al. 2004; Torfs et al. 2007). These functions indicate how many additional illnesses (e.g. chronic bronchitis) occur, or how many years of life are lost due to premature deaths, when 100,000 people are burdened with 1 μg m^{-3} higher background concentration for one year. As pointed out by Friedrich et al. (2011), the CRFs are linked with the ambient pollutant concentration, which is not able to reflect the actual exposure an individual suffers. Thus, it is necessary to transform the CRFs to exposure response functions (ERFs) to capture the real influence brought by the pollutants. For this thesis, the method from project HEIMTSA (Friedrich et al. 2011) is adopted to scale the CRFs to ERFs based on the relationship between background concentrations and personal exposures to PM2.5 and NO_2 experienced indoors from outdoor sources. Since most of the studies for CRFs were conducted after 1995, the values are developed with the average ratios between 1995 and 2015 at European level.

3.6.2 General procedure

The ERFs generated from CRFs are applied to calculate the health impacts (see Equation 3-12). The health impacts are considered for subgroups and subregions separately since the ERFs are age-structure dependent (see Table A-4) and the population data are available at the LAU2-regional level (see Section 3.6.3).

$$HE_i = \sum_{r=1}^{n} \sum_{g=1}^{m} EXP_{g,r} \times ERF_i \times POP_{g,r} \tag{3-12}$$

Where:

HE_i = health impact of endpoint i

$EXP_{g,r}$ = exposure of subgroup g in subregion r

ERF_i = exposure response function for health endpoint i

$POP_{g,r}$ = population for subgroup g in subregion r

To simplify the calculation, different health endpoints are transferred into one common metric, i.e. the disability adjusted life year (DALY). DALY is a summary measure calculated as the sum of the years of life lost (YOLL) due to premature mortality and the years lost due to disability (YLD) (WHO 2012). The DALYs for each endpoint are determined by multiplying the severity weight with the duration (Equation 3-13):

$$DALY = \sum_{i=1}^{n} HE_i \times DW_i \times DD_i \tag{3-13}$$

Where:

$DALY$ = total disability adjusted life years

HE_i = health impact of endpoint i

DW_i = severity weight for health endpoint i

DD_i = duration for health endpoint i

Another method is to transfer the health impacts into damage costs with monetary values with Equation 3-14.

$$DC = \sum_{i=1}^{n} HE_i \times MV_i \tag{3-14}$$

Where:

DC = total damage costs

HE_i = health impact of endpoint i

MV_i = monetary value for health endpoint i

3.6.3 Population data

The concentration or exposure simulated at the national or European level should be weighted with population data. Additionally, population data are necessary for estimating the health impacts due to PM2.5 and NO_2 as stated by Equation 3-12.

The population data are usually stratified by gender and age group. However, more detailed information for socio-economic structure is required since the discussion on exposure by population subgroup is one of the focuses in this thesis. To solve this problem, the following datasets and methods are introduced.

3.6.3.1 EU-SILC data

As a reference source for comparative statistics on income, poverty, social exclusion and living conditions, the EU-SILC (European Union Statistics on Income and Living Conditions) project was initiated by seven countries (six EU countries plus Norway) and implemented by other European countries gradually from 2003 onwards. Two types of data at European level are provided, i.e. the cross-sectional data and the longitudinal data (EUROSTAT 2013). The cross-sectional data contain comparable and detailed information of income, social exclusion, education and other living conditions over a given time period. The longitudinal data, which were applied for developing the life course trajectory model (see Section 3.5), focus on the changes at the individual level over a four-year time period.

In the EU-SILC data the comprehensive socio-economic information of the subjects is recorded. However, the addresses of the respondents are confidential. Instead, the region a subject lives in is given in the form of NUTS, which is a hierarchical classification of spatial units used for statistical production across the European Union (EUROSTAT 2015). The lowest territorial unit available in the EU-SILC data is NUTS2 and Figure 3-8 shows the regions in Spain as an example.

3.6.3.2 Other population data

The EU-SILC surveys provide data that are designed to represent the whole European population considering the socio-demographic structure. However, the surveys were conducted since 2003 and only a limited number of countries in the early years were available. In order to fill the data gaps, information from the following two sources is employed:

- UN data

  The United Nations (UN) provides population data[6] stratified by gender and 5-year age group each year from 1950 to 2015. However, the data are only available at country level and no further spatial stratification is available.

- LAU data

  Population data at the LAU2 level[7] from 1961 to 2011 for every 10 years are given by

[6]https://population.un.org/wpp/DataQuery/

[7]https://ec.europa.eu/eurostat/web/nuts/local-administrative-units

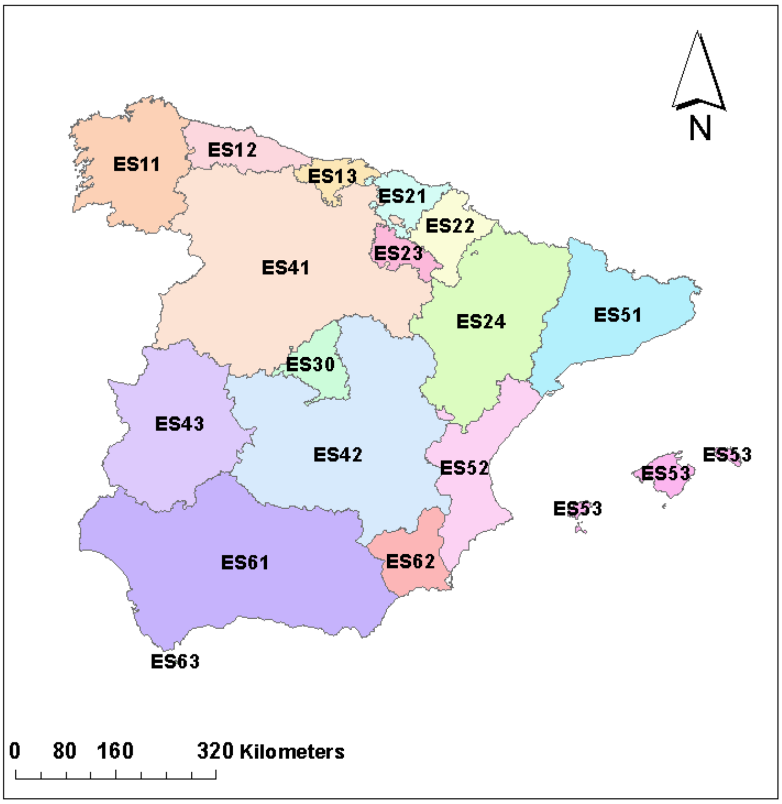

Figure 3-8: NUTS2 regions of Spain. Spain consists of altogether 19 NUTS2 regions. ES64 (Ciudad Autónoma de Melilla) and ES70 (Canarias) are not presented in this map due to the space limitation.

EUROSTAT. The LAU2 level, formerly the NUTS5 level, consists of municipalities or equivalent units in the EU27+2 countries. Figure 3-9 shows the LAU2 regions of Spain as an example. Nevertheless, no stratifications regarding age group or gender are made.

3.6.3.3 Rationale for synthesising the population datasets

In this thesis, the data from UN are used as the foundation of population data and further stratified by the author with other two datasets. Considering the temporal coverage, the data are manipulated for the following three time periods:

- After 2005

  The population data of UN are first distributed in each LAU2 region with the proportion that region takes in LAU dataset for the corresponding year. Since the LAU data are only available for every 10 years, the share of the population in each LAU2 region for the years in between are interpolated based on the assumption that the proportions are changing gradually every year.

  The population in each LAU2 region is further stratified into subgroups that are characterised by socio-economic variables (e.g. income level, education level) with

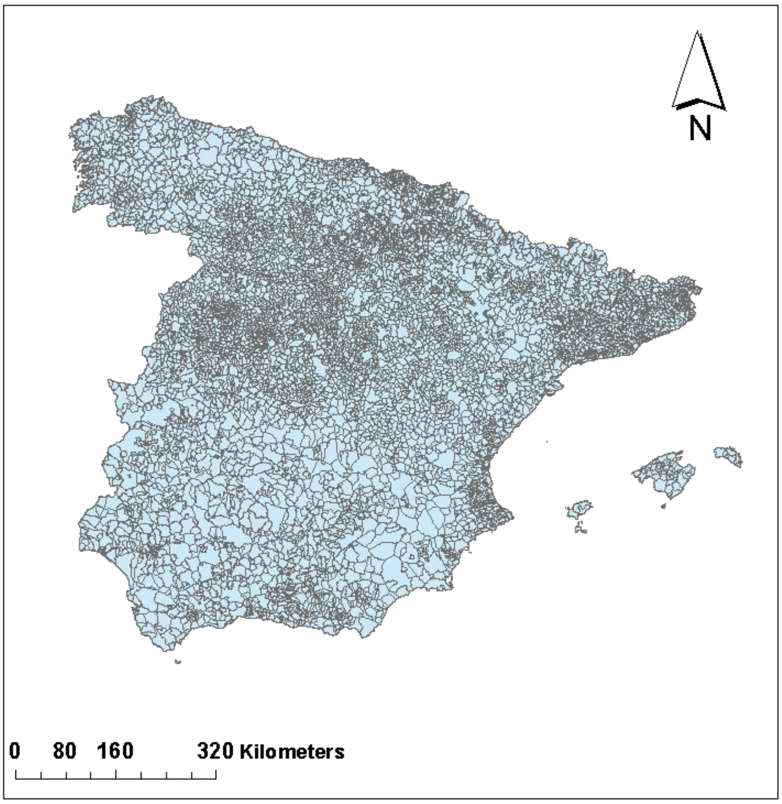

Figure 3-9: LAU2 regions, i.e. municipalities of Spain. Altogether there are 8118 municipalities in Spain. The municipalities in ES64 (Ciudad Autónoma de Melilla) and ES70 (Canarias) are not presented in this map due to the space limitation.

the information from EU-SILC cross-sectional data. The stratification is geographically based on two variables of the EU-SILC data: the territorial region, which is presented in NUTS2 and the degree of urbanisation. The degree of urbanisation creates a classification of all LAU2 regions into three categories, namely the "densely populated area", the "intermediated area" and the "thinly populated area" according to the typology of clusters (EUROSTAT 2018). Figure 3-10 shows the classification of degree of urbanisation in Spain as an example. Thus, each NUTS2 region can be further divided into three subregions based on the degree of urbanisation. The socio-demographic structure of the population subgroups in a LAU2 region is inherited from the NUTS2 subregion it belongs to.

This approach is applied for the population data after 2005 since the EU-SILC surveys have been implemented by the majority of the EU27+2 countries afterwards.

- Between 1961 and 2005

  For population data between 1961 and 2005, the method to distribute the UN data into LAU2 regions is same as for the data after 2005. Because of the data limitation, the EU-SILC data of 2005 is adopted to assign the population in LAU2 regions further into socio-economic subgroups.

- Before 1961

  As for the UN population assignment into LAU2 regions, the ratios are extrapolated retrospectively from the LAU data between 1961 and 1971. Likewise, the subsequent distribution into the population subgroups use the EU-SILC data of year 2005.

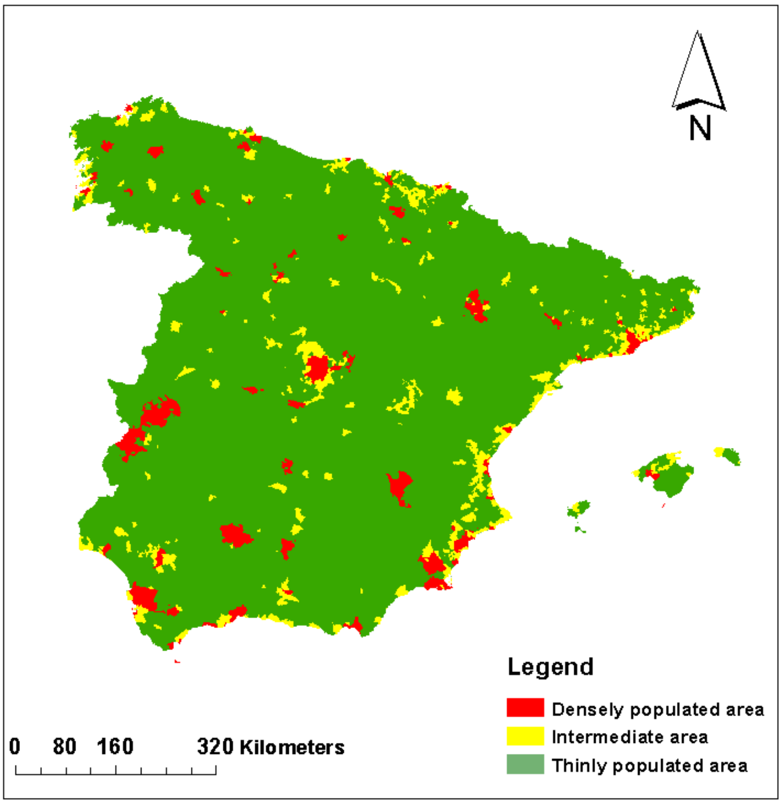

Figure 3-10: Classification of degree of urbanisation in Spain. The three categories of degree of urbanisation are defined by EUROSTAT as the “densely populated area” (red), the “intermediated area” (yellow) and the “thinly populated area” (green). The classification for ES64 (Ciudad Autónoma de Melilla) and ES70 (Canarias) are not presented in this map due to the space limitation.

3.7 Uncertainty assessment

3.7.1 Basics of uncertainty

Due to the complexity of the problems addressed, the integrated assessment is accompanied with uncertainties from different sources (Gabbert et al. 2010). Thus, it is necessary to perform the uncertainty analysis, which can be used to describe the range of the possible output results (Mountford et al. 2017).

Previous studies have established a variety of different classifications for uncertainties (van der Sluijs 1996). For example, Vesely and Rasmuson (1984) distinguished between data uncertainties, modelling uncertainties and completeness uncertainties, in which the modelling uncertainties could be further classified as the “incomplete understanding of the modelled phenomena” and the “numeral approximations used in mathematical representation”. Another highly-aggregated classification was made by Helton (1994) as the stochastic and subjective uncertainties. The stochastic uncertainties appear because the system under study can behave in different ways, while the subjective uncertainties arise due to the lack of knowledge (Rotmans and van Asselt 2001).

3.7.2 Methods for uncertainty analysis

Generally, the uncertainties can be assessed either qualitatively or quantitatively (van der Sluijs et al. 2005; Leroy and Froelich 2010). The qualitative uncertainty analysis describes or classifies the main uncertainties rooted in the study (Zhu et al. 2015). According to Salway and Shaddick (2016), the qualitative uncertainty analysis is implemented following three main steps: identification of the uncertainty sources, qualitative characterization of the uncertainty and reporting the qualitative uncertainty.

The advantage of this method is its flexibility and adaptability to a wide range of circumstances. However, the results of the qualitative analysis are less formalised, which hinders the comparison between different studies (IPCC 2005).

The quantitative uncertainty analysis appraises the range of uncertainties in the results numerically (Morgan et al. 1992). In spite of its drawback of less applicability, the quantitative analysis is the prevalent methodology for uncertainty assessment and widely applied in the field of environmental science (Fehr et al. 2003; Tainio et al. 2007; Schram-Bijkerk et al. 2009; Vlachokostas et al. 2009; Gillingham et al. 2018). The mainstream methods include the propagation of error, the Bayesian approach and the Monte Carlo simulation (IPCC 2000; Borsuk et al. 2004; Tran et al. 2009; Tainio et al. 2010; Mesa-Frias et al. 2013).

In regard to this thesis, the qualitative analysis is not capable of mathematically representing the uncertainties of the whole modelling framework and thus the quantitative analysis should be conducted. However, which method of quantitative assessment is more appropriate?

As pointed out by Gens (2012), the propagation of error is only applicable when the input parameters are normally distributed and the errors of these parameters are small. Nevertheless, most of the model input parameters in this thesis are assumed to be log-normally distributed with high uncertainty (see Chapter 4). The Bayesian approach incorporates several competing models and combines the prediction results with some statistical model techniques, including Bayesian computation, statistical model averaging approaches and likelihood measures (Mesa-Frias et al. 2013). Within the frame of this thesis, the Bayesian modelling is unnecessary since no competing models need to be formulated.

Thus, a Monte Carlo simulation is carried out for the uncertainty analysis. The Monte Carlo simulation is the most prevailing technique to propagate and analyse uncertainty (New and Hulme 2000; Vicari et al. 2007; Koornneef et al. 2010; Arunraj et al. 2013). The method relies on performing a great number of simulations using different sets of input parameters (Barreto and Howland 2005). The simulated results, i.e. exposure, health impacts and damage costs are available in the form of distribution and capable of taking account of the uncertainties of all the input parameters at the same time (Mesa-Frias et al. 2013).

3.7.3 Validation of residential concentration modelling

The input data for parameters of the mass-balance model show a large variability according to different references. Thus, it is necessary to implement a thorough data gathering and validate the model results with observation data.

The measurement data from the EXPOLIS study are chosen to validate the model for the concentration of residential buildings. As introduced in Section 2.1, EXPOLIS was a European multi-centre study for the measurement of air pollution exposures for the urban populations. The selected urban areas include Athens, Basel, Grenoble, Helsinki, Milan and Prague (Jantunen et al. 1998). During the field work from the summer of 1996 to the winter of 1997-98, the concentration for PM2.5, NO_2, VOCs and CO of home indoor and outdoor were monitored simultaneously.

The measurements of outdoor concentrations, together with time-activity patterns from MTUS and other model parameters, are used as the input data for the mass-balance model (see Equation 3-1). The output of model simulation is compared with the corresponding indoor measurement. If the comparison shows a low agreement for the sample points, a new round of data collection and fusion is conducted to adjust the values applied for the input parameters. This process continues until a high match is reached. Figure 3-11 presents the flow chart of the validation process.

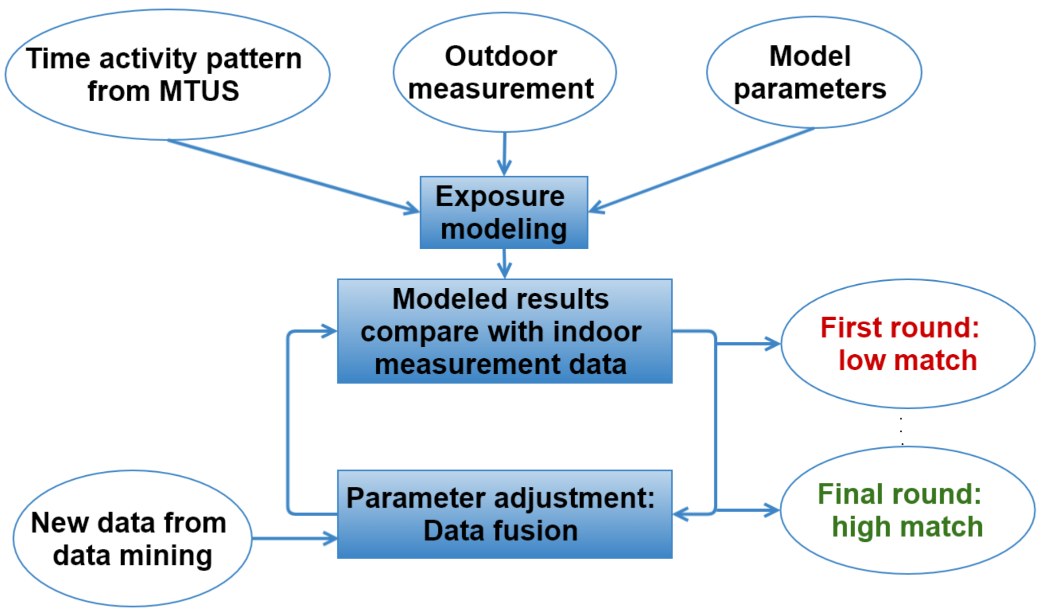

Figure 3-11: Flow chart of the validation process for the residential concentration modelling.

4 Data

4.1 Concentration maps

As described in Section 3.2, the concentration fields since the 1930s are simulated with the data displayed in Table 3-1. Figure 4-1 and Figure 4-2 show the annual average PM2.5 and NO_2 concentration for 2009 generated by the author following the interpolation methods of Horálek et al. (2007) and Horálek et al. (2017). For concentration fields between 1980 and 2000, the multiplicative bias adjustment method is applied in this thesis. Figure 4-3 and Figure 4-4 show respectively the concentration fields of PM2.5 and NO_2 after multiplicative bias adjustment for 1980. For concentration maps before the 1980s, the EDGAR-HYDE emission data are utilised as the input data for EcoSense model and the concentration fields simulated are subsequently adjusted with the multiplicative factors. The concentration data of PM2.5 and NO_2 for 1970 are displayed by Figure 4-5 as an example.

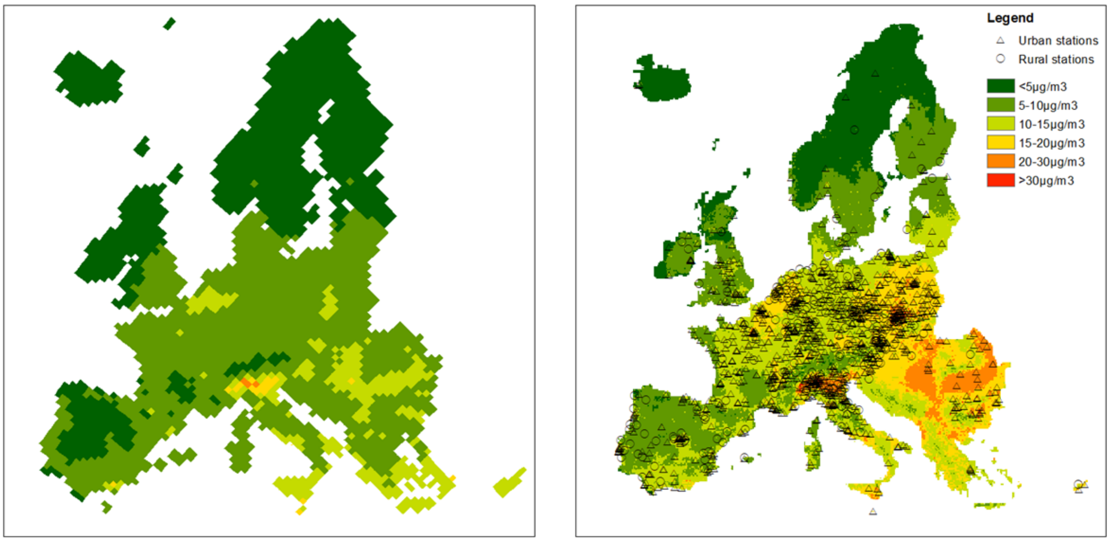

Figure 4-1: Maps showing the annual average PM2.5 concentration (in µg m^{-3}) for 2009 from EMEP/MSC-W (left) and after the interpolation (right). The right map is generated by the author following the method of Horálek et al. (2007).

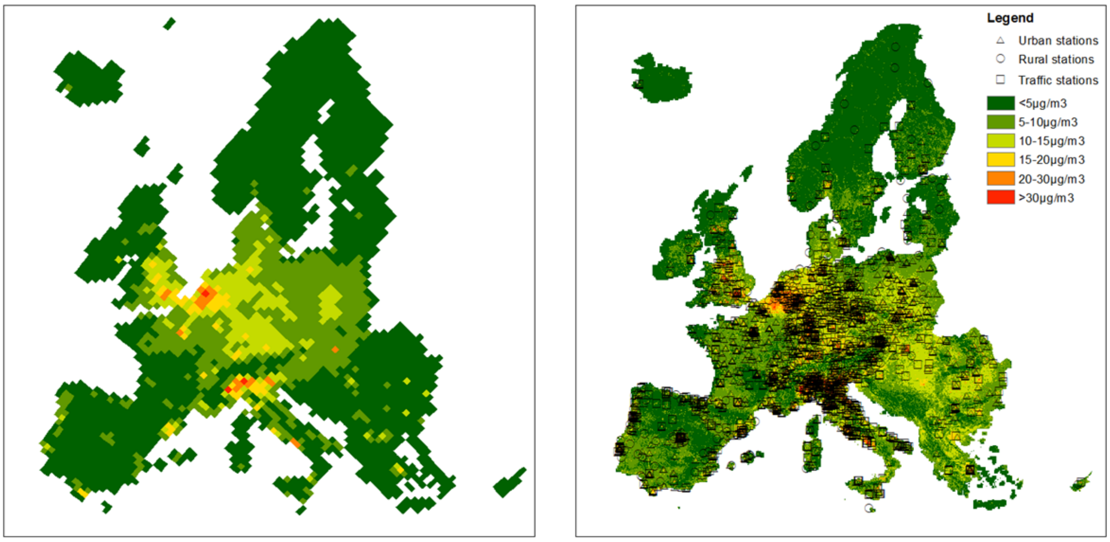

Figure 4-2: Maps showing the annual average NO_2 concentration (in µg m^{-3}) for 2009 from EMEP/MSC-W (left) and after the interpolation (right). The right map is generated by the author following the method of Horálek et al. (2017).

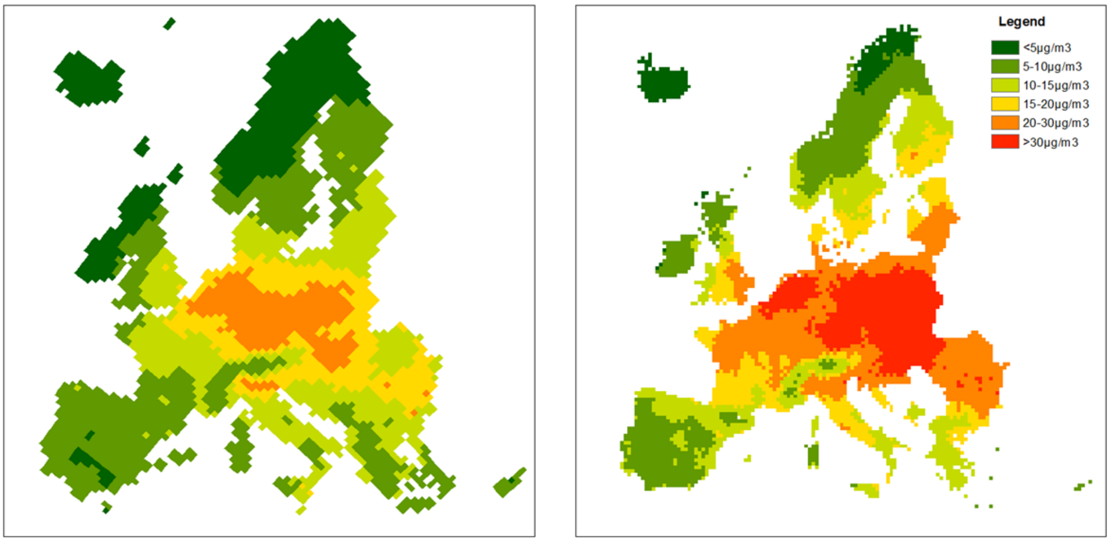

Figure 4-3: Maps showing the annual average PM2.5 concentration (in µg m^{-3}) for 1980. The left one represents the data from EMEP/MSC-W and the right one is the result after multiplicative adjustment by the author.

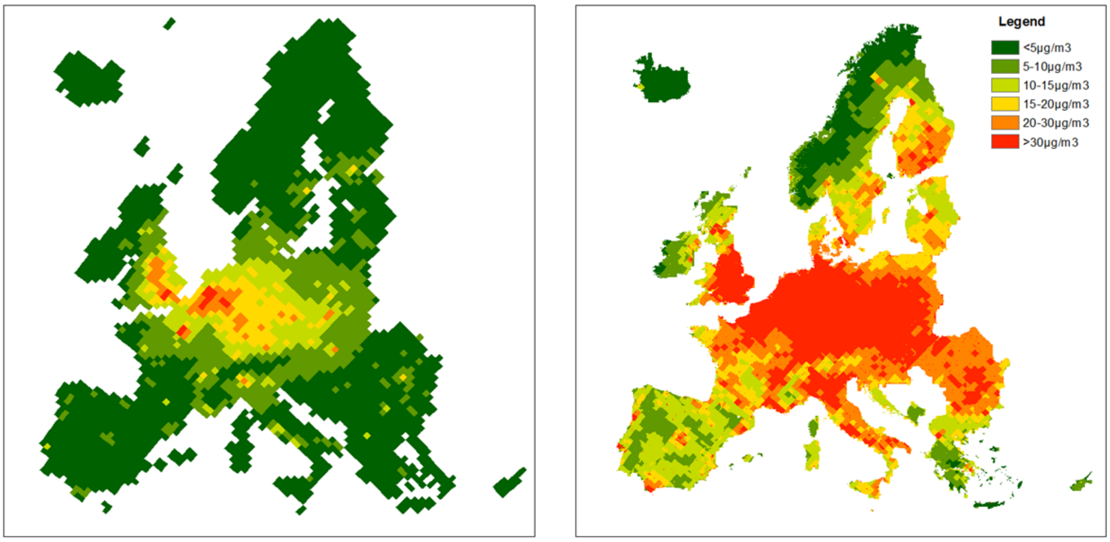

Figure 4-4: Maps showing the annual average NO_2 concentration (in µg m^{-3}) for 1980. The left one represents the data from EMEP/MSC-W and the right one is the result after multiplicative adjustment by the author.

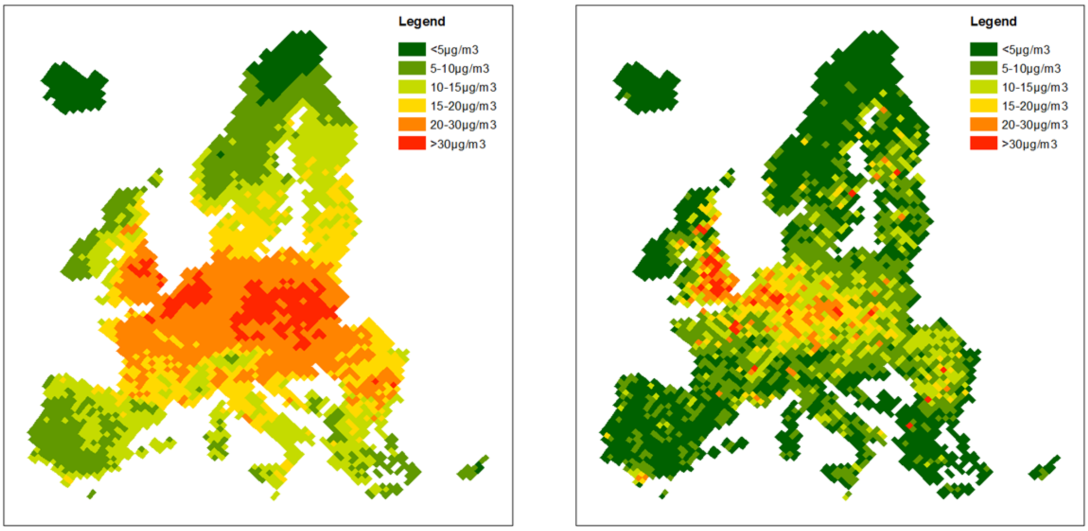

Figure 4-5: Maps showing the annual average PM2.5 (left) and NO_2 (right) concentration (in µg m^{-3}) for 1970. Both maps are simulated by the EcoSense model with EDGAR-HYDE emissions as the input data. The generated concentration fields are subsequently adjusted with multiplicative factors by the author.

4.2 Information for infiltration

4.2.1 Distribution of ventilation types

Ventilation is a crucial factor for estimating the indoor air quality. The building ventilation varies depending on climate, building construction characteristics and building age (Hodas et al. 2016). Numerous studies have reported the impacts of ventilation system on exposure to air pollutants, especially for particles (Fisk et al. 2002; Hänninen et al. 2005). The classification of ventilation systems is very complex in real life. According to Seppänen et al. (2012), the ventilation systems in Europe can be assigned to two main categories, i.e. natural ventilation and mechanical ventilation, which can be further divided into several subsystems.

Natural ventilation system is very common in old buildings. For such systems, the ventilation is mainly driven by infiltration and exfiltration of air through opening windows or doors. However, leakage can still take place via cracks and other unintentional openings in the building envelope since the buildings are not airtight. For such buildings, the penetration from outdoors is close to unity (Oezkaynak et al. 1996) and the particles decay solely due to deposition or other processes at relatively slow speed (Wallace 1996).

Since the last few decades, energy saving has become the driving force for building renovation. Old buildings with natural ventilation systems are retrofitted with airtight windows and doors to avoid unnecessary energy losses. However, the influence of ventilation has been usually ignored. Münzenberg et al. (2003) performed a study in unoccupied buildings with closed windows to measure the air change rates. The results showed that 90% of the naturally ventilated buildings with retrofitted airtight windows have an insufficient air exchange rate below 0.5 h^{-1}. The poor ventilation can result in high concentrations of indoor contaminants and affect the health of the people staying indoors (Sireesha 2017).

To solve the air quality problem brought by the insulation, mechanical ventilation systems can be installed. Such systems depend on powered air movement devices that exchange the indoor air with the ambient air. They usually are accompanied with heat recovery and particle filters. Applications of this system can be found in residential houses, apartments, schools, kindergartens and offices.

Compared to the category introduced above, a more advanced subcategory of mechanical ventilation is equipped with a supply and a return air fan to recirculate parts of the air. This is necessary, if the rooms have to be air conditioned, i. e. not only heated, but also cooled with the mechanical ventilation system, as then the airflow rate has to be much larger than with a pure ventilation system. A very common component of such a system is the AHU (air handling unit), which is an assembly usually containing fans, heating and cooling elements, filters and other equipments for realising heat recovery, air circulation and humidifying or dehumidifying. Firstly, the pollutants from indoor sources are removed more effectively than in renovated houses because of the sufficiently high air exchange rate that is achieved. Secondly, the removal of particles as well as other air pollutants can be accelerated by recirculating the indoor air through the filters in the AHU (Fisk et al. 2002; Nazaroff and Weschler 2004; Park et al. 2014). Such systems have been widely applied in offices and schools, and are becoming more popular in residential buildings.

In this thesis, the buildings are classified into four categories: naturally ventilated (“natural”), “retrofitted”, mechanically ventilated (“mechanical”) and “AHU”. Contrary to the “natural” buildings, the “retrofitted” buildings are insulated but without additional mechanical ventilation systems or air conditioners. The “mechanical” buildings are assumed to be equipped with heat recovery, filters and extract and/or supply fans. Compared to the “mechanical” system, the “AHU” system is able to recirculate a larger part of the air. The prevalence of different ventilation types varies by geographic scales, building types and construction years (Litiu 2012). In the metropolitan area of Helsinki less than 1% of the residences constructed before 1990 had supply air filters, whereas the majority of the single-family houses built after 2000 were equipped with a mechanical supply and exhaust ventilation system (Hänninen et al. 2005). By contrast, almost 100% of the existing residential buildings in Bulgaria were still naturally ventilated, while 20% of the office buildings were equipped with mechanical ventilation (Seppänen et al. 2012).

Thus, it is necessary to determine the prevalence of building types for European countries. The data are generated based on several pieces of information: Seppänen et al. (2012) provide data for the share of “mechanical” and “AHU” buildings stratified by building type and age; the age structures of the residential and non-residential buildings are derived from UN-ECE (2006) and Birchall et al. (2014) at country level; the degree of retrofitting (including “AHU”, “mechanical” and “retrofitted” buildings) is given by Blesl et al. (2011) and Kranzl et al. (2016). Due to the lack of information, two assumptions are made: 1) the degree of “AHU”, “mechanical” and “retrofitted” buildings is increasing gradually each year and 2) for countries where no data are available, the average value from the same geographic region is adopted following the method described in Section 3.4.4.

4.2.2 Air exchange rates

Air exchange rates (AER) are one of the most crucial factors affecting the indoor air quality and has been extensively discussed in Europe. The European standards on ventilation were published by European Committee for Standardization (CEN), however, the AERs of buildings in reality were often found to be below the regulation values (Dimitroulopoulou 2012; Asikainen et al. 2013). Several studies have also proven the association between inadequate ventilation and negative health impacts (Godish and Spengler 1996; Wargocki et al. 2002).

4.2.2.1 Residential buildings

The conditioning of indoor air for residential buildings has been intensively studied and a considerable variability has been reported (Ruotsalainen et al. 1991; Hänninen et al. 2011; Brelih 2012). For naturally ventilated dwellings, the data from EXPOLIS (Hänninen et al. 2004), which estimated the air exchange rate for residences in Athens, Basel, Helsinki and

Prague, are adopted. The method of Gens (2012) is applied to assign and extrapolate these values to four geographical regions (Southern: Athens, Eastern: Prague, Northwestern: Basel, Northern: Helsinki). Detailed information is summarised in Table 4-1.

For "retrofitted" buildings, measures are taken to reduce the energy consumption for heating or cooling (Fang et al. 2014; Aditya et al. 2017; Simona et al. 2017). Thermal insulation efficiently protects the building envelop from energy loss, however, the increased tightness slows down the air exchange with outside (Elmroth and Lögdberg 1980; Laussmann and Helm 2011). As a result, the retrofitted dwellings without additional mechanical ventilation often report an insufficient air exchange rate. In this thesis, an average AER of 0.35 with 0.15 h^{-1} as standard deviation is assumed based on the data reported by Kvisgaard et al. (1985), Kurnitski et al. (1996) and Münzenberg et al. (2003).

The air exchange rate of the "mechanical" dwellings is supposed to be higher than that of the "retrofitted" dwellings, as the main purpose of mechanical ventilation is to ensure a healthy environment and thus a sufficient air exchange rate. Based on the data from Kvisgaard et al. (1985), Bornehag et al. (2005) and Kurnitski et al. (2007), an average AER of 0.50 with 0.30 h^{-1} as standard deviation is assumed for "mechanical" dwellings.

For the buildings that are equipped with the AHU, most of the air is recirculated through the filters. However, a certain amount of the air is exchanged with outdoor to achieve a sufficient air quality. Thus the air exchange rate is assumed to be similar to the values applied for the "mechanical" buildings. As mentioned above, the indoor pollutant concentration is simulated with the model developed by Thornburg et al. (2001) (see Equation 3-5). The additional effect of the air recirculation is described in detail in Section 4.2.5.

The air exchange rates for dwellings are assumed to be log-normally distributed according to the study of Ryan et al. (1986), Moschandreas and Saksena (2002) and Kruize et al. (2003).

Table 4-1: Values of air exchange rate for micro-environment "home" categorised as "natural", "retrofitted", "mechanical" and "AHU" buildings (h^{-1}).

Micro-environment	Region	Building type			
		"natural"	"retrofitted"	"mechanical"	"AHU"
home	NWE	0.83 (± 0.46)	0.35 (± 0.15)	0.50 (± 0.30)	0.50 (± 0.30)
	SE	1.29 (± 1.09)			
	EE	0.75 (± 0.43)			
	NE	0.81 (± 0.85)			

4.2.2.2 Offices

Air conditioning in offices is widely discussed due to the fact that employees spend a large portion of time at workplaces. The AER for offices in European countries has been measured by various studies (Bluyssen et al. 1996; Wargocki et al. 2002; Zuraimi et al. 2006; Zaatari et al. 2014a). These studies have revealed that the variability of AER is driven by factors including weather conditions and ventilation systems. Data of AER for "natural" and "mechanical" offices are adopted from OFFICAIR (Dimitroulopoulou and Bartzis 2014), which was a European collaborative project focusing on the air condition in office buildings. The values and distribution proposed are shown in Table 4-2 and applied for each European country. No distinction is made between the countries since office buildings are one of the most standardised building types (Sakellaris et al. 2016).

For the "retrofitted" offices, the air exchange rates are supposed to be lower than those of the "natural" offices as a result of the insulation. The values applied are summarised based on the data from Sundell et al. (1994) and Ní Riain et al. (1999) for the under-ventilated and low-energy offices. For "AHU" offices, the data from Thornburg et al. (2001) are applied.

Table 4-2: Values of air exchange rate for micro-environment "work" categorised as "natural", "retrofitted", "mechanical" and "AHU" buildings (h^{-1}).

Micro-environment	Building type			
	"natural"	"retrofitted"	"mechanical"	"AHU"
work	(0.1, 0.6, 1.8), triangular	(0.1, 0.3, 0.8), triangular	(0.5, 1.4, 5.0), triangular	0.5 (± 0.3), log-normal

4.2.2.3 Schools

Air exchange rates in school have become an important topic (Lugg and Batty 1999; Coley and Beisteiner 2002; Blondeau et al. 2005; Canha et al. 2013) in Europe since 1) children spend pretty much time in school and 2) children are more vulnerable to air pollutants compared to adults (WHO 2005a; Guo et al. 2008a). For this thesis the data from project SINPHONIE (Schools Indoor Pollution and Health: Observatory Network in Europe) are adopted. SINPHONIE was a complex research project that focused on the air quality in schools and kindergartens. The AER, classroom area and ceiling height were measured for 23 countries in Europe. For the countries where data are missing, the average value from the same region is adopted following the method described in Section 3.4.4. The AER of schools is assumed to follow a triangular distribution and the data are summarised in Table A-3. For schools, no distinction is made for AER between the four building types because 1) SINPHONIE partners were requested to choose schools that were representative of the building stock in their country in terms of typology, construction techniques and age

(Csobod et al. 2014) and 2) data for building age structure of schools are rare and difficult to acquire.

4.2.3 Penetration factor

The penetration factor determines the ratio of the air pollutant that passes from outdoor to indoor environment (Meckler 1995; Deshpande et al. 2009). As one of the most relevant parameters that affect the indoor air quality, the penetration efficiency has triggered the worldwide concern (Thatcher and Layton 1995; Mosley et al. 2001; Liu and Nazaroff 2003; Thatcher et al. 2003). For PM2.5, particle size, geometry of the cracks in buildings, ventilation patterns, building fabrics and use of filtration lead to the enormous range of penetration factor (Tung et al. 1999; Liu and Nazaroff 2001; Thornburg et al. 2001; Chao et al. 2003; Chen and Zhao 2011). Thatcher and Layton (1995), Oezkaynak et al. (1996) and Wallace (1996) have reported that the penetration factor was "very close to unity", while Tung et al. (1999) and Long et al. (2001) calculated the penetration efficiency of less than one.

In this thesis, the penetration factor of PM2.5 for "natural" and "retrofitted" buildings is assumed to be 0.95 ($\pm$ 0.30) following a log-normal distribution. For the "mechanical" buildings, the effect of installed filters should be considered.

The filters are able to remove particulates and gases from air. Fisk et al. (2002) reported that up to 80% of the ambient fine particles can be eliminated through the filters. In reality, however, the performance of filters is influenced by various factors including particle size, the life of the filters as well as the behaviour of inhabitants. For example, the efficiency of filters is greatly reduced if the doors and windows are kept open.

Based on the information gathered from literature review (see Section 4.2.5.1), the filter efficiency for PM2.5 is assumed to be uniformly distributed between 0.10 and 0.70. Also considering the working time of the mechanical system of around 50% of the year (see Table 4-5), a penetration factor of 0.75 ($\pm$ 0.20) is applied for "mechanical" buildings.

Compared to the "mechanical" ventilation, the AHU system filters not only the particles from the ambient environment, but also the pollutants originated from indoor sources due to the air recirculation. For the exchanged air, the same penetration factor for "mechanical" buildings is applied for "AHU" buildings. The effect of filter on the recirculated air is explained by the AHU related filters (see Section 4.2.5) based on the adjusted form of the mass-balance model (Equation 3-5).

For NO_2, a penetration efficiency of 1 is employed for the "natural" and "retrofitted" buildings since gaseous pollutants often enter the buildings without absorption (Sirén 1993; Tung et al. 1999; Fabian et al. 2012). For "mechanical" and "AHU" buildings, a large variance of filter efficiency has also been revealed by different researches. Based on the information collected (see Section 4.2.5.1), a penetration efficiency of 0.70 ($\pm$ 0.12) is applied for the exchanged air.

The values applied for penetration factor are summarised in Table 4-3.

Table 4-3: Values of penetration factor for PM2.5 and NO_2 (h^{-1}).

Pollutant	Building type			
	“natural”	“retrofitted”	“mechanical”	“AHU”
PM2.5	0.95 (± 0.30), log-normal	0.95 (± 0.30), log-normal	0.75 (± 0.20), log-normal	0.75 (± 0.20), log-normal
NO_2	1, consistent	1, consistent	0.70 (± 0.12), log-normal	0.70 (± 0.12), log-normal

4.2.4 Decay rate

Even in buildings without additional ventilation units, the particles are eliminated from the indoor environment slowly due to the deposition and other decay processes (Wallace 1996; Hänninen et al. 2005). Like AER and penetration efficiency, the decay rate, or the deposition rate, is a vital parameter determining the indoor pollutant concentration (Ferro et al. 2004; Bouilly et al. 2005; Hussein et al. 2005; Chen et al. 2006; Lai and Chen 2007a; el Hamdani et al. 2008). Studies have revealed that the deposition rate is sensitive to factors including particle size, airflow pattern, wind speed, indoor and outdoor temperatures, turbulence level and properties of indoor surfaces (Abt et al. 2000; Wallace et al. 2004; Chen and Zhao 2011).

For buildings that are naturally ventilated, the author follows the summary of Hänninen et al. (2004) and the value of the PTEAM (Particle Total Exposure Assessment Methodology) study is employed (Oezkaynak et al. 1996). According to the literature review of Gens (2012), there is a slightly positive association between the deposition rate and AER. For the “retrofitted” buildings, an average value of 0.25 h^{-1} is adopted. For the “mechanical” and “AHU” buildings, a slightly lower deposition rate of 0.30 is applied (see Table 4-4).

NO_2 is removed from air by the surface reaction, which is strongly impacted by the materials present in a building. Thus, a large variability of measured values has been observed as well (Borrazzo et al. 1987; Spicer et al. 1989). Generally, NO_2 is removed faster from the indoor air compared to PM2.5 (Yamanaka 1984; Emmerich and Persily 1996; Shuai et al. 2013).

What are the differences of decay rate for NO_2 among the four types of buildings? Few studies have paid attention to the influence of ventilation systems on NO_2. Whitmyre and Pandian (2018) studied the indoor air quality of “tight” energy-efficient homes with AER less than 0.35 h^{-1} and summarised the decay rate ranging from 0.25 to 1 h^{-1}. Yang et al. (2004) reported a positive association between the AER and decay rate regarding the data collected in two cities in South Korea. Hence, an average value of 0.87 h^{-1} is employed for “natural” buildings following the suggestion of Fabian et al. (2012). As for the “retrofitted” buildings, a lower average value of 0.65 h^{-1} is applied (see Table 4-4). For the “mechanical” and “AHU” buildings, a higher average decay rate of 0.75 h^{-1} is employed.

Table 4-4: Values of decay rate for PM2.5 and NO_2 (h^{-1}).

Pollutant	Building type			
	"natural"	"retrofitted"	"mechanical"	"AHU"
PM2.5	0.39 (± 0.10), log-normal	0.25 (± 0.10), log-normal	0.30 (± 0.15), log-normal	0.30 (± 0.15), log-normal
NO_2	0.87 (± 0.30), log-normal	0.65 (± 0.15), log-normal	0.75 (± 0.25), log-normal	0.75 (± 0.25), log-normal

4.2.5 AHU parameters

As a part of the HVAC system, the AHUs improve the air quality in buildings since they recirculate a large amount of air and filter the pollutants with additional air cleaners or devices (Hanley et al. 1994; Emmerich and Persily 1996; Noh and Hwang 2010). Understanding the features of AHU is vital for simulating the quality of indoor air. Although Howard-Reed et al. (2003), MacIntosh et al. (2008) and Marsik and Johnson (2008) developed models for dwellings with HVAC system, the settings behind these methods were rather explicit and difficult to apply. With the aim to simplify the study, the adjusted mass-balance model from Thornburg et al. (2001) is used as stated by Equation 3-5. The detailed information is explained in the following sections.

4.2.5.1 Filter efficiency

The filter efficiency defines the proportion of pollutants that can be removed by the filters installed. For particles, Hinds (1999), Stephens et al. (2011), el Orch et al. (2014), Zaatari et al. (2014b) and Stephens (2018) have identified a wide range of filter efficiency, which is dependent on the particle size and minimum efficiency reporting value (MERV). For example, Azimi et al. (2014) reported an efficiency of over 99% for an HEPA (High-efficiency particulate absorber) filter, while the study from Fisk et al. (2002) measured a removal ratio of less than 10% for a filter with poor performance. In this thesis it is assumed that for particles the efficiency ranges from 10% to 70% (see Table 4-5).

For NO_2, very little information of AHU performances in reality is available. Yoo et al. (2015) examined the performance of an activated carbon filter, which can remove the NO_2 effectively (up to 96%). Alain and Dominique (2007) have reported a relatively high initial efficiency from 60% to 90% for the first year and a sharp decrease to only 25% after 10.5 months. Considering the poor performance of overuse, the filter efficiency for NO_2 is assumed to follow a uniform distribution ranging from 25% to 90% (see Table 4-5).

4.2.5.2 Recirculated air exchange rate

The recirculated air exchange rate is a substantial parameter in mass-balance model referring the AHU. Recirculation rates are a function of factors including system airflow rates and house volume (Stephens et al. 2011). A wide range of 0.67h^{-1} to 24 h^{-1} has been used in the models and experiments by Riley et al. (2002), Klepeis and Nazaroff (2006), Zuraimi et al. (2007) and Waring and Siegel (2008). In this thesis the recirculation rates are assumed to be log-normally distributed with a mean value of 5 h^{-1} and a maximal value of 25 h^{-1} (see Table 4-5).

4.2.5.3 Duty cycle of “AHU” systems

The duty cycle, i.e. the fractional of time that the AHU is operated, has been investigated or estimated by James et al. (1997), Thornburg et al. (2004), Klepeis and Nazaroff (2006), MacIntosh et al. (2010) and Kassas (2015). The data applied for the duty cycle of AHU are presented in Table 4-5.

Table 4-5: Values for AHU parameters.

Pollutant	Filter efficiency	Recirculated air exchange rate (h^{-1})	Duty cycle of AHU
PM2.5	0.10-0.70, uniform	5 (± 2), log-normal, $N \leq 25$	Residential: 0-1.0, uniform
NO_2	0.25-0.90, uniform		Non-residential: 0.5, constant

4.3 Indoor sources

Although the indoor sources have been reported by many studies, the data should be illustrated attentively since a wide range of values have been observed (Emmerich and Persily 1996; Mohammed et al. 2015). Except for the infiltration from ambient environment, cooking, cigarettes, wood burning, candles/incense and other activities are identified as the most potent indoor sources in this thesis as stated in Section 3.3.3. The detailed information is explained in the following sections.

4.3.1 Cigarettes

Environmental tobacco smoke (ETS) is one of the most argued indoor air quality issues because of its threat to human health (Bolte et al. 2008; Piccardo et al. 2010). Due to limitation of data, only the exposures to ETS at home and workplace indoors are involved in this thesis. As expressed by Equation 3-7, the emission rate for cigarettes is calculated based on information of the number of cigarettes smoked, source strength of the cigarette, as well as the fraction of people exposed to ETS.

4.3.1.1 Number of cigarettes per day

The daily average cigarette consumption after 2005 for both males and females in each country is generated by the author using the data from the European Health Interview Survey (EHIS) (EUROSTAT 2010). Information about the daily cigarette consumption before the 2000s was quite rare. Hill and Laplanche (2005) and Ritchie and Roser (2018) have reported a trend of growth from the 1930s to 1985 and a decrease afterwards for European countries including Switzerland, Spain, Germany and France. This tendency is adopted in this thesis and the average number of cigarettes smoked per day is assumed to be changed gradually by 0.2 per year based on the data from Hill and Laplanche (2005) and Eurobarometer (2006).

The daily cigarette consumption is distributed proportionally based on the total time spent at home or workplace since only the ETS exposure in these two micro-environments are taken into account. It should be pointed out that people are not able to smoke apparently during sleep. Thus, when scaling the cigarette consumption, the time spent on sleeping should be neglected.

4.3.1.2 Source strength of cigarette

Many studies have measured the source strength of PM2.5 and NO_2 from cigarettes. According to previous studies, the cigarette particle emissions vary among different brands and types (Mueller et al. 2012; Gerber et al. 2015; Wasel et al. 2015; Kant et al. 2016). Additives, tar content and filter types also affect the PM emission from cigarettes (Rustemeier et al. 2002; Saha et al. 2007). The source strength of PM2.5 is assumed to follow a normal distribution of 10,950 (± 2,000) μg per cigarette as reported by Daisey et al. (1994) and Oezkaynak et al. (1996).

According to the National Research Council (1986) and WHO (1997), initially all the NO_x is emitted in the form of NO; once emitted, NO is gradually oxidised to NO_2. Similar to PM2.5, the emission of NO_2 from cigarettes is impacted by brands and types (Jenkins and Gill 1980). For NO_2 the source strength is assumed to follow a normal distribution of 1,930 (± 64.4) μg per cigarette based on monitoring data from Rickert et al. (1987) and Searl (2004).

4.3.1.3 Fraction of people exposed to ETS

Data for the fraction of people exposed to ETS is given by the Survey on Tobacco Analytical report (Eurobarometer 2009). The study investigated separately the proportion of smokers and non-smokers exposed to ETS at home indoors with the question "Do you or any other person living with you smoke inside your home every day or almost every day?". Again it should be noted that this portion is specific for the exposure inside the house. As displayed in Table 4-6, the smokers generally have a higher chance to be affected by ETS compared to the non-smokers.

Table 4-6: Fraction of people exposed to ETS at home by country and smoking habit (all respondents, non-smokers and smokers) (Eurobarometer 2009).

Country	All respondents	Non-smokers	Smokers
Austria	0.16	0.14	0.21
Belgium	0.18	0.18	0.18
Bulgaria	0.29	0.23	0.38
Cyprus	0.29	0.31	0.25
Czech Republic	0.23	0.16	0.35
Denmark	0.23	0.17	0.35
Estonia	0.17	0.16	0.18
Finland	0.02	0.02	0.03
France	0.11	0.09	0.15
Germany	0.15	0.13	0.19
Greece	0.29	0.28	0.30
Hungary	0.20	0.12	0.33
Ireland	0.17	0.14	0.23
Italy	0.14	0.11	0.22
Latvia	0.15	0.12	0.20
Lithuania	0.30	0.28	0.34
Luxembourg	0.10	0.08	0.17
Malta	0.12	0.10	0.17
Netherlands	0.16	0.15	0.19
Norway	0.09	0.06	0.19
Poland	0.24	0.21	0.31
Portugal	0.15	0.13	0.21
Romania	0.25	0.23	0.30
Slovakia	0.19	0.13	0.30
Slovenia	0.16	0.14	0.19
Spain	0.23	0.20	0.30
Sweden	0.05	0.03	0.11
Switzerland	0.17	0.14	0.24
United Kingdom	0.11	0.07	0.24

The distinction between the smokers and non-smokers leads to the necessity of determining the prevalence of smoking in each country. The proportion of smokers is summarised by the author based on the data from EHIS. Eurobarometer (2007; 2009; 2012) have revealed that the smoking habit is influenced by socio-economic variables. Thus, the information for smoking prevalence is stratified by country, gender, age group and employment status. It should be noted that, EHIS only contains data after the 2000s.

Information for the prevalence of smoking before 2000 is very limited. Sardu et al. (2006) revealed that the trend of smoking was differentiated by gender in Italy after the 1950s. For men, the percentage of regular smokers decreased since the 1950s. For women, the trend of growth lasted only until the 1980s. Hill and Laplanche (2005) announced a similar finding for men, nevertheless, the prevalence of female smokers in France continually increased from the 1950s to 2000s. However, the difference between the 1980s and 2000s was rather unremarkable (from 20% to 22%). Data from WHO (2015) showed that in European countries the portion of female smokers decreased after the 2000s.

Hence, the data for smoking prevalence in the 2000s are extrapolated to earlier years in this thesis based on the assumption that 1) the proportion of male smokers was reducing evenly since the 1950s and 2) the prevalence of female smokers was increasing steadily from the 1950s to a peak in 1985, declining afterwards. Due to the lack of knowledge, the portion of smokers was assumed to stay unchanged for the earlier time periods before the 1950s.

For offices less data are accessible. Eurobarometer (2009) investigated the amount of exposure to tobacco smoke at the workplace (see Table 4-7). For simplicity, people "Hardly ever" and "Never exposed" are classified as the group that is not influenced by ETS at the workplaces. These data are applied in this thesis regardless of smoking habits or socio-economic status considering the restriction of data availability.

Since the harm of second-hand smoke on human health has been confirmed, many European countries have implemented smoking bans with the aim to protect people from the effects of ETS (Origo and Lucifora 2010; Mons et al. 2013). According to expert judgement, it is assumed in this thesis that the fraction of people exposed to ETS at home and workplace after 2010 was only 60% and 10% of the corresponding values in 2009.

4.3.2 Cooking

4.3.2.1 Source strength

Cooking as one of the most vital indoor sources for PM2.5 has been studied worldwide (Baxter et al. 2007; Gerharz et al. 2009; Brown et al. 2012; Fabian et al. 2012). For particles, the emission rates of cooking are related to various factors including food type, oil type, cooking method, cooking temperature and stove type (Buonanno et al. 2009; Hu et al. 2012; Zhao and Zhao 2018). According to He et al. (2004) and Dacunto et al. (2013), the emission factors range from about 30 $\mu g\ min^{-1}$ for oven cooking to 15,200 $\mu g\ min^{-1}$ for frying chicken breast with olive oil.

Table 4-7: Fraction of people exposed to ETS at workplace by country (Eurobarometer 2009).

Country	Never exposed	Hardly ever	Exposed
Austria	0.68	0.11	0.21
Belgium	0.71	0.13	0.16
Bulgaria	0.40	0.21	0.39
Cyprus	0.37	0.18	0.45
Czech Republic	0.63	0.14	0.23
Denmark	0.72	0.10	0.18
Estonia	0.66	0.14	0.20
Finland	0.76	0.12	0.12
France	0.71	0.14	0.15
Germany	0.71	0.12	0.17
Greece	0.30	0.11	0.59
Hungary	0.52	0.24	0.24
Ireland	0.70	0.16	0.14
Italy	0.77	0.07	0.16
Latvia	0.51	0.23	0.26
Lithuania	0.51	0.18	0.31
Luxembourg	0.77	0.10	0.13
Malta	0.69	0.07	0.24
Netherlands	0.77	0.09	0.14
Norway	0.78	0.09	0.13
Poland	0.45	0.23	0.32
Portugal	0.81	0.06	0.13
Romania	0.44	0.20	0.36
Slovakia	0.53	0.19	0.28
Slovenia	0.77	0.08	0.15
Spain	0.75	0.06	0.19
Sweden	0.82	0.10	0.08
Switzerland	0.70	0.13	0.17
United Kingdom	0.77	0.12	0.11

For NO_2 exposure, the cooking activity, even with the electric stove, is an important factor (Fortmann et al. 2001; Fabian et al. 2012). Similar to PM2.5, considerable variance has been reported for NO_2 source strength (Moschandreas et al. 1987; Persily 1998). As pointed out by Dennekamp et al. (2001), NO_2 emitted from the gas stoves was generally higher than the electric ones. Melia et al. (1978) even demonstrated that "the average hourly concentration of NO_2 in gas kitchens was more than seven times greater than that in electric kitchens".

To make the question less complicated, the source strength of cooking for PM2.5 and NO_2 is assumed to be normally distributed and summarised based on the information from Burke et al. (2001), Fortmann et al. (2001), He et al. (2004) and Fabian et al. (2012) (see Table 4-8). For NO_2, distinction has been made between the electric and other stove types (primarily the gas stove). Data for the prevalence of stove type are derived from the TIMES PanEU model (Blesl et al. 2011) for each country in Europe (see Table A-5).

4.3.2.2 Capture efficiency of kitchen hood

Studies have demonstrated that the pollutants generated during the cooking process can be mitigated through the operation of a hood or an exhaust fan in the kitchen (Li et al. 1997; Chiang et al. 2000; Huang et al. 2004; Sjaastad and Svendsen 2010). The capture efficiency (CE), i.e. the ratio of between the amount of pollutants captured by the hood and the total amount of contaminants generated at the source, is applied to determine the performance of cooking hood (Li and Delsante 1996).

The effectiveness of a cooking hood varies with factors including hood type, exhaust airflow rate, hood design, hood position and whether front, back, or oven burners are used (Lunden et al. 2015). Singer et al. (2012) reported a potential CE of over 90% at a flow rate of 138 litres per second for the front burners, while around 15% when operated at minimum flow (Delp and Singer 2012). In some cases, range hoods do not exhaust pollutants to outdoors but simply recirculate the particles indoors, which can also result in an unsatisfactory performance (Rim et al. 2012).

To simplify the study, the CE is assumed to follow a uniform distribution and data employed in this thesis are summarised based on information from Delp and Singer (2012), Singer et al. (2012) and Lunden et al. (2015) (see Table 4-8).

In spite of the mitigation effect of hoods on cooking-generated emissions, people may not always turn them on due to the lack of knowledge or the concerns about noise (Parrott et al. 2003). Thus, it is assumed in this thesis that the actual portion of households using hoods during cooking increased gradually since 1937 (the year exhaust hoods were introduced) and reached the peak in the 2000s with a value of 85%. This value remained constant until 2015.

4.3.2.3 Cooking time

As interpreted by Equation 3-8, the time spent on cooking is another influencing factor. The data are recorded by activity "food preparation, cooking" in the MTUS data for each

Table 4-8: Values for cooking-related parameters.

Pollutant	Stove type	Source strength (µg min^{-1})	Capture efficiency
PM2.5	-	1125 (±280), normal	15%-90%, uniform
NO_2	Electric	270 (±75), normal	20%-95%, uniform
	Gas	1800 (±450), normal	

diary. Additionally, it is assumed that thirty percent of the total food preparing time is spent on actual cooking or frying.

4.3.3 Wood burning

Wood combustion is a source of enormous amount of particle and NO_2 emissions for the indoor environment (Emmerich and Persily 1996; McDonald et al. 2000; Guo et al. 2008b; McNamara et al. 2013; Salthammer et al. 2014; Semmens et al. 2015). Due to the data limitation, only the wood burning at home is taken into account in this thesis. As interpreted by Equation 3-9, the emission rates of wood burning are determined by several pieces of information: source strength of wood, time of burning wood, heat demand, room volume, removal ratio and last but not least, the fraction of people exposed to biomass at the national level.

The source strength of wood for PM2.5 and NO_2, as well as the average burning time per day, is the weighted average value of different stove types based on the data from Struschka et al. (2008). The values are assumed to follow the uniform distribution. Noonan et al. (2012) and Hartinger et al. (2013) have revealed the impressive mitigation effect of chimneys since the dominant part of pollutants from fireplaces or wood stoves is led to the outside environment through the chimneys directly. Thus, a removal ratio (the portion of the emission emitted to outdoors) ranging between 0.950 and 0.995 is adopted. Data for the fraction of people influenced by stoves and fireplaces in each country are also taken from Torfs et al. (2007).

With regard to the heating demand, the influence of insulation should be additionally considered. The main objective of insulation is to improve the energy performance of buildings and reduce the emission of CO_2 and other air pollutants. However, to what extend the heating demand can be decreased is depending on various factors, e.g. insulation techniques, materials used and climate zones (Fadzil et al. 2017; Simona et al. 2017; Yang and Tang 2017).

To simplify the study, an average heating demand of 85 kJ m^{-3} h^{-1} is applied for “natural” buildings according to Torfs et al. (2007). For “retrofitted”, “mechanical” and “AHU” buildings, this value is set to be 20% less in terms of the data from Zomorodian and Nasrollahi (2013), Ibrahim et al. (2016) and Eleftheriadis and Hamdy (2018).

Detailed information for the biomass parameters, as well as the fraction of people exposed to biomass for each country, can be found in Table 4-9 and Table A-6.

Table 4-9: Values for parameters of wood burning.

Pollutant	S_{wood} (μg kJ^{-1})	t_{wood} (h)	$R_{removal}$
PM2.5	13-146, uniform	0.3-2.8, uniform	0.950-0.995, uniform
NO_2	58-185, uniform		

4.3.4 Candles/incense

People are also exposed to other combustion sources, e.g. candles and incense, which are commonly utilised for aesthetic or religious purposes (Sørensen et al. 2005; Stabile et al. 2012). A wide range of the emission strength has been found by the literatures (Lee and Wang 2004; Afshari et al. 2005; Pagels et al. 2009). The variability of the results is probably owing to the diverse sorts of substances added in the candles or incense. Also, the incense burning is generally shown to have higher PM2.5 emissions compared to the candles (Hu et al. 2012).

In this thesis the source strength of candles/incense is summarised based on the data from He et al. (2004), Sørensen et al. (2005), Hu et al. (2012) and Stabile et al. (2012), and assumed to follow a uniform distribution (5.5 to 910 μg min^{-1}). Due to the lack of information, three assumptions are made: 1) 10% of the population are exposed to PM2.5 emissions from candles or incense burning, 2) only the exposure to candles/incense at home is considered and 3) the average burning time ranges from 5 to 120 minutes per day.

4.3.5 Other activities

It is found by the PTEAM study that the unidentified indoor sources accounted for around 14% of the indoor PM2.5 concentration (Oezkaynak et al. 1996), which were hypothesised by Wallace (2000) as the resuspension of dust from human activities. According to Thatcher and Layton (1995), even normal activity, like walking or sitting, dramatically increased the indoor concentration of particles with an aerodynamic diameter greater than 1 μm. Several studies have analysed the influence of housework activities (e.g. vacuuming, dusting, sweeping) (Long et al. 2000; Abt et al. 2000), however, the source strengths have not been established. Data for PM2.5 resuspension are summarised based on the information from He et al. (2004) and Ferro et al. (2004). The source strength of resuspension activity is assumed to follow the uniform distribution and data are presented in Table 4-10. For other activities that involve movement and multiple people, e.g. party, games and other indoor leisure, a range between 60 and 300 μg min^{-1} has been assigned. Data for the time spent on these activities are recorded in the MTUS data for each diary.

Table 4-10: Values for source strength of other activities (μg min^{-1}).

Activity	Min	Max
Set table, wash/put away dishes	20	180
Cleaning/other domestic work	90	440
Laundry, ironing, clothing repair	20	180
Imputed personal or household care	20	80
Wash, dress, care for self	20	80

4.4 Room volume

4.4.1 Residential buildings

Room volume is a necessary parameter to assess the pollutant concentration generated by the indoor sources. The information for residential size is derived from the EU-SILC data (EUROSTAT 2013). The data are summarised and stratified by country and other SES variables. It can be seen from the data that the SES variables are important driving factors of the room size. Figure 4-6 shows the histogram of the residential size in Germany by income level as an example. It is not surprising that the room size is positively correlated with the household income. It can also be concluded from the figure that the density histogram is similar to a log-normal distribution.

The height of residential rooms should guarantee the minimum value regulated in the Manual of Standard Building Specifications (European Commission 2011). According to the manual and the estimation of Hänninen et al. (2013), the dwelling height is assumed to accord with the uniform distribution with a range from 2.4 to 3.5 m.

It must be noted that, most of the existing models simulated the cooking-generated pollutant concentration based on the assumption that the emission diffused evenly in each residential area (e.g. SHEDS-PM). However, researchers reported higher concentration in the kitchen compared to other spaces of residence (Huboyo et al. 2011; Poon et al. 2016). This is due to the fact that the kitchen is “detached” from other rooms in the living space when people keep the door closed during the cooking process. On the other hand, Lai and Chen (2007b), Lai and Ho (2008) and Kim et al. (2018) argued that the emission diffused not only in the cooking area, but also rapidly to the adjacent living room, even other spaces of the residence, especially for the open plan kitchen.

Based on the information stated above, the room volume for calculating the emission from cooking is considered additionally in this thesis. To simplify the study, the affected area is estimated by multiplying the total residence area with a random factor from 0.2 to 0.9.

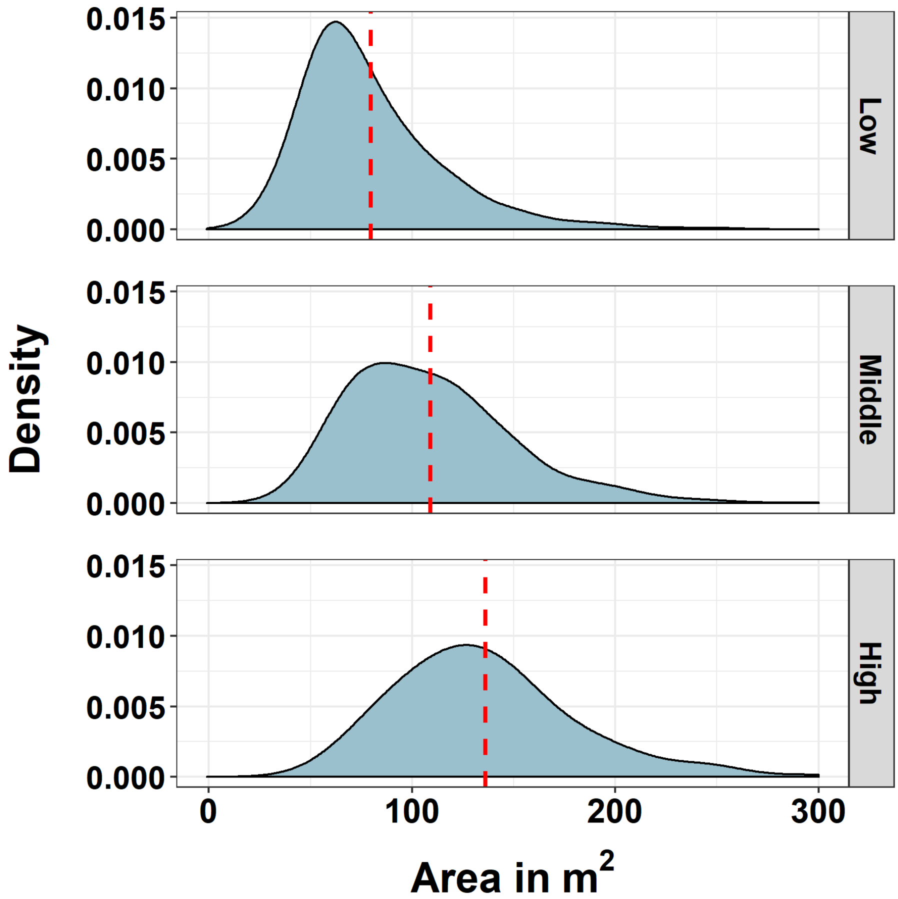

Figure 4-6: Density histogram of the residential size in Germany by income level. The red dash lines represent the average value by category respectively (80 m^2 for "Low", 109 m^2 for "Middle" and 136 m^2 for "High").

4.4.2 Offices

The data of office volume are derived from the questionnaire survey from EU project OFFICAIR (Bartzis et al. 2013), which investigated 160 buildings and offices across Europe. The offices were classified as either cellular or landscape and the volume of both categories were assumed to accord with the triangular distribution. The summaries of office volumes are shown in Table 4-11. Since very limited data can be found for the prevalence of two office types, the distributions are assumed to be identical as the survey samples. These values are applied to each country since offices are the most standardised buildings as described in Section 4.2.2.2.

4.4.3 Schools

The data for areas and heights of classrooms are gathered from database provided by the EU project SINPHONIE (Csobod et al. 2014). Both area and height are assumed to accord with the triangular distribution. For countries where data are missing, the same method of gap filling described in Section 4.2.2.3 is applied. Detailed information for each

Table 4-11: Parameters for the probabilistic modelling of office volume (m^3) (University of Western Macedonia 2013).

Type	Sample size	Min	Max	Mode
Landscape	95	44.4	1,680.0	166.8
Cellular	45	23.4	176.0	77.3

country is presented in Table A-7.

4.5 Traffic ME factor

The traffic ME factor defines the ratio between the concentration in traffic to the background concentration. According to measurement results of Adams et al. (2001), Rank et al. (2001), Chan and Chung (2003), Gómez-Perales et al. (2004) and Briggs et al. (2008), the concentration in traffic is influenced by many factors including traffic modes, type of vehicles, type of roads, ventilation in vehicles and traffic load.

Even though a wide range has been reported, however, the contribution of transportation to the total exposure can be relatively small. According to the MTUS data, the median value of the total time people spent on travel each day in 2010 was just 30 min. The result of Gens (2012) showed that only 3% of the PM2.5 exposure was originated from "travel".

Thus, no detailed analysis is made in the frame of the thesis with the aim to simplify the study. Instead, an average ME factor is used for all transport activities despite the large variance between different transport modes. The author summarised the data from Zagury et al. (2000), Riediker et al. (2003), van Roosbroeck et al. (2006) and Zuurbier et al. (2010). It is assumed that for PM2.5 and NO_2 the ME factors are both normally distributed with 2 (±1.7) and 2.5 (±2.1) respectively.

4.6 Data for health impact assessment, aggregation and valuation

As defined by Equation 3-12, Equation 3-13 and Equation 3-14, the assessment for health impacts, DALYs and damage costs needs the following input data: CRFs (impact functions), DALY severity weight, duration and monetary values. The data are driven from the projects INTARESE, HEIMTSA and HRAPIE (Friedrich et al. 2011; Gens 2012; WHO 2013a; Huang et al. 2016, see Table A-4, Table A-8 and Table A-9). Both the CRFs and monetary values are given in the form of normal distribution.

For the formulation of CRFs or ERFs, many studies assumed a threshold, or a counterfactual value. Below this threshold, it is presumed that there is no health impact brought by the air pollutant. For PM2.5, the author follows the method of Burnett et al. (2018)

and a counter-factual value of 2.4 μg m^{-3} is applied to all the health endpoints. For NO_2, a threshold of 20 μg m^{-3} for endpoint YOLL is taken from WHO (2013a).

5 Results and uncertainties

5.1 The overall temporal development

5.1.1 PM2.5

Figure 5-1 displays the population-weighted arithmetic mean of the PM2.5 exposure in Europe stratified by source, including infiltration from outdoors, cooking, wood burning, smoking, candle/incense burning and other sources. The black line in the figure represents the population-weighted average PM2.5 ambient concentration in Europe. It should be noted that although the concentration fields from the 1930s to 1950s have been simulated by the author, the results are not presented in the figure due to the fact that 1) large uncertainty exists in the input emission data (see Section 3.2.4) and 2) no population data before the 1950s is available to weight the background concentration and overall exposure. As demonstrated by the figure, the overall exposure in Europe increased continuously from 1950 at 19.0 (95% CI: 3.3-55.7) µg m^{-3} to a peak in 1980 at 37.2 (95% CI: 9.2-113.8) µg m^{-3} and declined after that considerably to 2015 at 20.1 (95% CI: 5.8-51.2) µg m^{-3}.

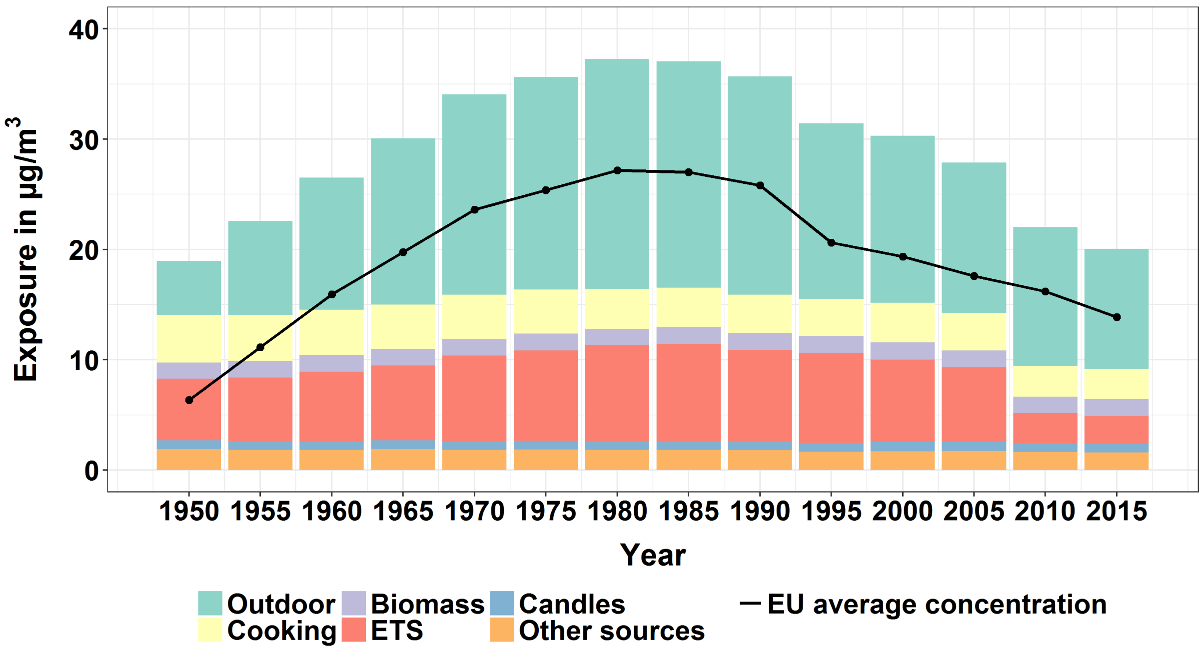

Figure 5-1: Population-weighted arithmetic mean PM2.5 exposure by source (infiltration from outdoors, cooking, wood burning, smoking, candle/incense burning and other sources) for European countries from 1950 to 2015. The black line indicates the average background PM2.5 concentration in Europe.

What is striking in Figure 5-1 is that the trend of the total exposure matches well with the temporal course of the ambient concentration, which indicates that the outdoor concentration is strongly correlated with the overall exposure (r=0.78, p=0). The population-weighted background concentration was only 6.4 μg m^{-3} in 1950 and rose to a high point of 27.2 μg m^{-3} in the 1980s. The concentration fell down after the 1980s notably to 2015 at 13.9 μg m^{-3}. This sharp decrease is mainly due to the emission reduction policies for PM2.5 and its precursors (NO_x, NH_3 and SO_2), e.g. the Air Quality Directives and the EURO Standards for Vehicles (Granier et al. 2011; Crippa et al. 2016).

Accordingly, the exposure due to outdoor air was only 4.9 μg m^{-3} and accounted for 26% of the total exposure in 1950. However, the total exposure boosted to a peak at 37.2 μg m^{-3} in 1980, within which over half (55%) was contributed by the ambient concentration (20.6 μg m^{-3}). The exposure owing to infiltration declined to only 10.9 μg m^{-3} in 2015, however, the role played by the outdoor air was still significant (54%).

Another dominant source for PM2.5 exposure is the ETS indoors. The temporal development of the ETS exposure was similar to the trend of the outdoor concentration. The maximum value was in 1985 at 8.8 μg m^{-3} as a result of the peak cigarette consumption in the 1980s. The exposure from ETS kept reducing afterwards and dramatically to only 2.5 μg m^{-3} in 2015. The reasons behind are the measures taken widely in European countries for tobacco control, including the smoking bans in public places and at workplaces, the health-related warnings on cigarette packages and the high taxation of tobacco products (Spinney 2007; Hammond 2011; Gorini et al. 2015; Bertollini et al. 2016).

From the plot it can be also noticed that there is a trend of slight reduction for exposure stemming from cooking. The MTUS data reveals that people spend less time nowadays on cooking than before. Figure 5-2 compares the density histogram of the daily cooking time of Dutch women in different time periods. The average time that Dutch women spent on cooking in 1975 (95 min) was 58% more than in 2005 (60 min). This outcome has been observed in other European countries as well. The other reason for the decline of exposure due to cooking is the increasing prevalence of kitchen hoods.

For the exposures from biomass, candles/incense and other sources no evidential variances can be observed among different time periods. This is unfortunately due to the limitation of data availability, especially for earlier years. It would be helpful if new data can be collected in the future.

The annual total exposure of PM2.5 and its source distribution for 1950, 1980 and 2015 are listed in Table 5-1. Within these three years, the greatest contribution resulting from the indoor sources was in 1950 (74%), within which almost one third stemming from the indoor smoking. Although the absolute exposure due to the indoor sources increased in 1980, the proportion they occupied dropped to 45% as a result of the extremely high outdoor concentration. Because of the decreasing exposure from ETS and cooking, the indoor sources made up to less than half of the total exposure in 2015 as well.

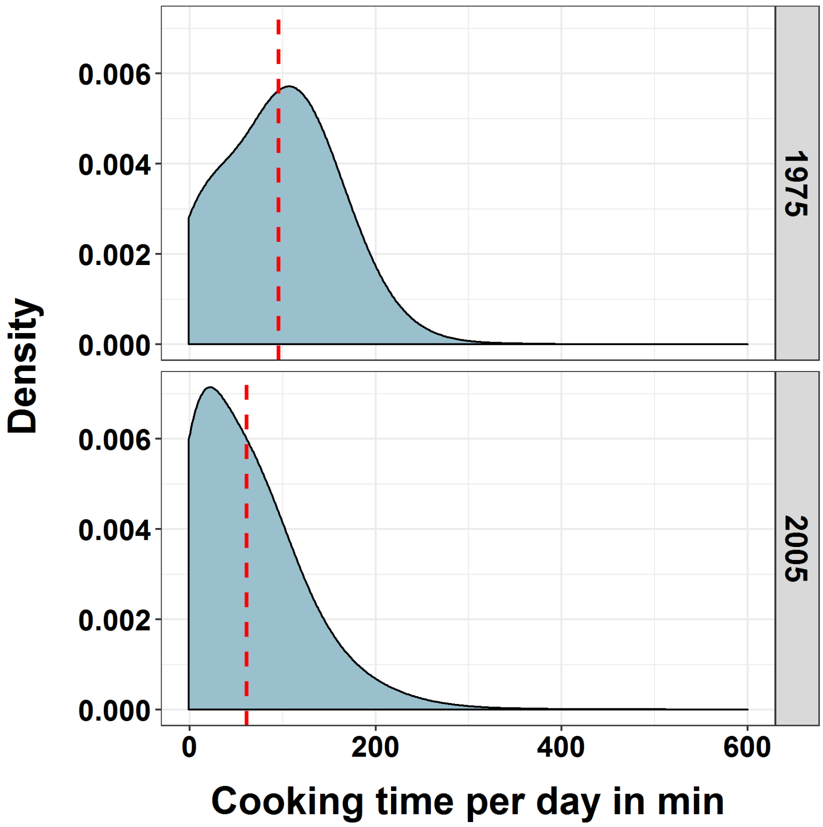

Figure 5-2: Density histogram of the daily total cooking time for the Dutch women in 1975 and 2005. The red dash line represents the arithmetic mean value (95 min for 1975 and 60 min for 2005).

Table 5-1: The source distribution of the population-weighted mean exposure to PM2.5 for the EU27+2 countries in 1950, 1980 and 2015.

Year		Outdoor	Cooking	Wood burning	Smoking	Candles	Other sources	Total
1950	Exposure (µg m^{-3})	4.9	4.1	1.6	5.6	0.9	1.9	19.0
	Percentage (%)	26	22	8	29	5	10	100
1980	Exposure (µg m^{-3})	20.6	3.6	1.6	8.7	0.9	1.8	37.2
	Percentage (%)	55	10	4	23	3	5	100
2015	Exposure (µg m^{-3})	10.9	2.8	1.5	2.5	0.8	1.6	20.1
	Percentage (%)	54	14	8	12	4	8	100

5.1.2 NO_2

Figure 5-3 presents the population-weighted arithmetic mean exposure for NO_2 by source, including infiltration from outdoors, cooking, biomass and ETS, from 1950 to 2015. The black line in the plot shows the average outdoor NO_2 concentration in Europe. Similar to PM2.5, the exposure to NO_2, as well as the background concentration, kept on increasing from 1950 at 10.4 (95% CI: 0.9-36.8) µg m^{-3} to a peak at 21.4 (95% CI: 6.3-51.8) µg m^{-3} and then began to decline steadily until 2015 at 15.5 (95% CI: 4.8-36.8) µg m^{-3}.

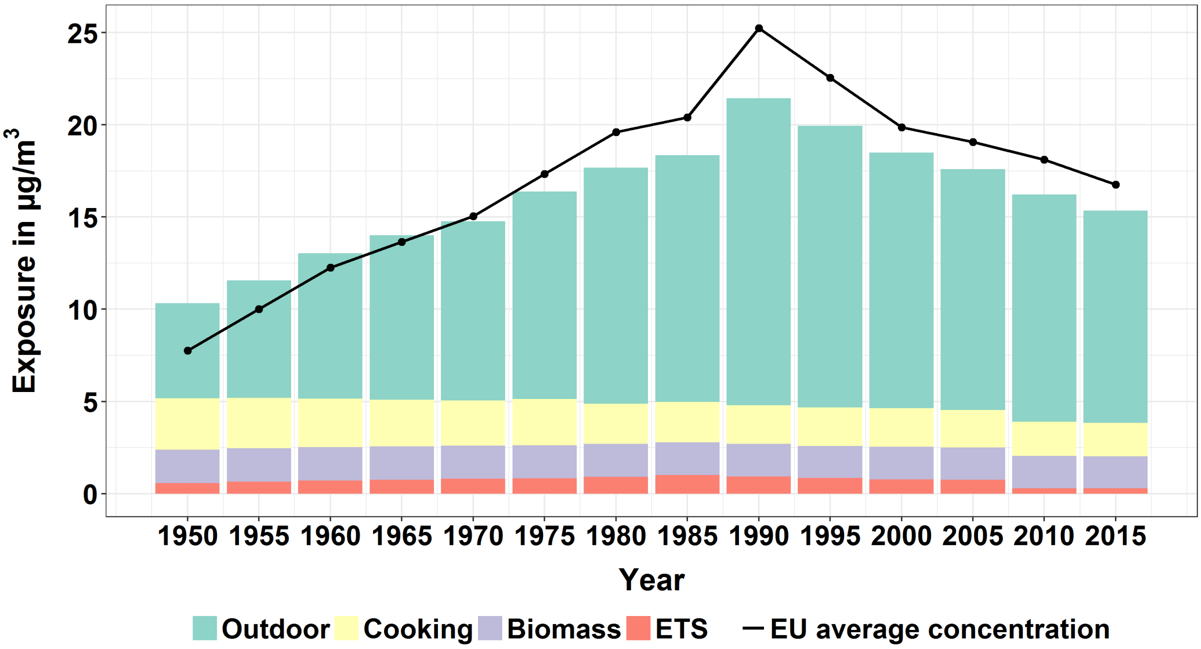

Figure 5-3: Population-weighted arithmetic mean NO_2 exposure by source (infiltration from outdoors, cooking, wood burning and smoking) for European countries from 1950 to 2015. The black line indicates the average background NO_2 concentration in Europe.

Compared to PM2.5, the correlation between ambient concentration and the overall exposure is more substantial (r=0.81, p=0). The dominant contribution of exposure originated from the infiltration. Even in 1950 when the exposure from infiltration accounted for only 5.2 µg m^{-3}, the proportion it took was approximately half. As for 1990, this ratio even increased to 78%.

It can be observed from Figure 5-1 that PM2.5 exposures were higher than the outdoor concentrations for all the time periods. Conversely, the ambient concentration of NO_2 exceeded the total exposure for the corresponding years from 1970 to 2015 (see Figure 5-3). It can be concluded that, the Indoor/Outdoor (I/O) ratios for NO_2 stayed less than one since the 1970s. This result is mainly due to two reasons: the relatively high decay rate (see Table 4-4) and the less important roles played by the indoor sources.

In comparison with PM2.5, the biggest difference for the indoor sources is demonstrated by the contribution of ETS. For NO_2 the ETS made up to just 6% of the overall exposure in 1950 and even dropped to only 2% after the 2010s. The relatively small amount of NO_2

added by the ETS has been confirmed by the studies from Koo et al. (1990) and Adgate et al. (1992) as well.

For NO_2 cooking and biomass were the most influential sources. In 1950 the contribution they made was almost half (44%). Similar to PM2.5, the exposure from cooking kept on decreasing since the 1950s.

The source distribution of population-weighted NO_2 exposures in 1950, 1990 and 2015 is shown in Table 5-2. More discussion in detail is given in Section 5.2.2.

Table 5-2: The source distribution of the population-weighted mean exposure to NO_2 for the EU27+2 countries in 1950, 1990 and 2015.

Year		Outdoor	Cooking	Wood burning	Smoking	Total
1950	Exposure (μg m^{-3})	5.2	2.8	1.8	0.6	10.4
	Percentage (%)	50	27	17	6	100
1990	Exposure (μg m^{-3})	16.6	2.1	1.8	0.9	21.4
	Percentage (%)	78	10	8	4	100
2015	Exposure (μg m^{-3})	11.7	1.7	1.8	0.3	15.5
	Percentage (%)	75	11	12	2	100

5.2 Annual average exposure at country level

5.2.1 PM2.5

In the following sections, the annual exposure at country level for 2015, 1980 and 1950 are selected and discussed in detail.

5.2.1.1 PM2.5 exposure in 2015

Figure 5-4 shows the density histogram for the exposure to PM2.5 of the German population in 2015. As displayed by the figure, the results of the Monte Carlo analysis for PM2.5 seem to be log-normally distributed. The similar results have been demonstrated by Moschandreas and Saksena (2002), Burke (2005) and Gens (2012). Since the distribution of the exposure has a long "tail" to the right, the geometric mean (blue line, 14.2 μg m^{-3}) lies on the left side of the arithmetic mean (red line, 16.6 μg m^{-3}).

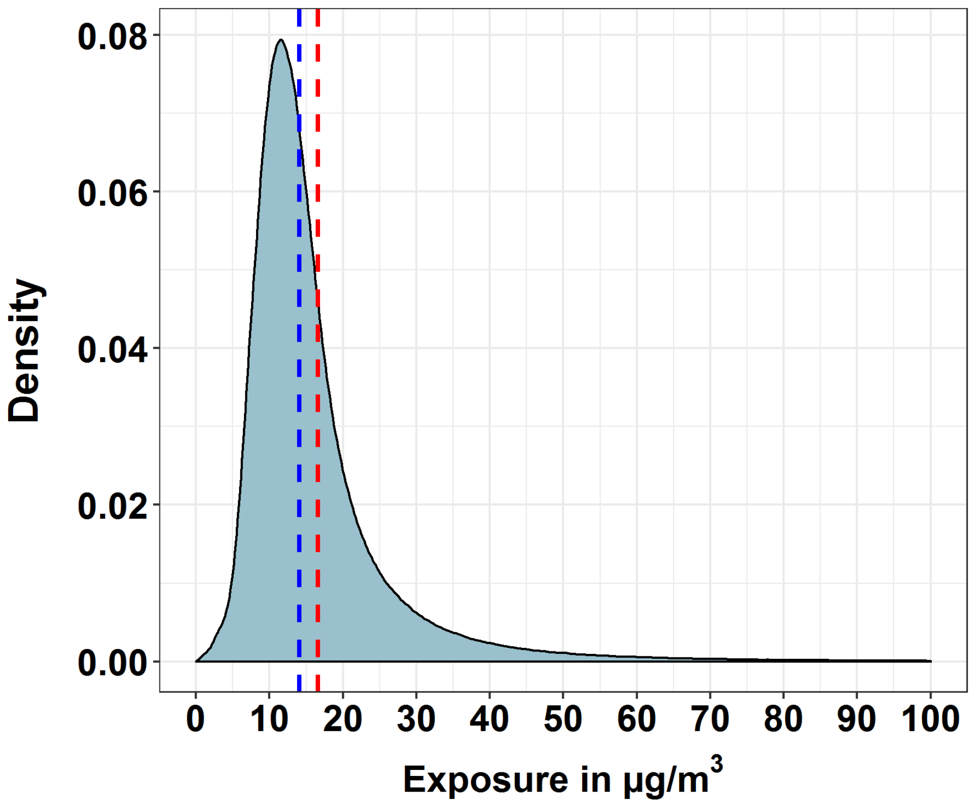

Figure 5-4: Density histogram of the PM2.5 exposure for Germans in 2015. The red dash line represents the arithmetic mean value (16.6 µg m^{-3}), while the blue line indicates the geometric mean (14.2 µg m^{-3}) of the Monte Carlo realisations.

Figure 5-5 displays the geometric mean of PM2.5 exposure in 2015 for European countries. For simplicity, the names of the countries are given in the form of the EUROSTAT country code (see Table A-3). The blue bars display the geometric mean of the exposure, while the yellow error bars show the range of the "geometric mean multiplies the geometric standard deviation squared (GSD^2)" and "geometric mean divided by GSD^2". Since the results are deemed as log-normally distributed, this range indicates the 95% of confidence interval (95%CI) (Fantke et al. 2012; Gens 2012). For PM2.5 the geometric mean and GSD of exposure for the EU27+2 countries were 17.0 µg m^{-3} and 1.7 in 2015.

Figure 5-6 shows the arithmetic mean of the PM2.5 exposure by country and source in 2015. As displayed in Table 5-1, the majority of the PM2.5 exposure originated from the outdoor concentration (54%). Within the indoor sources, smoking (12%) and cooking (14%) played the most important roles.

As presented by Figure 5-5 and Figure 5-6, the total PM2.5 exposure, as well as its source distribution, varied considerably among different countries. The highest exposures were found in Eastern European (EE) countries due to the following reasons:

- EE countries had very high background concentrations, e.g. Bulgaria was suffered from the highest average exposure from the outdoor air (18.6 µg m^{-3}), which was almost double of the European average value.

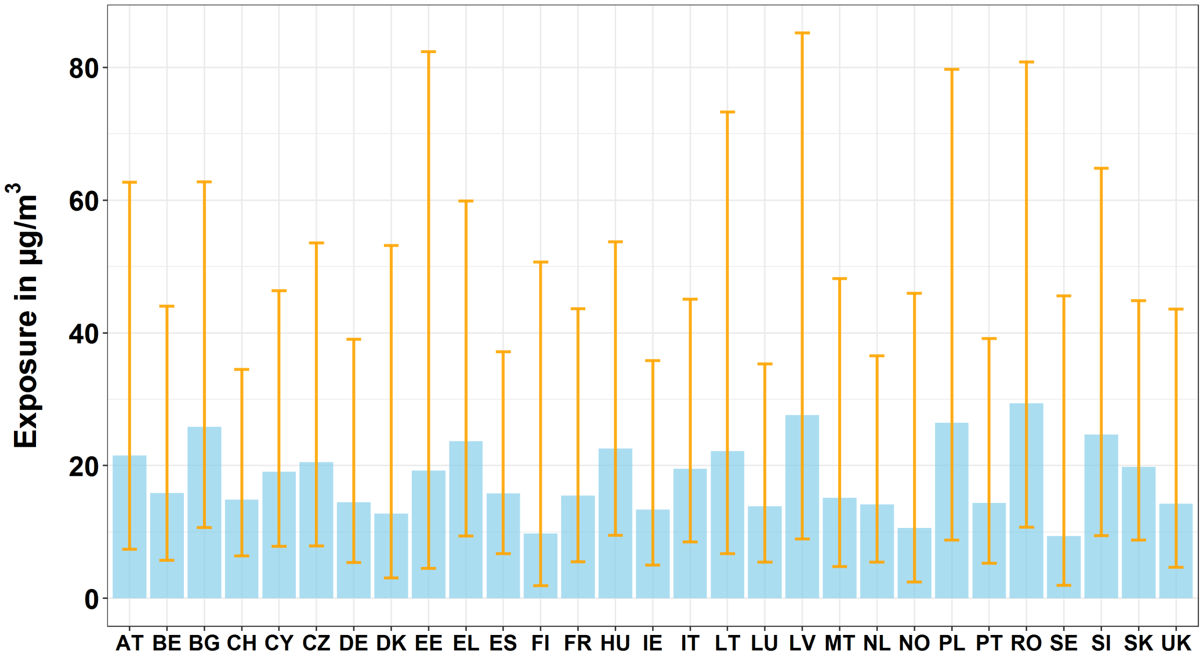

Figure 5-5: Population-weighted geometric mean PM2.5 exposure by country in 2015. The blue bars represent the geometric mean of the Monte Carlo realisations for each country and the yellow error bars depict the range of "geometric mean multiplies GSD^2" and "geometric mean divided by the GSD^2". The population-weighted geometric mean and GSD of exposures for the EU27+2 countries were 17.0 µg m^{-3} and 1.7.

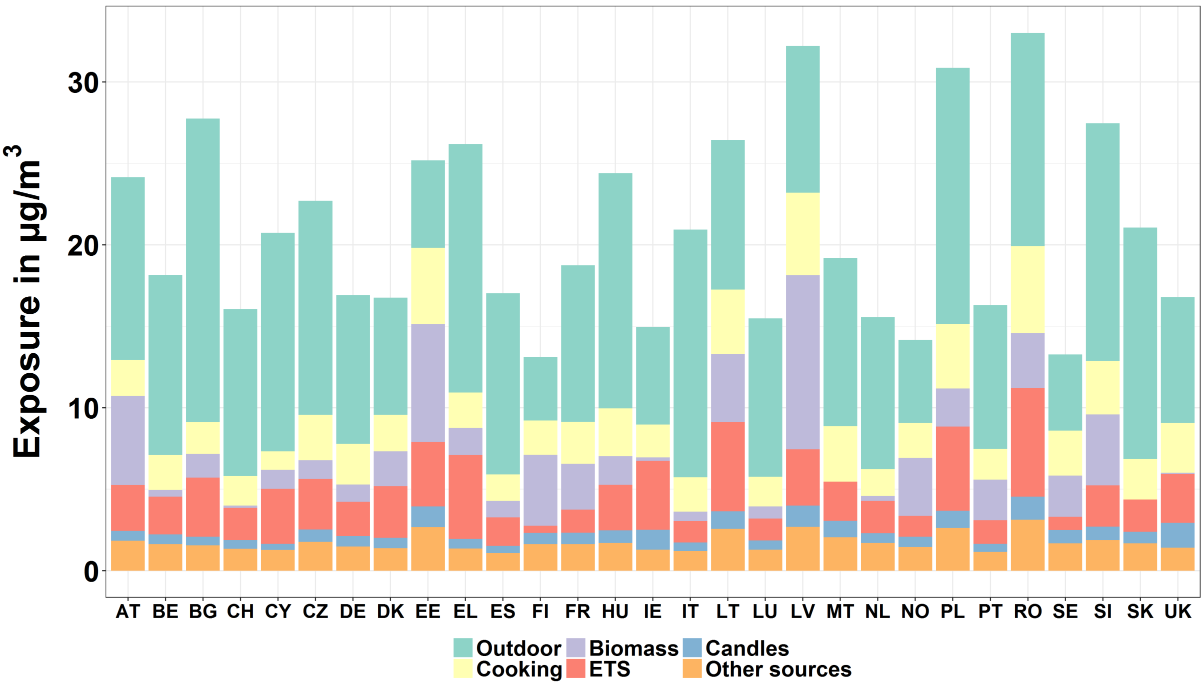

Figure 5-6: Population-weighted arithmetic mean PM2.5 exposure by country and source in 2015. For Europeans the mean exposure was 20.1 µg m^{-3}.

- A lot of people in EE countries were exposed to ETS indoors (e.g. Lithuania, Poland), or wood burning at home (Latvia, Estonia) (see Table 4-6, Table 4-7 and Table A-6).

- The population in EE countries tended to spend more time on cooking, which led to higher level of exposure. Figure 5-7 shows the density histogram of the daily total cooking time for women from EE and NE (Northern European) countries in 2000, with the red lash lines representing the arithmetic mean values. As shown in the figure, the women in EE spent on average 67% more time for cooking (105 min for EE and 63 min for NE).

- The dwelling rooms in EE countries were relatively tiny compared to other regions. Figure 5-8 compares the dwelling area in EE and NE countries as an example. Even though the small room area is not relevant to the exposure from infiltration, it hinders the dilution of the indoor pollutant sources (see Equation 3-4).

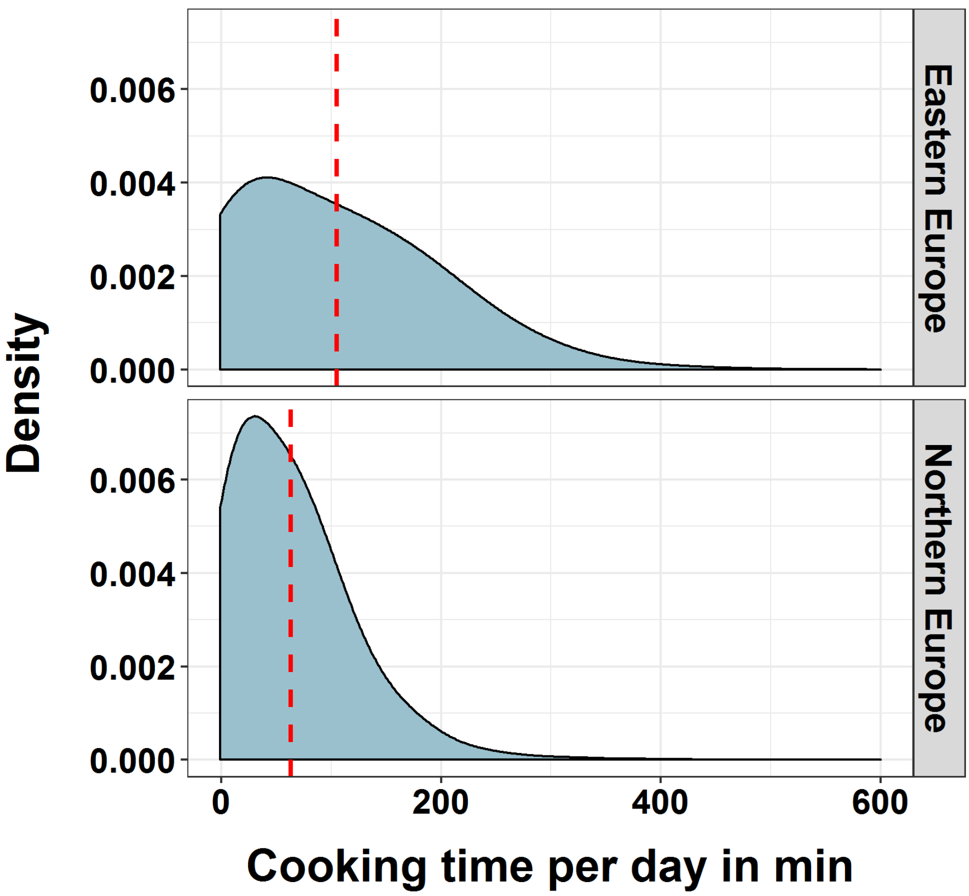

Figure 5-7: Density histogram of the daily total cooking time for women from Eastern and Northern European countries in 2000. The red dash line represents the arithmetic mean value (105 min for EE and 63 min for NE).

In contrast, NE countries were shown as least affected by the fine particles even though the biomass exposures were relatively high (e.g. Finland, Sweden, Norway). This was

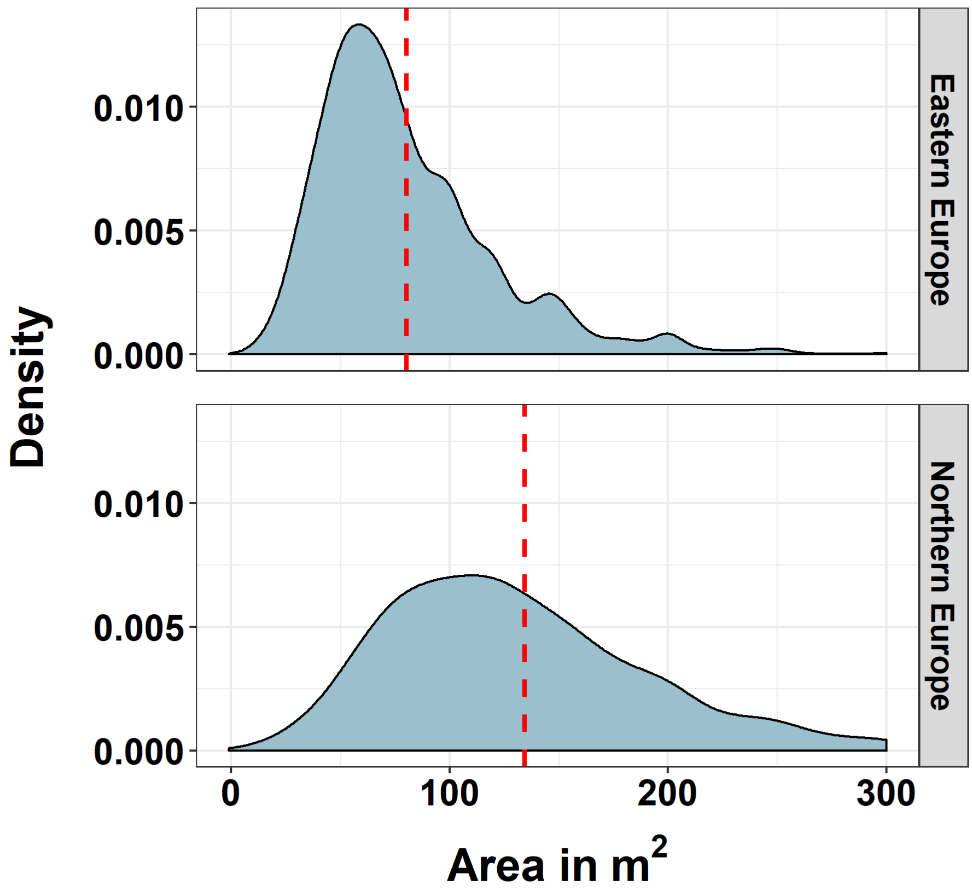

Figure 5-8: Density histogram of the dwelling size for Eastern and Northern European countries. The red dash line represents the arithmetic mean value (80 m^2 for EE and 134 m^2 for NE).

mainly due to the low background concentration, high prevalence of HVAC systems, less cooking time, large dwelling area and low proportion of population exposed to ETS. The similar result has also been revealed by some Northwestern European (NWE) countries including Luxembourg, Netherlands and Switzerland.

5.2.1.2 PM2.5 exposure in 1980

Likewise, the Monte Carlo realisations for 1980 are also similar to log-normal distribution with a geometric mean of 31.2 µg m^{-3} and GSD of 2.0. Figure 5-9 represents the source distribution of PM2.5 exposure in each country. For European countries the population-weighted arithmetic mean exposure was 37.2 µg m^{-3}.

Again the highest level of exposures were found in EE countries like Poland, Romania, Hungary and Latvia. Noticeably in Poland the value even reached 66.2 µg m^{-3}. The exposures to PM2.5 in NWE countries, e.g. Austria and Belgium, were relatively high as well except for Ireland. In contrast, Southern European (SE) and NE countries including Portugal, Norway and Sweden were the least affected.

The predominant role played by the ambient concentration applied to almost all coun-

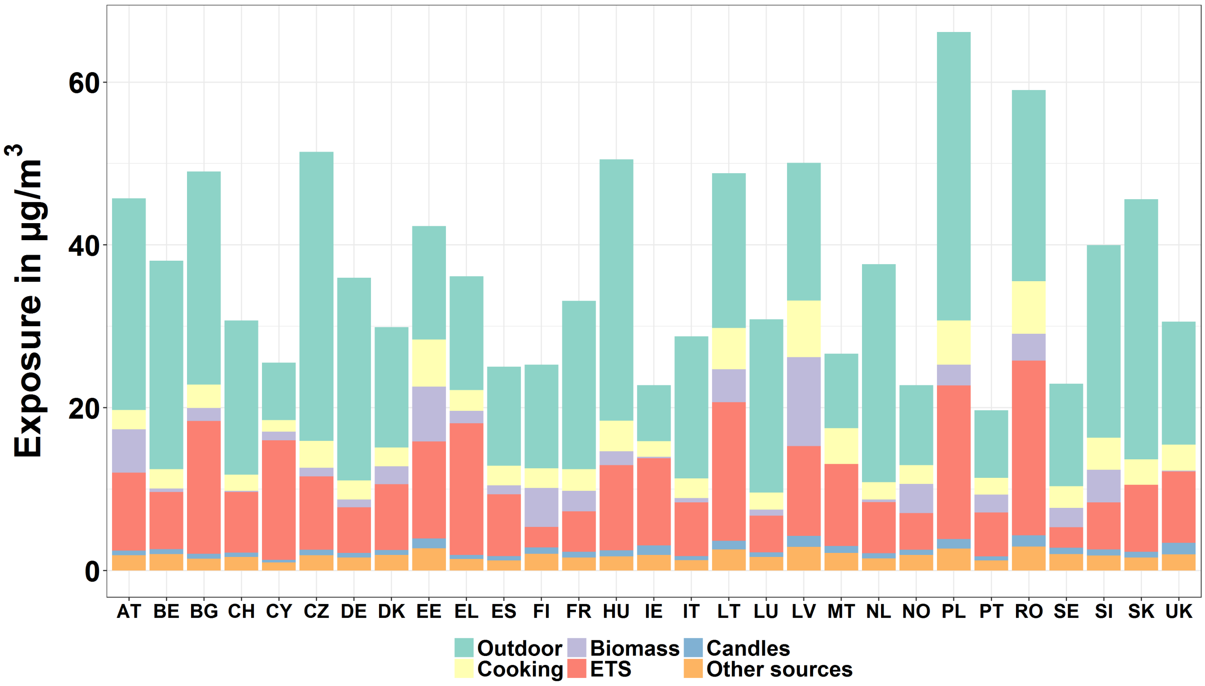

Figure 5-9: Population-weighted arithmetic mean PM2.5 exposure by country and source in 1980. For Europeans the mean exposure was 37.2 µg m^{-3}.

tries except some SE, NWE and NE countries including Ireland, Cyprus, Portugal, Malta, Norway, Spain, Sweden and Finland. The greatest contribution of outdoor air was mainly observed by EE countries, e.g. Czech Republic (35.5 µg m^{-3}), Poland (35.4 µg m^{-3}), Hungary (32.1 µg m^{-3}) and Slovakia (32.0 µg m^{-3}).

Another critical contributor for PM2.5 in 1980 was the ETS. The influences in most of the EE countries were extremely significant. The contribution of ETS to the total exposure in countries like Greece, Cyprus and Ireland, also ranked top on the list. Especially in Cyprus, 58% of the total exposure was stemming from the indoor smoking. Compared to these countries, the impacts of the ETS in NE countries (Sweden, Finland and Norway) were rather limited. However, the absolute values of 1980 were still around three times higher than those of 2015.

Although the contributions were less remarkable, the exposures from cooking, wood burning, candles and other sources still took altogether 22% of the overall exposure. In EE countries, especially in Latvia, PM2.5 exposures originated from these sources amounted up to an outstanding part of 44%.

5.2.1.3 PM2.5 exposure in 1950

In 1950 the overall geometric mean exposure for the EU27+2 countries was 13.8 µg m^{-3} with a GSD of 2.1. Figure 5-10 represents the source distribution of PM2.5 exposure in each country. The exposures in 1950 were less influenced by the outdoor sources even compared to 2015. Quite the opposite, the contribution from the indoor sources was substantial. This made EE countries again suffer from the highest exposure level.

Romania ranked top with 33.0 μg m^{-3}, within which 43% was owing to the indoor smoking.

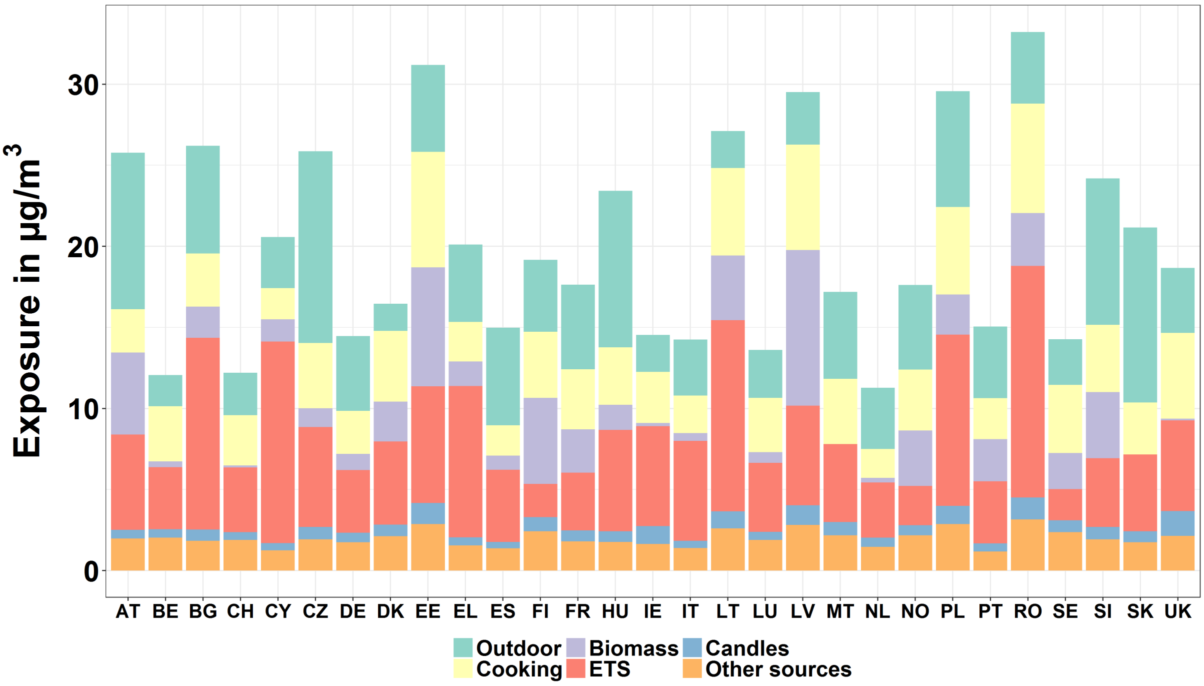

Figure 5-10: Population-weighted arithmetic mean PM2.5 exposure by country and source in 1950. For Europeans the mean exposure was 19.0 μg m^{-3}.

Austria was also outstanding among the list as a result of rather high exposure due to background concentration (9.7 μg m^{-3}). Additionally, over half (52%) of the population in Austria were exposed to wood burning indoors.

5.2.2 NO_2

Similarly, the analysis for NO_2 exposure is conducted for 2015, 1990 and 1950 separately in the following sections.

5.2.2.1 NO_2 exposure in 2015

Figure 5-11 presents the density histogram for the exposure to NO_2 of Germans in 2015. Similar to PM2.5, the results of Monte Carlo simulation also seem to be log-normally distributed. For Europeans, the geometric mean and GSD of exposures were 14.0 μg m^{-3} and 1.6. Figure 5-12 displays the arithmetic mean of the population-weighted exposure and its source distribution, including the infiltration from outdoors, cooking, wood burning and ETS for each country. For the EU27+2 countries, the average exposure was 15.5 μg m^{-3}.

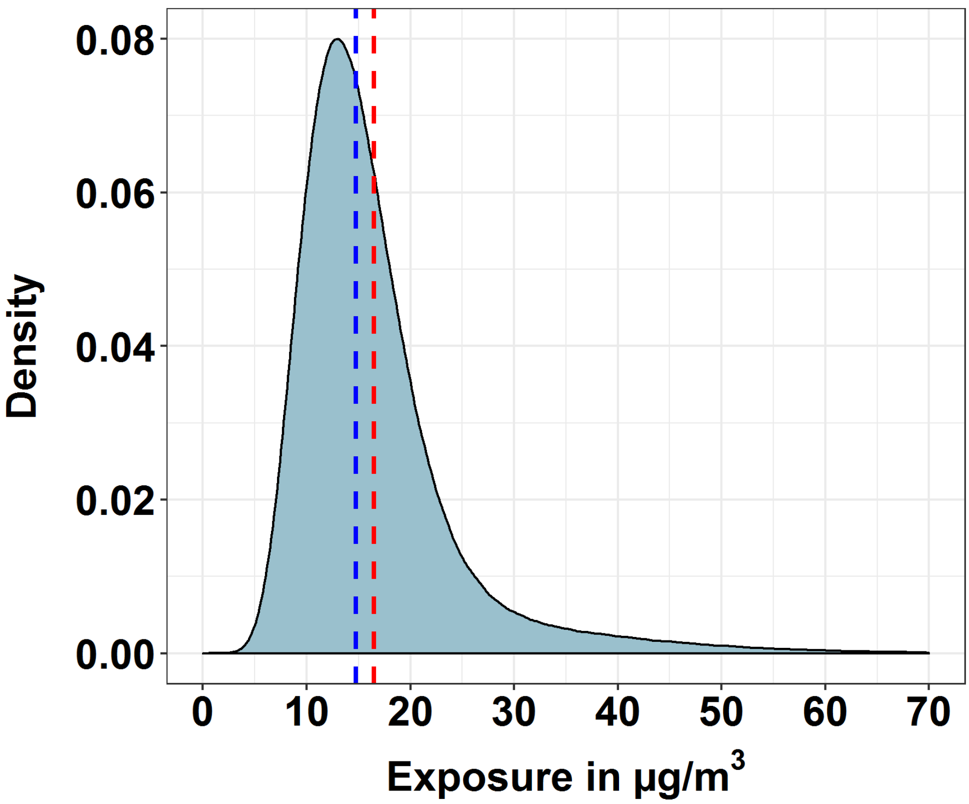

Figure 5-11: Density histogram of the NO_2 exposure for Germans in 2015. The red dash line represents the arithmetic mean value (16.4 µg m^{-3}), while the blue line indicates the geometric mean (15.3 µg m^{-3}) of the Monte Carlo realisations.

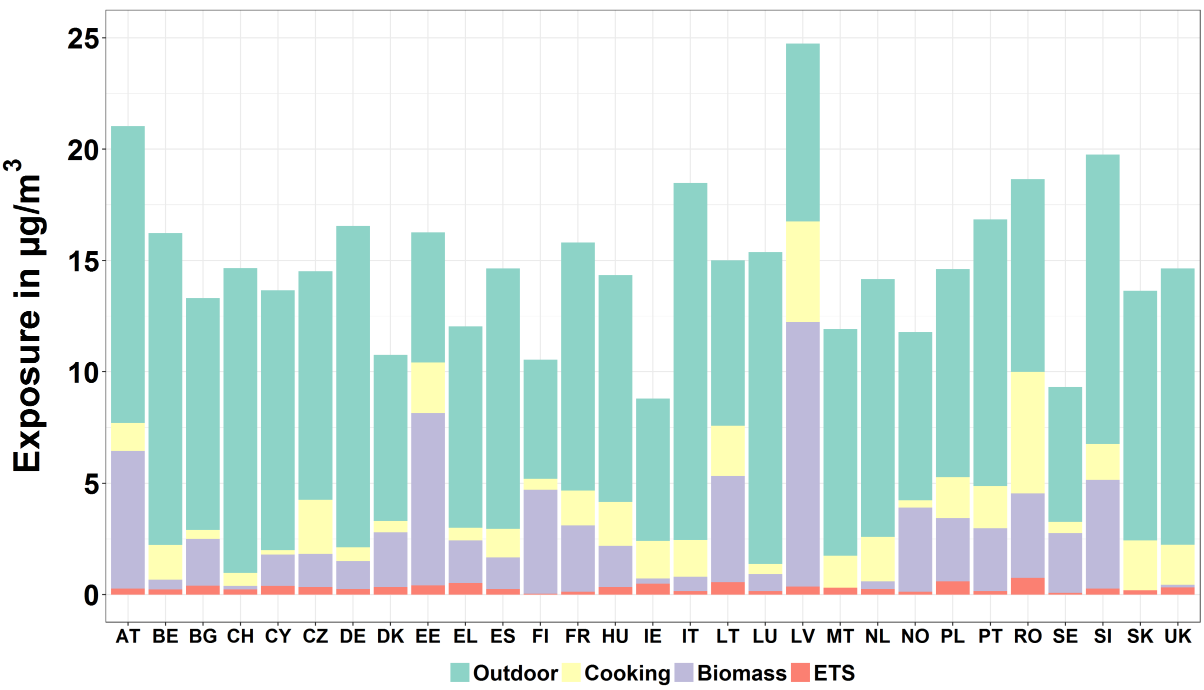

Figure 5-12: Population-weighted arithmetic mean NO_2 exposure by country and source in 2015. For Europeans the mean exposure was 15.5 µg m^{-3}.

As pointed out in Section 5.1.2, the ambient concentration acted as the key source for all the countries. The highest exposure owing to infiltration was found in some SE and NWE countries, e.g. Italy (16.0 μg m^{-3}), Germany (14.4 μg m^{-3}), Belgium (14.1 μg m^{-3}) and Luxembourg (14.0 μg m^{-3}). The NO_2 ambient concentration is significantly influenced by the degree of urbanisation and traffic density: the denser the population and traffic are, the higher the background concentration is. In these countries, especially in Italy, the outdoor concentrations of NO_2 in urban areas were extremely high. To make things worse, the majority of the population in these countries lived in urban or semi-urban areas as recorded by the EU-SILC data. In contrast, the contribution made by the outdoor air was not so significant in some NE countries like Finland and Sweden. According to the EU-SILC data, most of the population in these countries were residing in rural areas. In Sweden, this portion even reached a remarkably high value of 63%.

Biomass is a significant contributory factor in respect to the indoor sources. For example, Latvia was suffered from the highest NO_2 of 24.7 μg m^{-3}, within which 48% was owing to the wood burning at home. The significant influence of biomass has been also found by Estonia (7.7 μg m^{-3}) and Austria (6.2 μg m^{-3}).

What also stands out in Figure 5-12 is the notable variance of cooking exposure among different countries. High exposures were mainly observed by EE countries, e.g. Romania (5.5 μg m^{-3}), Latvia (4.5 μg m^{-3}), Czech Republic (2.4 μg m^{-3}) and Estonia (2.3 μg m^{-3}). Except for the room size and cooking time, the elevated exposure can also be explained by the dominance of gas cooking stove in these countries. Nevertheless, Bulgaria (0.40 μg m^{-3}) as an exception, was least affected by the cooking exposure thanks to the high prevalence of electric stove. The comparably low exposure was also revealed by NE countries including Norway (0.3 μg m^{-3}), Finland (0.5 μg m^{-3}), Denmark (0.5 μg m^{-3}) and Sweden (0.5 μg m^{-3}).

5.2.2.2 NO_2 exposure in 1990

Figure 5-13 displays the population-weighted arithmetic mean NO_2 exposure by country and source in 1990. Compared to 2015, the outdoor concentration had even more significant effect on the overall exposure. Especially in Italy (22.2 μg m^{-3}) and some NWE countries such as United Kingdom (21.7 μg m^{-3}), Germany (21.1 μg m^{-3}) and Belgium (21.0 μg m^{-3}), the infiltration took up to almost 90% of the total exposure. In contrast, the lowest exposure due to infiltration was found in Ireland (6.8 μg m^{-3}), Malta (7.4 μg m^{-3}) and Cyprus (7.5 μg m^{-3}).

Again the most vital indoor sources for NO_2 were cooking and wood burning. The largest contributions from these two sources were found in EE countries, e.g. Latvia (17.6 μg m^{-3}), Estonia (10.2 μg m^{-3}), Romania (9.5 μg m^{-3}) and Lithuania (7.4 μg m^{-3}).

The European average exposure originated from indoor smoking amounted to 0.9 μg m^{-3}, which was almost three times of the value in 2015. Still this value was not vital compared to other sources.

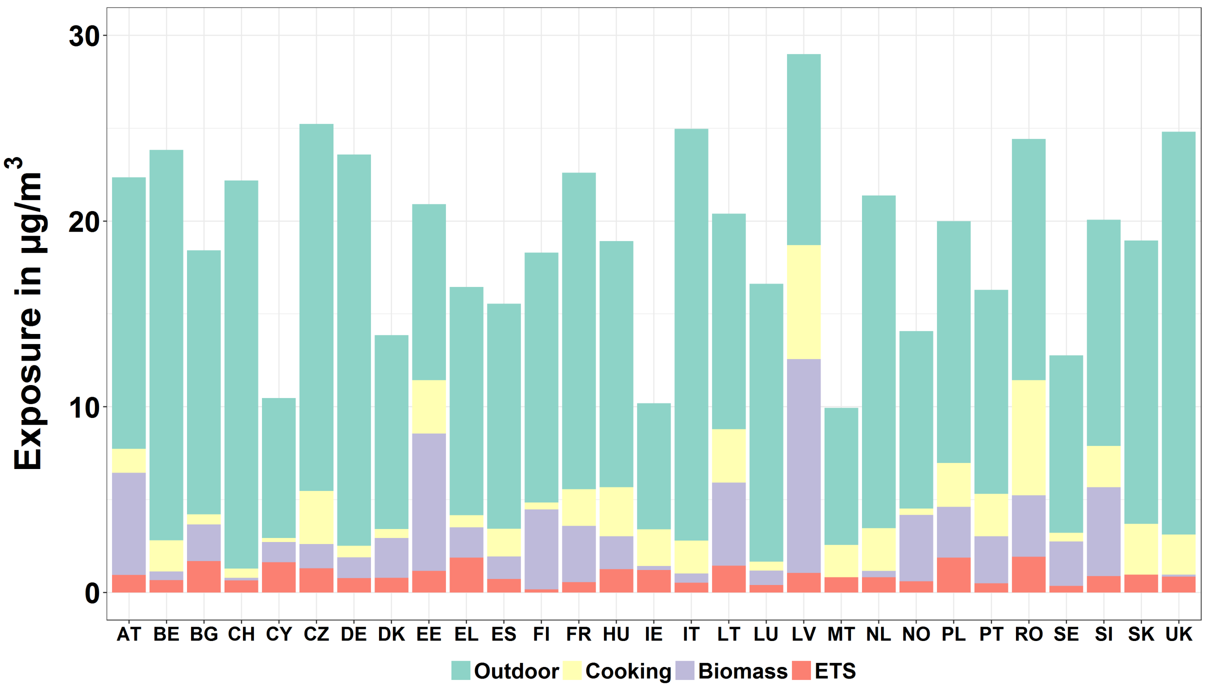

Figure 5-13: Population-weighted arithmetic mean NO_2 exposure by country and source in 1990. For Europeans the mean exposure was 21.4 µg m^{-3}.

5.2.2.3 NO_2 exposure in 1950

In contrast to the results for 2015 and 1990, the total NO_2 exposure in 1950 was less influenced by the background concentration (Figure 5-14). The highest level of infiltration was observed mainly in NWE and EE countries, e.g. Czech Republic (12.1 µg m^{-3}), Belgium (7.9 µg m^{-3}), Austria (7.6 µg m^{-3}), Slovakia (7.4 µg m^{-3}) and Germany (7.2 µg m^{-3}). Quite the opposite, the population of Denmark (0.9 µg m^{-3}), Ireland (1.0 µg m^{-3}) and Cyprus (1.4 µg m^{-3}) were burdened with the lowest exposure from outdoor air.

As displayed in Table 5-2, the contribution made by the outdoor concentration to the overall exposure has dropped to 50%. In contrast, the portion the indoor sources took has increased. Especially for most of the EE countries, including Latvia (17.9 µg m^{-3}), Romania (12.8 µg m^{-3}), Estonia (12.3 µg m^{-3}), and Lithuania (8.9 µg m^{-3}), the proportion even exceeded 70%.

5.3 Exposure by socio-economic status

As introduced in Section 1.2, the environmental inequalities are experienced by certain vulnerable groups. In the following sections, the influence of the socio-demographic factors on exposure to both pollutants is discussed at European level.

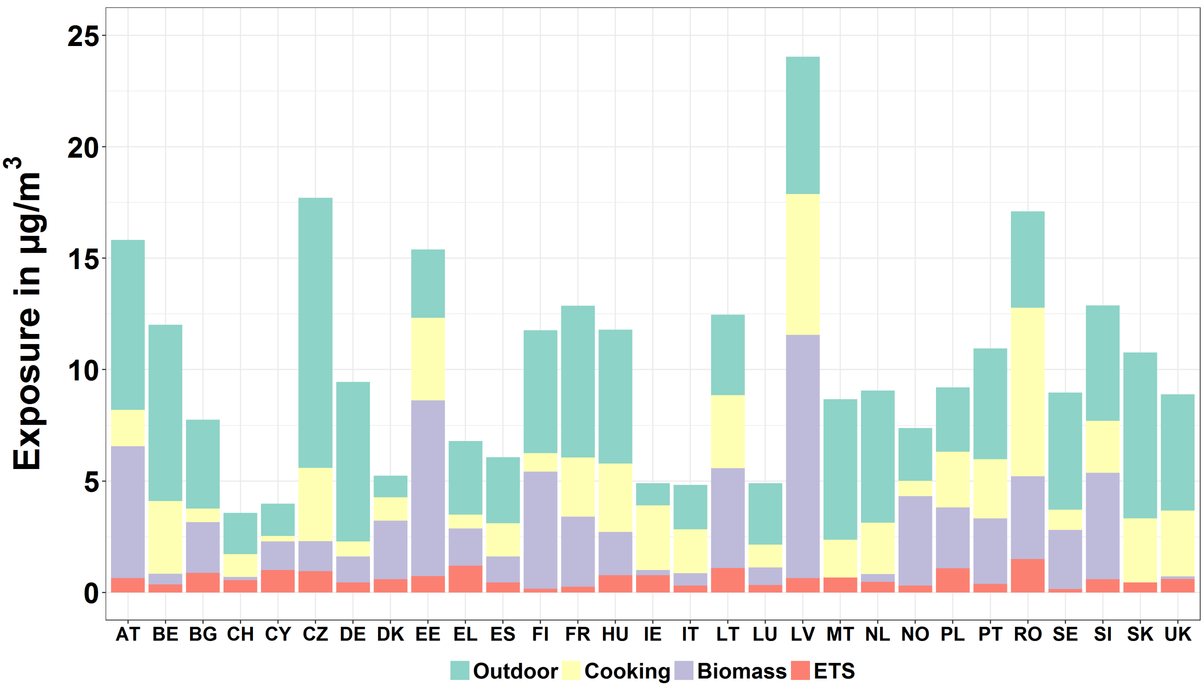

Figure 5-14: Population-weighted arithmetic mean NO_2 exposure by country and source in 1950. For Europeans the mean exposure was 10.4 µg m^{-3}.

5.3.1 By gender

5.3.1.1 PM2.5

Figure 5-15 shows the temporal development of the population-weighted arithmetic mean PM2.5 exposure by source and gender for European countries from 1950 to 2015. As shown in the figure, before 1980 the total exposure of men was lower than the level of women. The opposite situation took place in the 1980s and continued until 2010, when again the PM2.5 exposure suffered by the women exceeded the level for men.

The exposure owing to ETS for men is higher than that for women for all the time periods (r=-0.41 for 1950, r=-0.31 for 1980, r=-0.18 for 2015, p=0). As displayed by Table 4-6, the smokers are more influenced by the indoor smoking than the non-smokers. According to the data of Eurobarometer (2010), the prevalence of male smokers was higher than the females (35% to 25%). The difference between the two genders was even more apparent in the earlier time periods (Hill and Laplanche 2005). Therefore, the heavier PM2.5 exposure due to ETS was experienced by men.

In contrast, the exposure due to cooking tells another story: women experience higher exposure than men. This is due to the large difference of cooking time between the two groups. According to the MTUS data, the average time women spent on cooking was over nine times higher than that of men (116 min to 12 min) in the 1960s. This difference followed a trend of declining, still the exposure caused by cooking for women was twice higher than for men in 2015. This similar tendency has also been observed by “other sources” since women spent more time on activities related to housework.

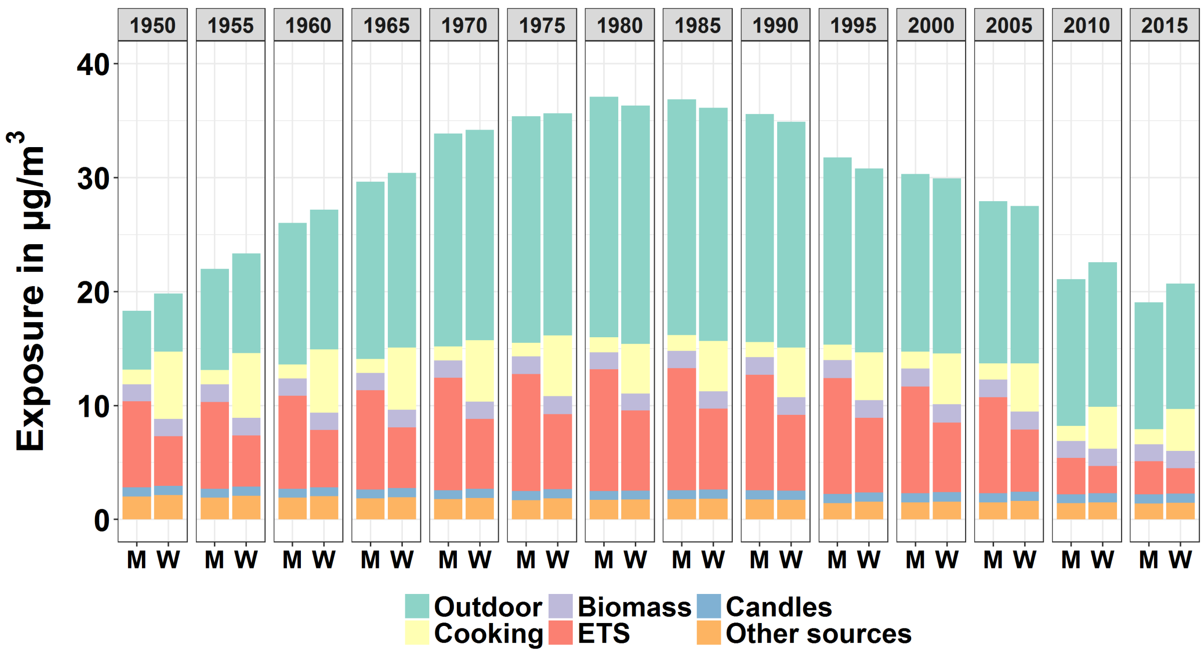

Figure 5-15: Population-weighted arithmetic mean PM2.5 exposure by source and gender for European countries from 1950 to 2015. The "M" stands for "Men" and "W" stands for "Women" (more details in Table 5-3).

Table 5-3: Categories and definitions of the social-economic variables.

Variable	Abbreviation	Value	Categories
Gender	M	1	Men
	W	2	Women
Income level	H	3	Highest 25%
	M	2	Middle 50%
	L	1	Lowest 25%
Degree of urbanisation	U	1	Urban or suburban areas
	R	0	Rural areas
Civil status	C	1	In a couple, lives with spouse/partner
	S	0	Not in a couple
Employment status	E	1	Employed
	U	0	Unemployed
Education level	H	1	Above secondary education (ISCED level 5 or above)
	L	0	Completed/uncompleted secondary education (ISCED level 4 or below)

The two major indoor sources explain the temporal course of the overall PM2.5 by gender. Before 1980 the higher exposure due to cooking resulted in the heavier total exposure experienced by women, despite the severe exposure stemming from smoking suffered by men. However, between 1980 and 2005 the smoking took the dominant role of the indoor sources since the gap of cooking time between the two groups has dropped. After 2005, nevertheless, the exposure from smoking has reduced sharply due to the Europe-wide introduction of smoking bans. Thus, the exposure for women overtook again afterwards.

5.3.1.2 NO_2

Figure 5-16 shows the course of the population-weighted arithmetic mean NO_2 exposure by source for both males and females from 1950 to 2015. Unlike the course for PM2.5, the NO_2 burdened by women was higher than men for all the time periods. The main reason was the large difference originated from cooking. As mentioned in the Section 5.3.1.1, women spent generally more time cooking in the kitchen than men. However, the declining tendency of the difference since the 1950s can still be noticed. As shown by the MTUS data, the gap for cooking time between the two groups has dropped from 104 min in the 1960s to only 59 min in the 2010s.

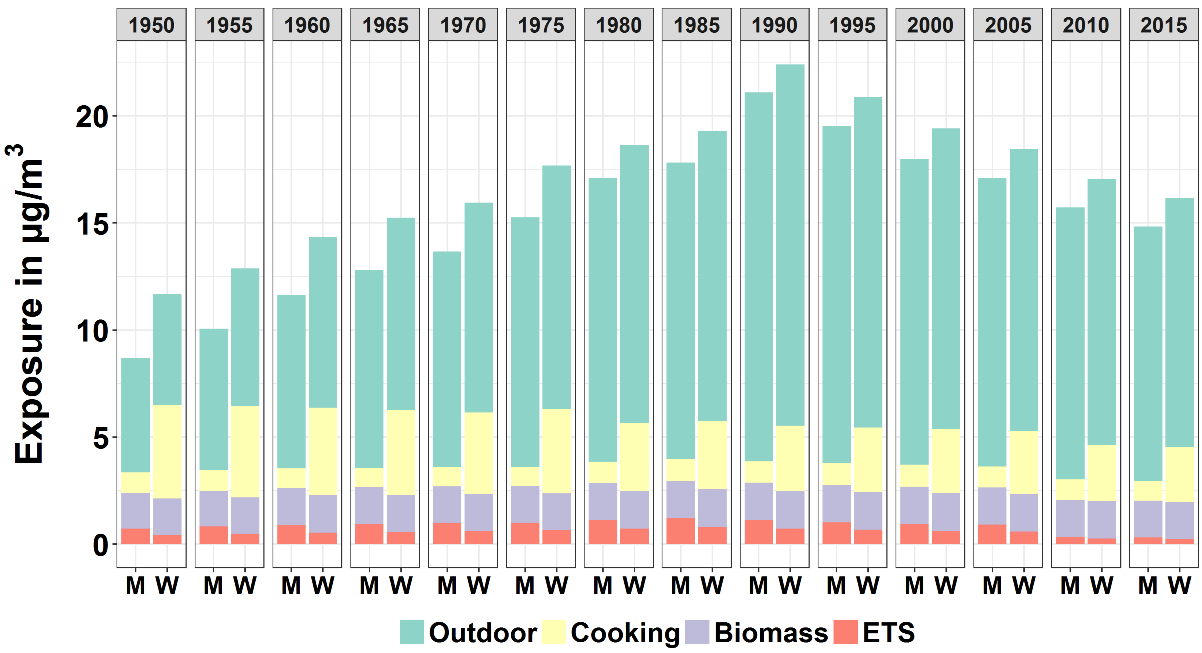

Figure 5-16: Population-weighted arithmetic mean NO_2 exposure by source and gender for European countries from 1950 to 2015. The "M" stands for "Men" and "W" stands for "Women" (more details in Table 5-3).

Regarding the ETS, males experienced higher exposure compared to females. However, the influence of smoking was not noticeable with respect to the total NO_2 exposure.

5.3.2 By income level

5.3.2.1 PM2.5

Figure 5-17 shows the temporal development of the population-weighted arithmetic mean PM2.5 exposure by source and household income level for European countries from 1950 to 2015. As displayed by the figure, the population subgroup with low income level was burdened with the highest exposure to PM2.5 for all the time periods. What stands out in the figure is the difference of exposure from cooking among the three subgroups. As discussed in Section 4.4.1, the average residential area is positively related to the household income level (r=0.29, p=0). Thus, the population with low income tended to live in rooms with smaller area, which slowed down the dilution of the indoor pollutants.

The other factor affecting the exposure due to cooking among the three subgroups is the time spent on cooking. Figure 5-18 shows the density histogram of the time spent on cooking each day for women with three different income levels in 1980 as an example. As demonstrated by the figure, the cooking time was negatively correlated with the income level (r=-0.15, p=0). This again increases the exposure due to cooking for the low-income group.

With respect to other indoor sources, the negative relationship with income level was observed as well. However, the influence was less significant than with cooking.

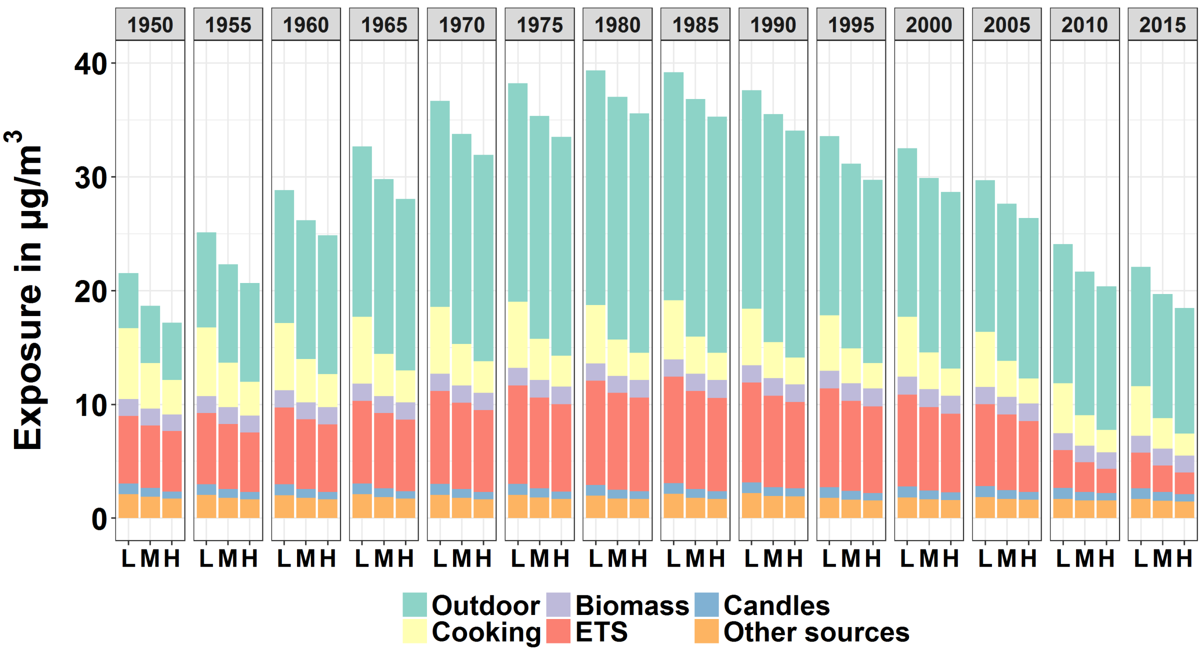

Figure 5-17: Population-weighted arithmetic mean PM2.5 exposure by source and income level for European countries from 1950 to 2015. The “L”, “M” and “H” stand for the “Low”, “Median” and “High” level of income (more details in Table 5-3).

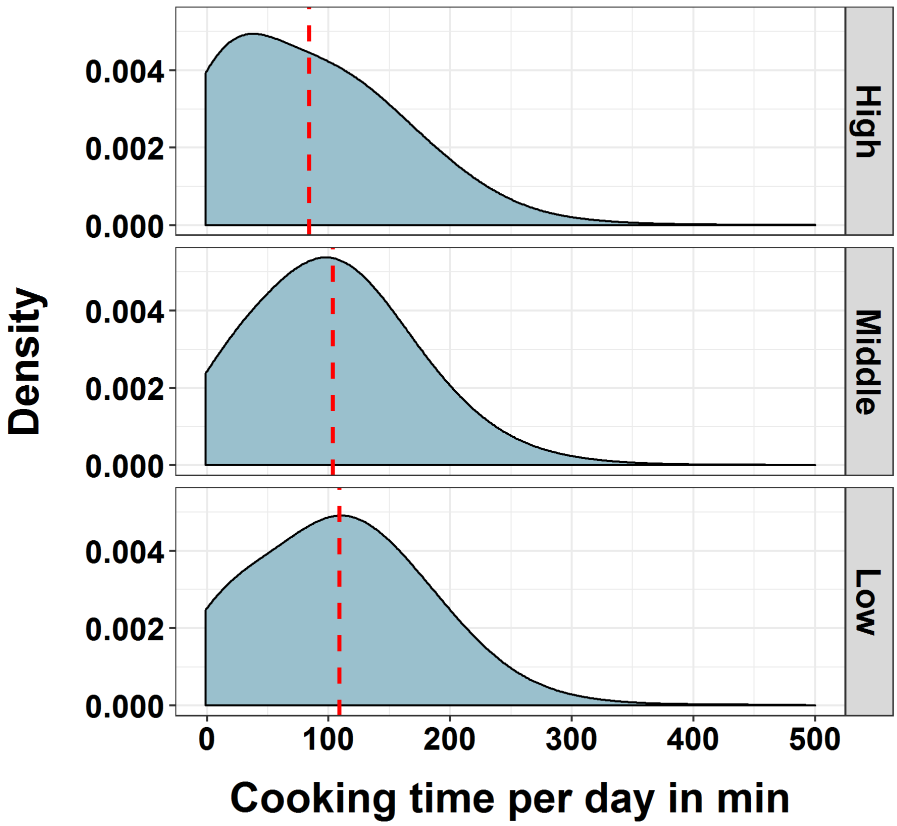

Figure 5-18: Density histogram of the daily total cooking time for women with "Low", "Middle" and "High" household income level in 1980. The red dash line represents the arithmetic mean value (84 min for "High", 103 min for "Middle" and 109 min for "Low").

5.3.2.2 NO_2

Figure 5-19 represents the temporal development of the population-weighted arithmetic mean NO_2 exposure by source and household income level for European countries from 1950 to 2015. Similar to PM2.5, the total NO_2 also revealed the negative correlation with the household income level. The main reason for this result is the exposure from cooking as well. As demonstrated by the figure, the difference among the three subgroups was more evident in the earlier years.

5.3.3 By degree of urbanisation

5.3.3.1 PM2.5

Figure 5-20 shows the population-weighted arithmetic mean PM2.5 exposure by source and degree of urbanisation for European countries from 1950 to 2015. As presented by the figure, the overall exposure in urban or suburban areas was higher than that in rural areas (r=0.15 for 1950, r=0.11 for 1990, r=0.11 for 2015, p=0).

The first reason for this finding is the heavier exposure due to the outdoor air as a

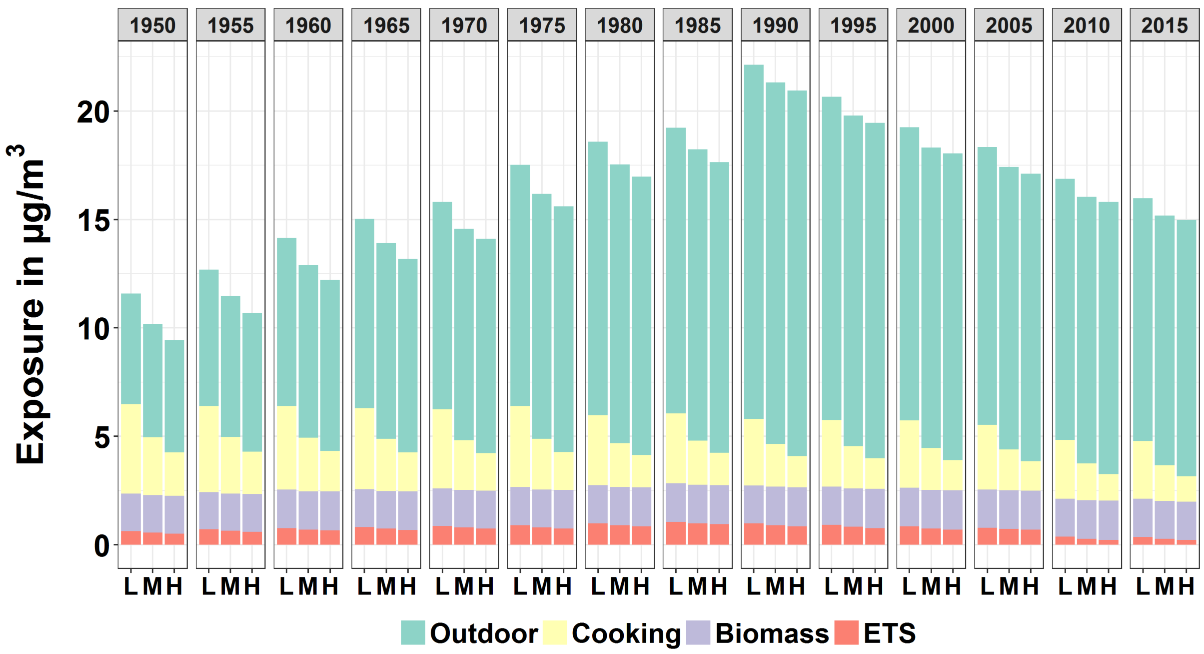

Figure 5-19: Population-weighted arithmetic mean NO_2 exposure by source and income level for European countries from 1950 to 2015. The "L", "M" and "H" stand for the "Low", "Median" and "High" level of income (more details in Table 5-3).

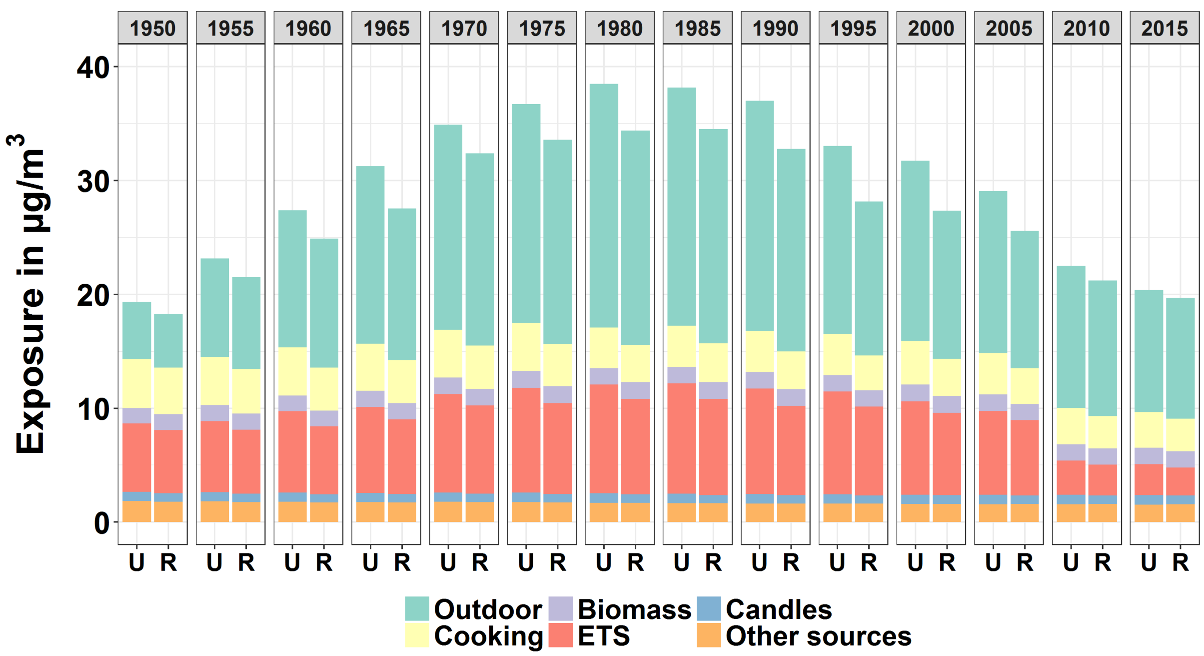

Figure 5-20: Population-weighted arithmetic mean PM2.5 exposure by source and degree of urbanisation for European countries from 1950 to 2015. The "U" stands for "Urban or suburban areas" while the "R" stands for "Rural areas" (more details in Table 5-3).

consequence of the urban increment, which is described as the notably higher pollutant concentration in urban or suburban areas than in rural environments (de Leeuw et al. 2002; Stedman and Derwent 2008; Ortiz 2012).

Secondly, the total exposure originated from indoor sources was higher in the densely or intermediately populated areas as a result of the smaller room area. As discussed above, the room area is a crucial factor affecting the exposure due to the indoors sources. Figure 5-21 shows the density histogram of the dwelling size for (sub)urban and rural areas in Germany. It is not surprising that Germans living in rural areas had larger dwelling space compared to the other group. This finding is applicable to other European countries as well. The tiny room size raised the indoor concentration and further enhanced the overall exposure for the people living in cities.

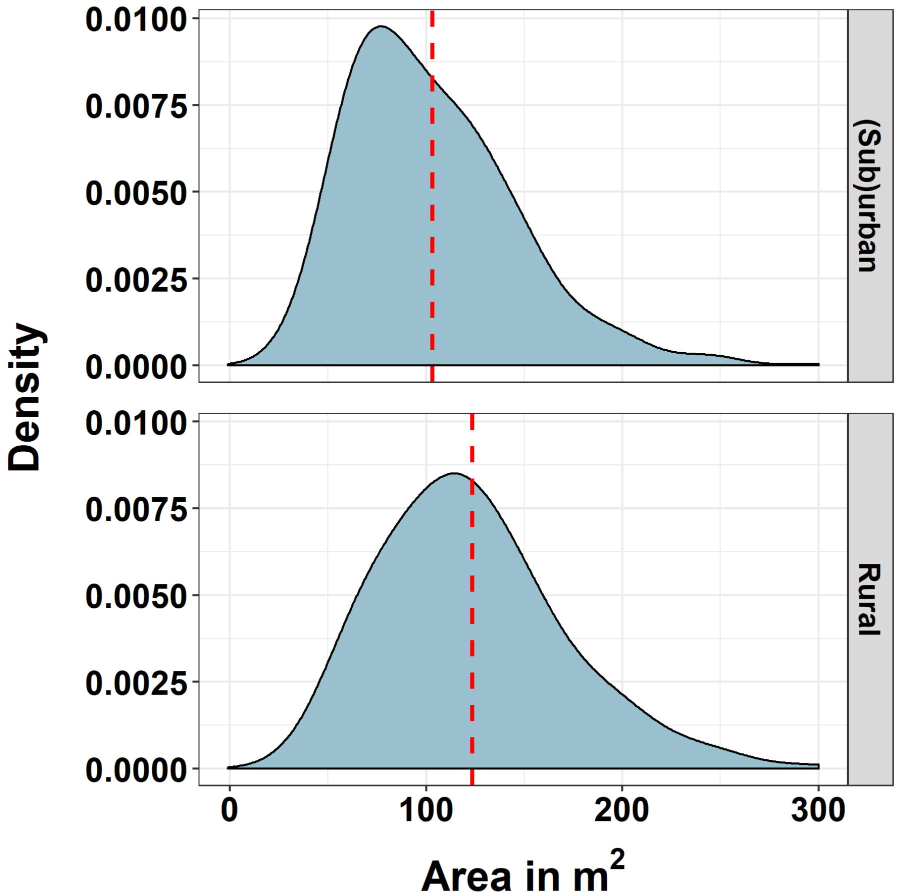

Figure 5-21: Density histogram of the dwelling area for (sub)urban and rural areas in Germany. The red dash line represents the arithmetic mean value (103 m^2 for "Urban" and 123 m^2 for "Rural").

5.3.3.2 NO_2

Figure 5-22 presents the population-weighted arithmetic mean NO_2 exposure by source and degree of urbanisation for European countries from 1950 to 2015. Similar to PM2.5, the exposure burdened by the population in densely or intermediately populated areas was higher than the level in rural areas. Especially in terms of the exposure originated from the outdoor environment, the gap was even more notable (r=0.20 for 1950, r=0.55 for 1990, r=0.61 for 2015, p=0) compared to PM2.5. Likewise, the NO_2 exposure due to the indoor sources was higher in (sub)urban areas as a consequence of smaller residential sizes (see Figure 5-21).

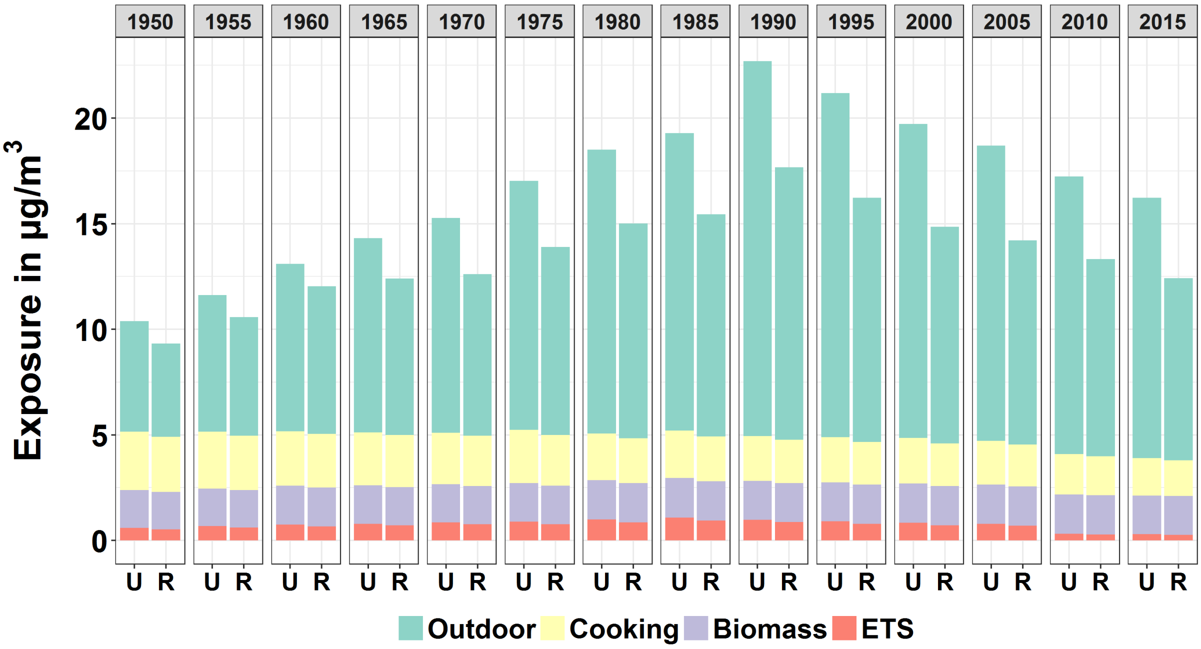

Figure 5-22: Population-weighted arithmetic mean NO_2 exposure by source and degree of urbanisation for European countries from 1950 to 2015. The "U" stands for "Urban or suburban areas" while the "R" stands for "Rural areas" (more details in Table 5-3).

5.3.4 By civil status

5.3.4.1 PM2.5

Figure 5-23 demonstrates the population-weighted arithmetic mean PM2.5 exposure by source and civil status for European countries from 1950 to 2015. As displayed in the figure, the overall exposure for the group "Couple" was slightly higher before 1965, while afterwards the group "Single" took over. Even though no obvious difference was detected for the exposure from the outdoor air, a distinction was shown for the indoor sources. The greatest difference was found by the exposure due to cooking (r=0.15 for 1950, r=0.12 for 1980, r=0.05 for 2015, p=0). The reason behind is the large gap of cooking time between the two subgroups. As shown in Figure 5-24, the average daily cooking time for the married Spanish women in 2010 was almost twice the value of the single group. The gap in cooking time has also been noted by other countries according to the MTUS data.

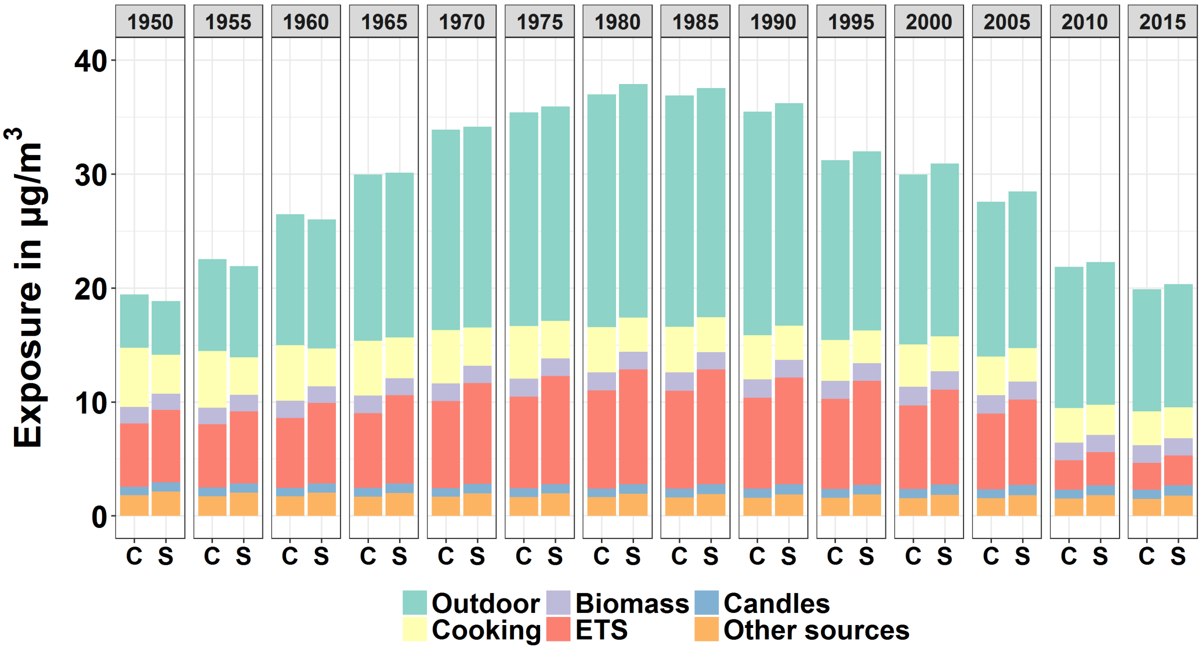

Figure 5-23: Population-weighted arithmetic mean PM2.5 exposure by source and civil status for European countries from 1950 to 2015. The "C" stands for "Couple" and the "S" stands for "Single" (more details in Table 5-3).

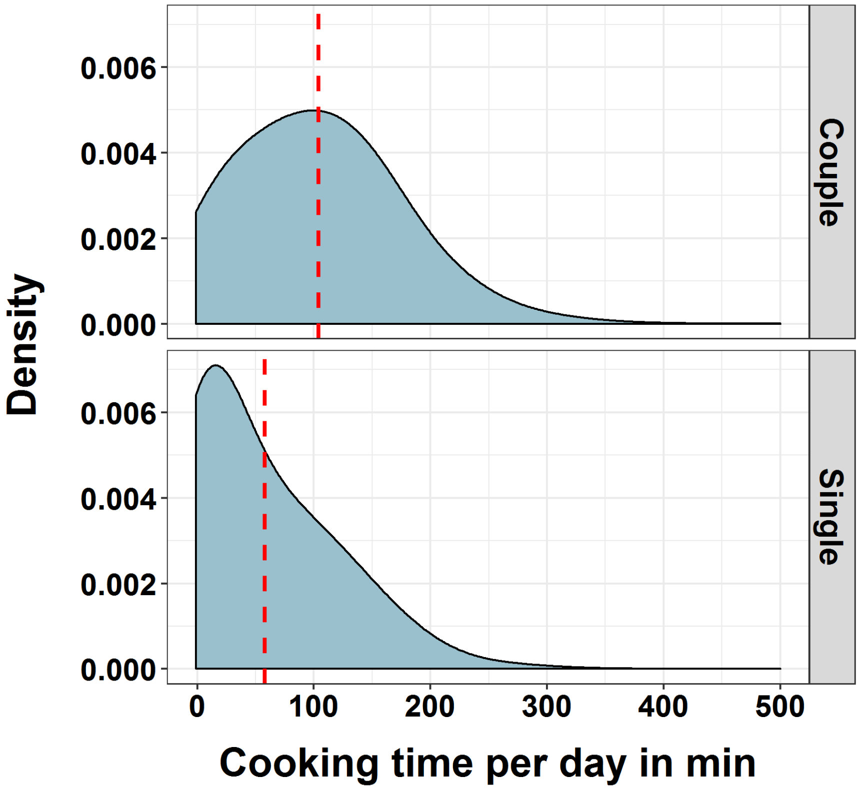

Figure 5-24: Density histogram of the daily total cooking time for married or single women in Spain for 2010. The red dash line represents the arithmetic mean value (104 min for group "Couple" and 58 min for group "Single").

As for other indoor sources, the converse results were observed for the two subgroups. This outcome can be explained by the smaller residence size occupied by the single group (r=0.18, p=0). Figure 5-25 shows the density histogram of the dwelling size in Germany for both groups as an example. In Europe the average residential size for the couples (106 m^2) was 23% larger than the single group (86 m^2).

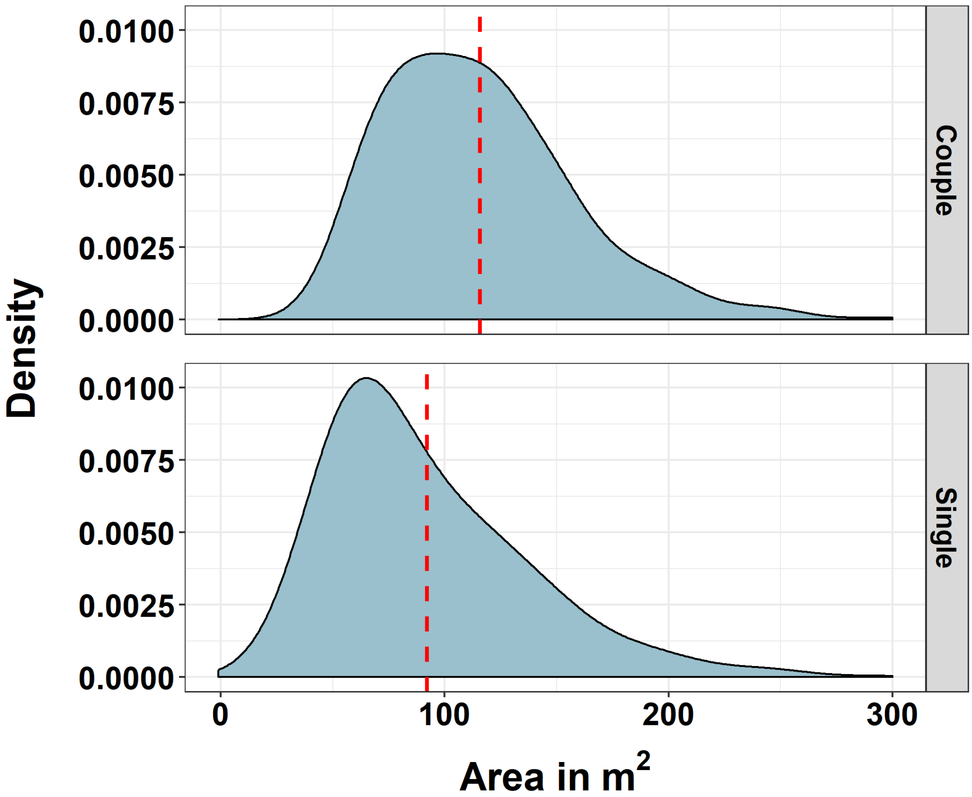

Figure 5-25: Density histogram of the dwelling size for "Couple" or "Single" in Germany for 2010. The red dash line represents the arithmetic mean value (116 m^2 for group "Couple" and 92 m^2 for group "Single").

5.3.4.2 NO_2

Figure 5-26 displays the population-weighted arithmetic mean NO_2 exposure by source and civil status for European countries from 1950 to 2015. Unlike the finding for PM2.5, the total NO_2 exposure experienced by the group "Couple" was generally higher than the "Single" for all the time periods. Since between the two subgroups, the difference of other indoor sources, especially the indoor smoking, became less evident, the role played by cooking turned to be the crucial factor. Similar to the result for PM2.5, the NO_2 owing to cooking was much higher for couples as a consequence of the elevated cooking time.

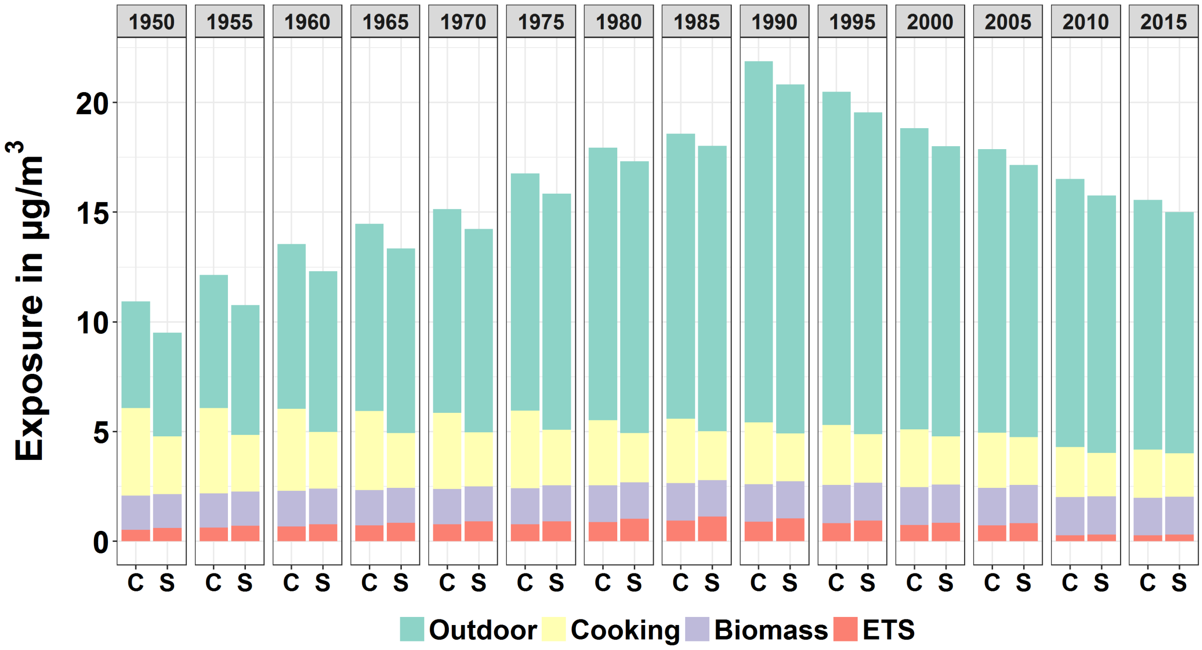

Figure 5-26: Population-weighted arithmetic mean NO_2 exposure by source and civil status for European countries from 1950 to 2015. The "C" stands for "Couple" and the "S" stands for "Single" (more details in Table 5-3).

5.3.5 By employment status

5.3.5.1 PM2.5

Figure 5-27 shows the population-weighted arithmetic mean PM2.5 exposure by source and employment status for European countries from 1950 to 2015. As displayed in the figure, the total exposure was higher for the unemployed for all the time periods. The most outstanding difference was the indoor sources, especially the exposure due to cooking and smoking.

For cooking, the elevation of exposure experienced by the unemployed stemmed from the longer cooking time and smaller dwelling size. As displayed in Figure 5-28, the average time the unemployed women spent on cooking in 2010 was 55% longer than the employed group. This gap was even larger in the earlier time periods (154 min to 54 min in the 1960s). With respect to the residence area, the average value for the employed was 12% larger than the other group (125 m^2 to 111 m^2).

Referring to ETS, the higher level of exposure experienced by the unemployed was, on the one hand, due to the smaller dwelling size as mentioned above, on the other hand, as a result of the high prevalence of smokers in this group. According to the Eurobarometer (2012), the unemployed were "the most likely of all groups to report they smoke (49%)". In comparison, this ratio for the employed group was 30%.

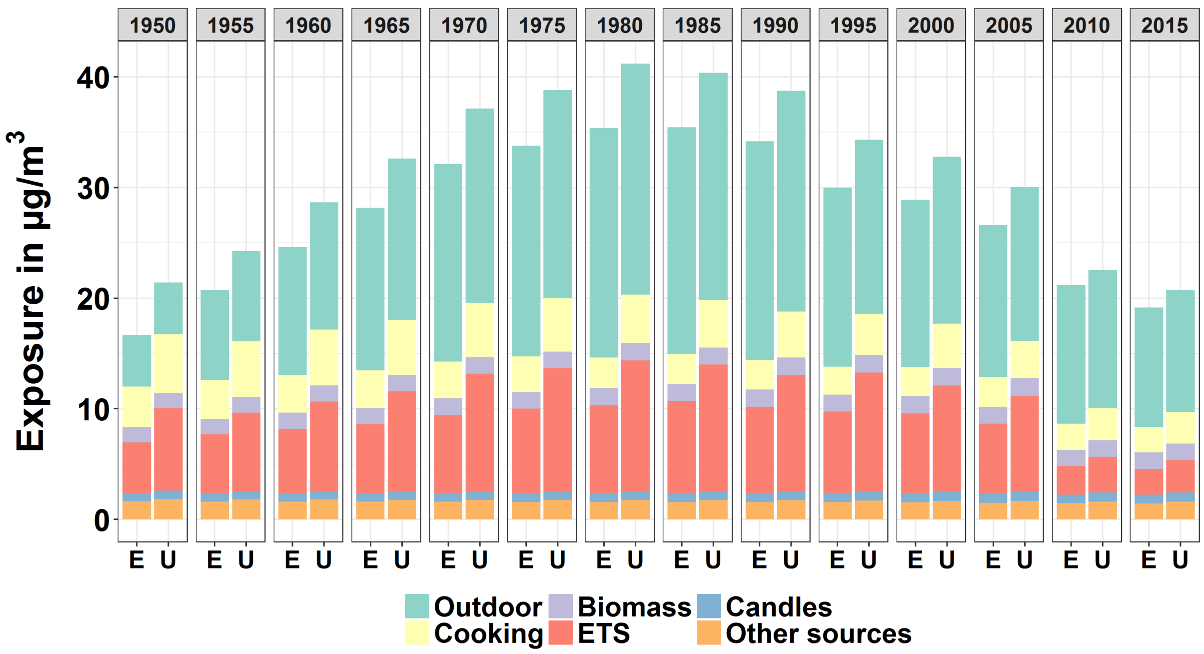

Figure 5-27: Population-weighted arithmetic mean PM2.5 exposure by source and employment status for European countries from 1950 to 2015. The “E” stands for the “Employed” and the “U” for the “Unemployed” (more details in Table 5-3).

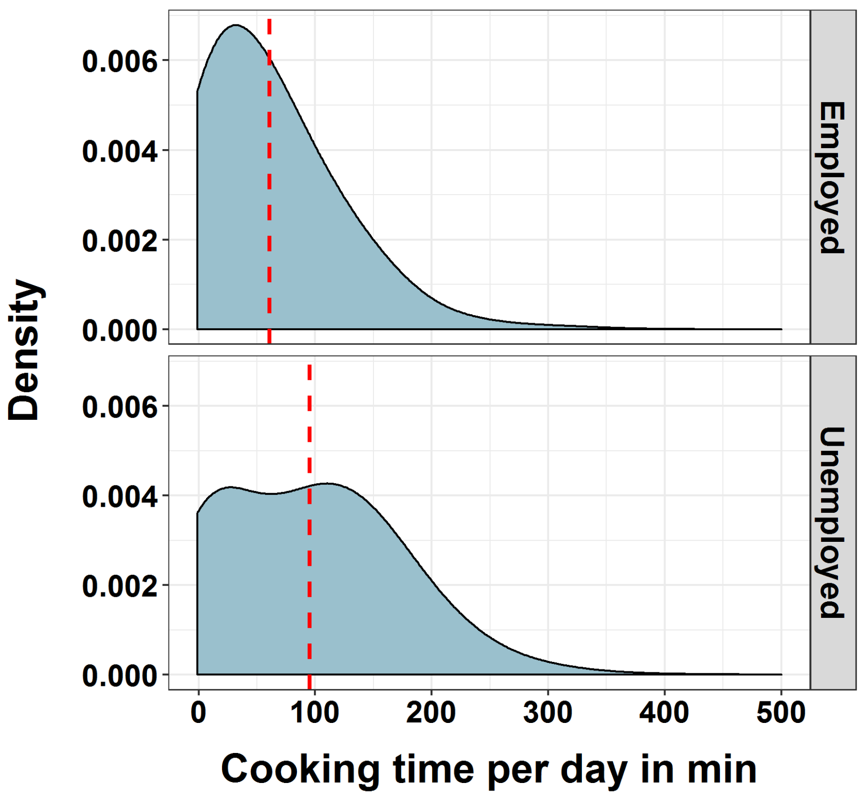

Figure 5-28: Density histogram of the daily total cooking time for employed or unemployed women in Spain for 2010. The red dash line represents the arithmetic mean value (61 min for group “Employed” and 95 min for group “Unemployed”).

5.3.5.2 NO_2

Figure 5-29 displays the population-weighted arithmetic mean NO_2 exposure by source and employment status for European countries from 1950 to 2015. Similar to the result for PM2.5, the overall exposure to NO_2 for the unemployed was higher than the other group. The dominant difference of exposure was found for cooking as well (r=-0.21 for 1950, r=-0.26 for 1990, r=-0.23 for 2015, p=0), which was the result of longer cooking time and smaller dwelling size as discussed above. The difference was observed for NO_2 from smoking as well, however, the contribution to the total exposure was much less influential.

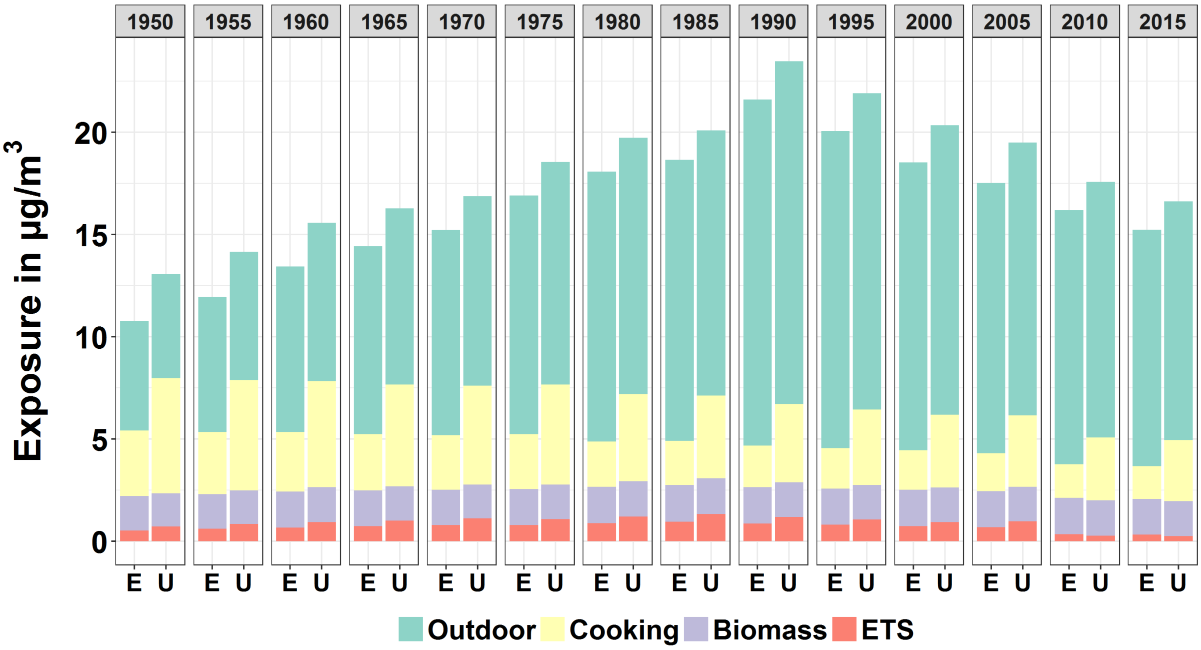

Figure 5-29: Population-weighted arithmetic mean NO_2 exposure by source and employment status level for European countries from 1950 to 2015. The "E" stands for the "Employed" and the "U" for the "Unemployed" (more details in Table 5-3).

5.3.6 By education level

5.3.6.1 PM2.5

Figure 5-30 shows the population-weighted arithmetic mean PM2.5 exposure by source and education level for European countries from 1950 to 2015. As shown by the figure, the total exposure burdened by the population with low education was higher, which was mainly resulted from the contribution of indoor sources. First of all, the longer cooking time for this group gave rise to the higher exposure from cooking. Figure 5-31 shows the density histogram of the daily total cooking time for women with high or low education level in Spain for 2010 as an example.

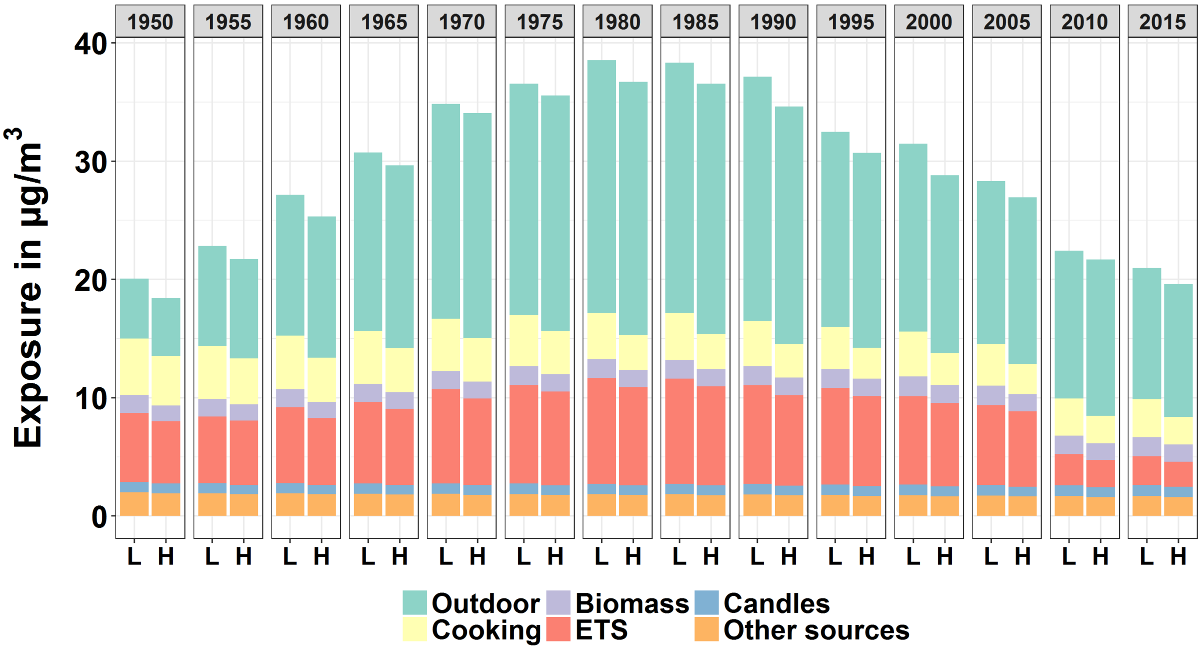

Figure 5-30: Population-weighted arithmetic mean PM2.5 exposure by source and education level for European countries from 1950 to 2015. The "L" stands for the "Low education" and "H" for the "High education" (More details in Table 5-3).

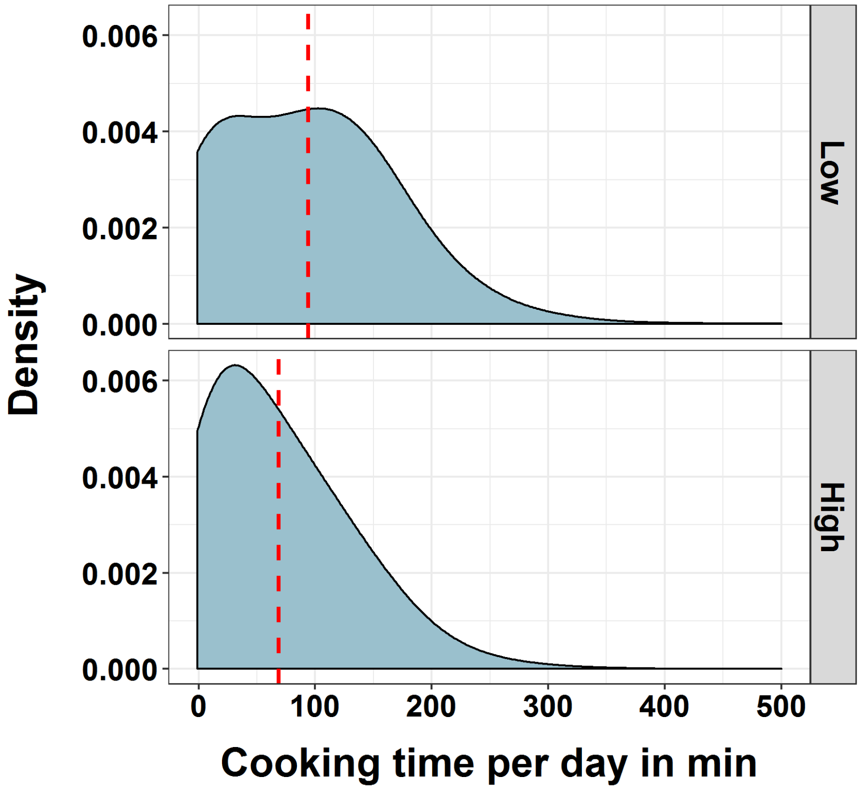

Figure 5-31: Density histogram of the daily total cooking time for women with high or low education level in Spain for 2010. The red dash line represents the arithmetic mean value (94 min for group "Low" and 68 min for group "High").

Secondly, distinction has been reported for the prevalence of smokers. According to Eurobarometer (2012), the most likely group of smokers were the respondents who just finished full-time education (33%). In comparison for the subjects with higher education level, this portion dropped to less than a quarter (22%) .

Last but not least, the total indoor exposure for the highly-educated group was mitigated due to the larger room size. A positive relationship has been observed between the education level and dwelling size (r=0.11, p=0).

Based on the reasons stated above, the exposure to PM2.5 displayed a general pattern of elevated level by the group with lower education.

5.3.6.2 NO_2

Figure 5-32 displays the population-weighted arithmetic mean NO_2 exposure by source and education level for European countries from 1950 to 2015. The total exposure burdened by the people with lower education was higher compared to the other group. The main reason behind was the heavier indoor sources, especially from cooking. However, the gap between the two groups was not as obvious as for PM2.5 since the contribution from ETS became less vital.

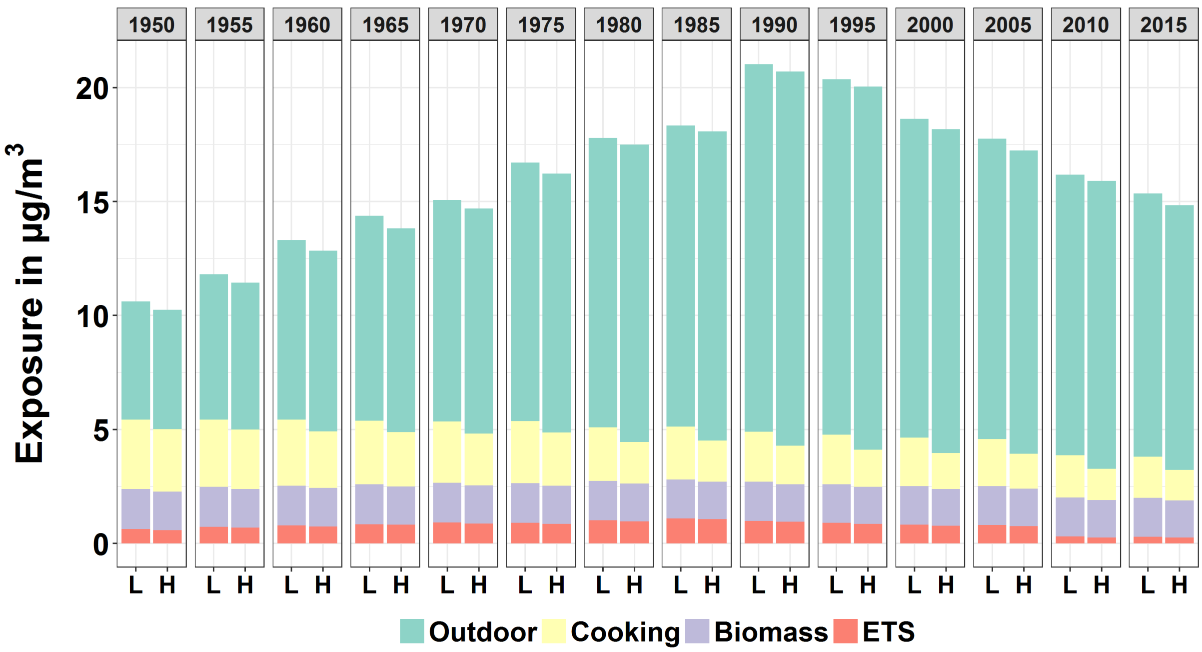

Figure 5-32: Population-weighted arithmetic mean NO_2 exposure by source and education level for European countries from 1950 to 2015. The "L" stands for the "Low education" and "H" for the "High education" (More details in Table 5-3).

5.4 Influence of the ventilation system

As stated in Section 4.2.1, the indoor pollutant concentration is influenced by the ventilation system used. Figure 5-33 displays the average PM2.5 exposure of Germans in 2015 for “natural”, “retrofitted”, “mechanical” and “AHU” buildings. Compared to the naturally-ventilated, non-insulated buildings, the exposures for “AHU” buildings were obviously lower (40%). The operation of “AHU” relieved both the exposures due to the outdoor and indoor sources remarkably.

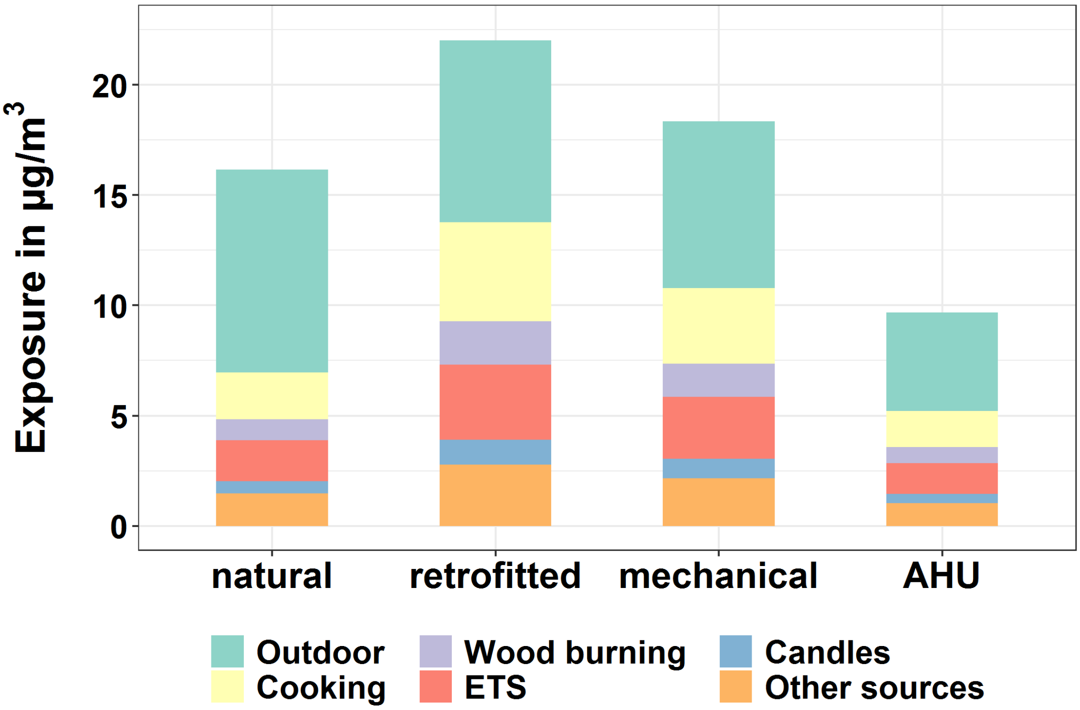

Figure 5-33: Mean PM2.5 exposure of Germans in 2015 for “natural”, “retrofitted”, “mechanical” and “AHU” buildings.

Even though the goal is set to achieve adequate air exchange, the air exchange rates of “mechanical” buildings are very often under the requirement in real life. This led to a higher overall exposure compared to the “natural” buildings. However, the lower exposure originated from the infiltration (15%) should be noticed. This was mainly resulted from the installation of filters.

In contrast, the PM2.5 exposure increased by 35% if the buildings were solely insulated without the operation of additional mechanical ventilation. Even though the exposure due to infiltration was reduced, the insulation raised the exposure from indoor sources by almost 88%.

However, does this conclusion apply to NO_2 as well? Figure 5-34 shows the average NO_2 exposure of Germans by building type in 2015. The “AHU” buildings still mitigated the exposure noticeably, yet naturally ventilated buildings suffered from slightly higher level (10%) of exposure compared to the “retrofitted” ventilation. This was owing to the fact that the simulated NO_2 exposure was less affected by indoor sources compared to PM2.5. Especially for a country like Germany, the indoor sources were only responsible for less than 13% of the total NO_2 exposure. Thanks to the operation of filters, the exposure from infiltration for the "mechanical" ventilation was notably reduced compared to the "natural" buildings.

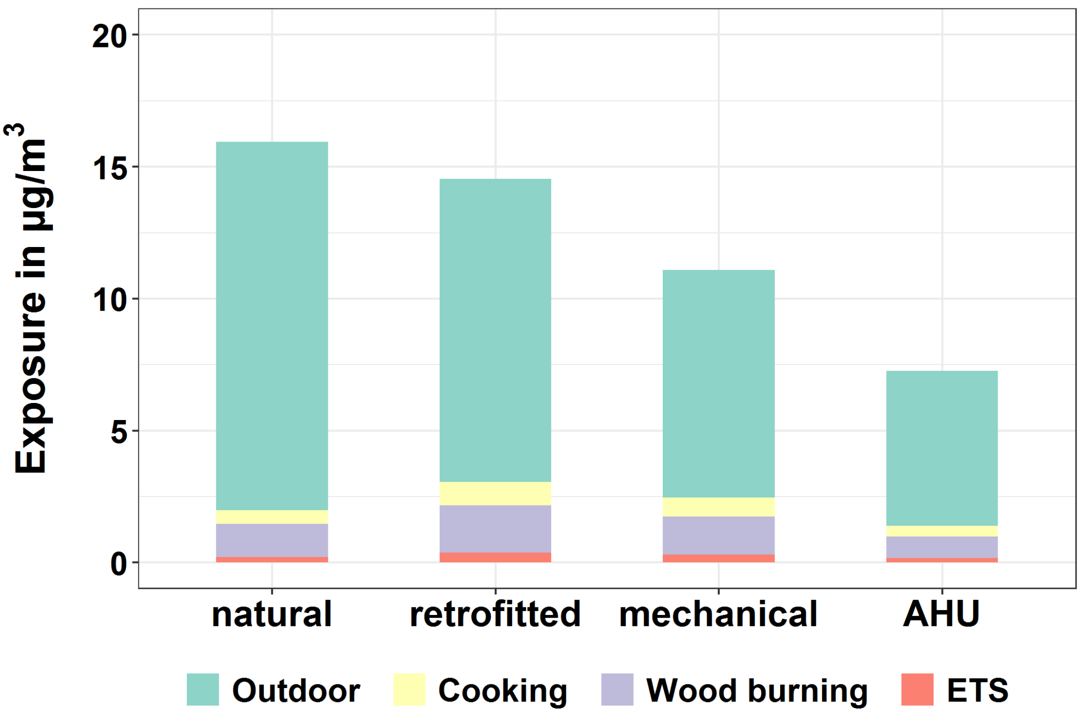

Figure 5-34: Mean NO_2 exposure of Germans in 2015 for "natural", "retrofitted", "mechanical" and "AHU" buildings.

In contrast, for a country like Latvia where the indoor sources had a major influence, the overall exposure for "retrofitted" buildings was extraordinarily high (see Figure 5-35).

These findings lead to a question: should the air exchange rate be decelerated, or should the buildings even be insulated? It is without doubt that insulation is beneficial for reducing the overall energy consumptions and pollutant emissions, which is essential for minimizing the exposure in the long term (Levy et al. 2003; Korsholm et al. 2012; Bønløkke et al. 2015).

Nevertheless, the problem becomes very tricky when it refers to the personal exposure. This controversial question has been observed by Hänninen et al. (2005) for the study of the modern ventilation systems. On the one hand, insulation results in tighter building envelopes and lower AER, which lead to decreased infiltration from outdoor air according to Equation 3-4. On the other hand, the lower AER raises the exposure stemming from the indoor sources (Lin and Deng 2003; Thornburg et al. 2004).

Hence, when planning the building ventilation systems and the exposure reduction policies, the issues discussed above should be considered carefully. If conditions allow, the introduction of mechanical ventilation with heat recovery and a filtering system is

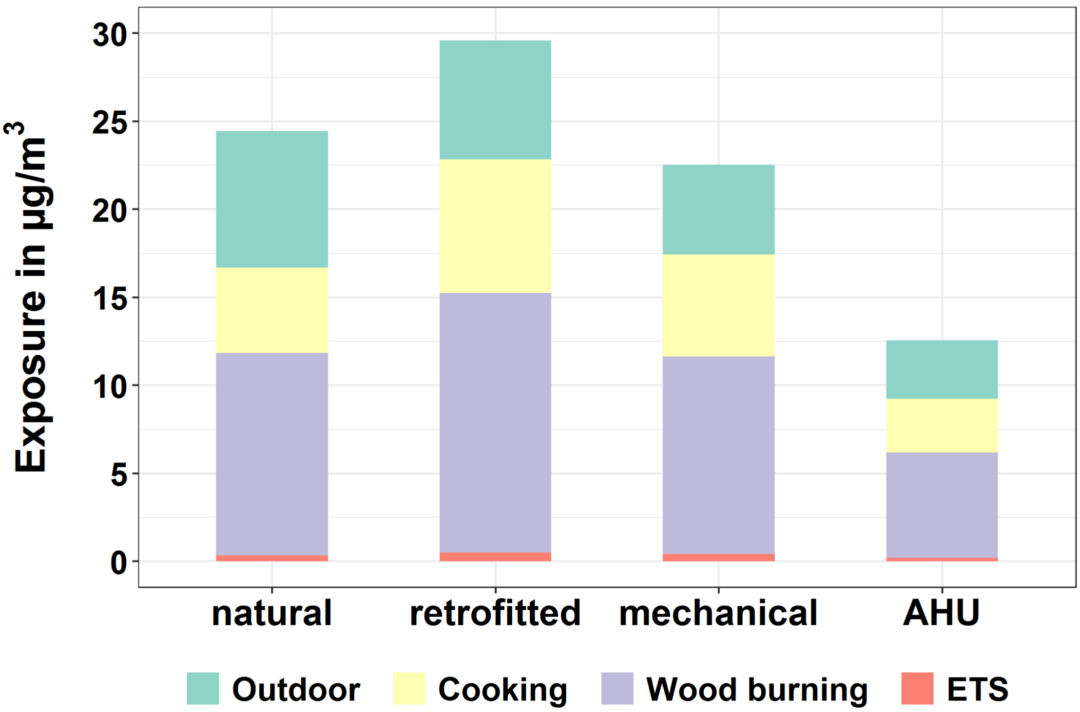

Figure 5-35: Mean NO_2 exposure of Latvian people in 2015 for "natural", "retrofitted", "mechanical" and "AHU" buildings.

recommended.

5.5 Health impacts, DALYs and damage costs

Figure 5-36 shows the density histogram of the DALYs due to years of life lost (PM2.5) in Germany, 2015. Similar to the result of exposure, the DALYs are likely to be log-normally distributed. This observation is also applicable for the results of damage costs.

Figure 5-37 presents the health impacts for the total EU27+2 population in 2015 by different endpoints. More detailed information for health impacts, DALYs and damage costs is presented in Table 5-4. According to WHO (2013a), the long-term effects of NO_2 partly overlap with those caused by the particles. Thus, the author follows the suggestion in the literature and an overlap factor of 33% is taken for NO_2 in this thesis. This means only 67% of the health impacts, DALYs and damage costs due to NO_2 are considered in the calculation.

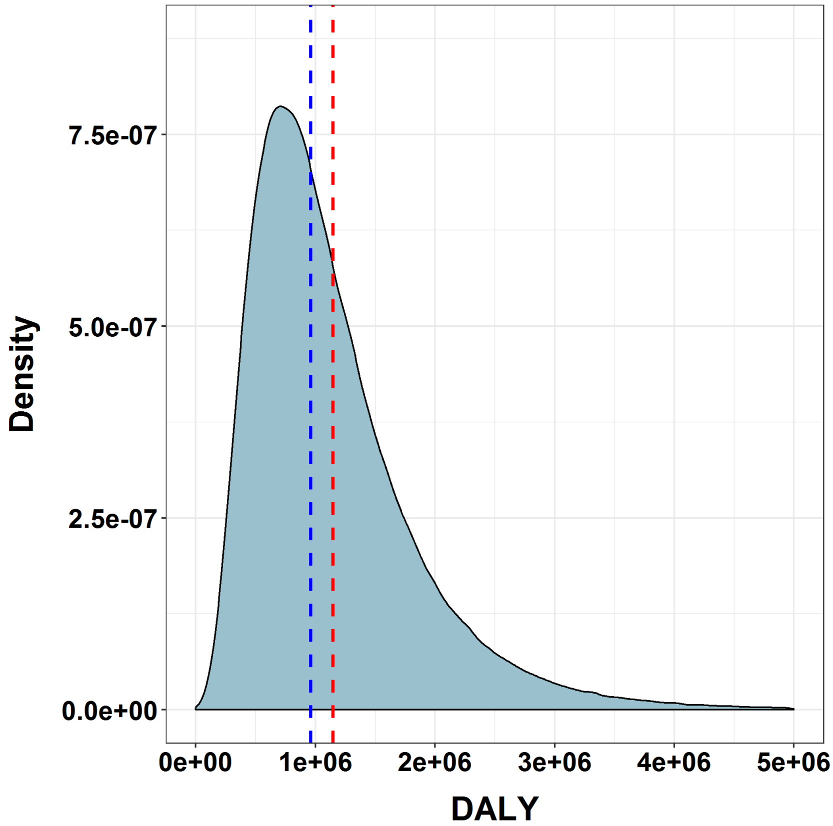

Figure 5-36: Density histogram of the DALYs due to years of life lost (PM2.5) in Germany, 2015. The red and the blue dash line represents the arithmetic mean (1.15×10^6) and geometric mean (9.61×10^5) respectively.

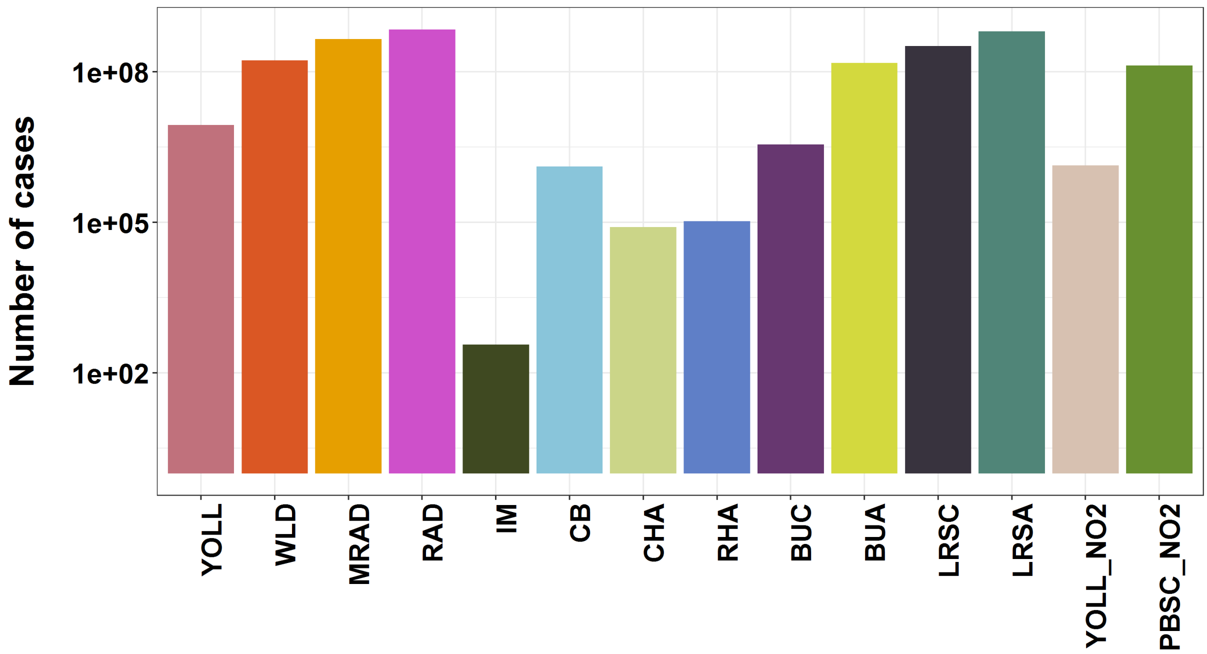

Figure 5-37: Health impacts for the total EU27+2 population in 2015. Details for abbreviations of the health endpoints are given in Table A-4.

Table 5-4: Health impacts, DALYs and damage costs for the total EU27+2 population in 2015. The information for the abbreviations of the health endpoints is given in Table A-4.

Endpoint	Cases		DALYs		Damage costs (€)	
	Mean	95% CI	Mean	95% CI	Mean	95% CI
YOLL_PM2.5	8.72×10^{6}	1.42×10^{6}-2.51×10^{7}	8.72×10^{6}	1.42×10^{6}-2.51×10^{7}	5.24×10^{11}	1.99×10^{9}-1.82×10^{12}
WLD	1.70×10^{8}	2.90×10^{7}-4.80×10^{8}	4.62×10^{4}	7.87×10^{3}-1.30×10^{5}	7.51×10^{10}	1.28×10^{10}-2.12×10^{11}
MRAD	4.46×10^{8}	7.53×10^{7}-1.27×10^{9}	8.56×10^{4}	1.44×10^{4}-2.43×10^{5}	2.54×10^{10}	4.29×10^{9}-7.21×10^{10}
RAD	6.98×10^{8}	1.19×10^{8}-1.97×10^{9}	1.89×10^{5}	3.22×10^{4}-5.34×10^{5}	1.35×10^{11}	2.31×10^{10}-3.82×10^{11}
IM	3.70×10^{2}	3.69×10^{1}-1.15×10^{3}	2.96×10^{4}	2.95×10^{3}-9.23×10^{4}	1.66×10^{9}	1.66×10^{8}-5.18×10^{9}
CB	1.29×10^{6}	8.09×10^{4}-4.25×10^{6}	1.28×10^{6}	8.01×10^{4}-4.21×10^{6}	8.50×10^{10}	5.20×10^{9}-2.86×10^{11}
CHA	8.03×10^{4}	1.15×10^{4}-2.42×10^{5}	2.17×10^{3}	3.09×10^{2}-6.53×10^{3}	2.40×10^{8}	2.82×10^{7}-7.76×10^{8}
RHA	1.06×10^{5}	1.72×10^{4}-3.04×10^{5}	2.58×10^{3}	4.19×10^{2}-7.39×10^{3}	3.18×10^{8}	4.19×10^{7}-9.78×10^{8}
BUC	3.57×10^{6}	0.00-3.15×10^{7}	2.15×10^{3}	0.00-1.90×10^{4}	2.86×10^{8}	0.00-2.52×10^{9}

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Table 5-4 – *Continued from previous page*

Endpoint	Cases		DALYs		Damage costs (€)	
	Mean	95% CI	Mean	95% CI	Mean	95% CI
BUA	1.51×10^8	0.00-6.73×10^8	9.08×10^4	0.00-4.06×10^5	1.20×10^{10}	0.00-5.39×10^{10}
LRSC	3.27×10^8	4.90×10^7-9.46×10^8	8.88×10^4	1.33×10^4-2.57×10^5	1.87×10^{10}	2.80×10^9-5.39×10^{10}
LRSA	6.38×10^8	4.13×10^7-2.10×10^9	1.73×10^5	1.12×10^4-5.70×10^5	3.64×10^{10}	2.36×10^9-1.20×10^{11}
YOLL_NO_2	1.36×10^6	0.00-7.81×10^6	1.36×10^6	0.00-7.81×10^6	8.18×10^{10}	0.00-5.32×10^{11}
PBSC	1.33×10^8	4.22×10^7-3.08×10^8	8.04×10^4	2.55×10^4-1.86×10^5	1.07×10^{10}	3.36×10^9-2.48×10^{10}
Sum	-	-	1.22×10^7	3.61×10^6-2.95×10^7	1.01×10^{12}	3.46×10^{11}-2.37×10^{12}

The DALYs and damage costs by pollutant and source are demonstrated in Figure 5-38. As displayed in the figure, the dominant contribution for both DALYs and damage costs were made by the outdoor air. The other obvious finding was the much heavier damages caused by PM2.5 compared to NO_2 (7.4 times for DALYs and 9.9 times for damage costs).

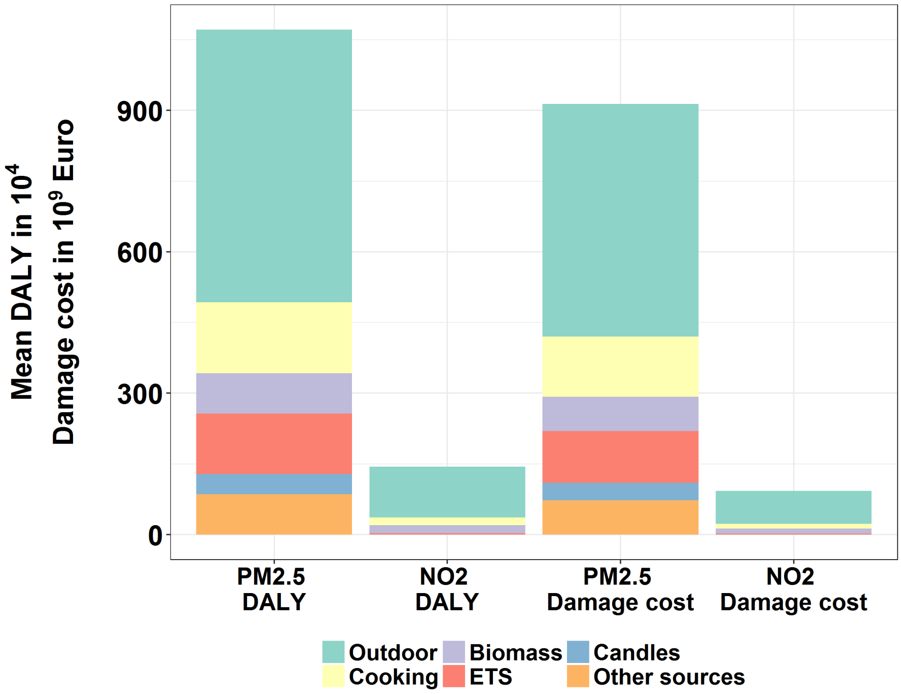

Figure 5-38: Total DALYs and damage costs by pollutant and source for the EU27+2 countries in 2015.

Table 5-5 compares the exposure, DALYs and damage costs simulated in this thesis for 2015 and the outcome for the 2020 Policy scenario generated by Gens (2012). Even though the source distribution is very different (details see Table 5-7), the total exposure, DALYs and damage costs show very high agreement between the two studies.

5.6 Lifelong exposure

The lifelong exposure to both pollutants for the eldest cohorts of population data (80-year-old) was simulated within the thesis. Large variance has been observed by the exposure among different countries. Figure 5-39 shows the comparison of the temporal course for the lifelong exposure to PM2.5 between the 80-year-old male from Poland and Sweden as an example. As shown by the figure, the exposure to PM2.5 experienced by the Polish male was apparently higher than the Swedish male for each life year. The average exposure over lifetime burdened by a Polish man was 42.22 (95% CI: 3.38-153.58) µg m^{-3}, while the value for a Swedish man was only 14.03 (95% CI: 1.33-54.14) µg m^{-3}. The variance among the countries has been found by the exposure to NO_2 as well (see Figure 5-40). Likewise, the average exposure for a Polish man reached 12.55 (95% CI: 0.75-39.06) µg m^{-3}, whereas for a Swedish man this value was only 8.10 (95% CI: 0.62-33.79) µg m^{-3}.

Table 5-5: Comparison of the mean PM2.5 exposure, DALYs and damage costs by source from this thesis for 2015 and the 2020 Policy scenario generated by Gens (2012). The exposure, DALYs and damage costs are given in µg m^{-3}, thousand DALYs and million €.

		Outdoor	Biomass	ETS	Other indoor sources	Total
This thesis	Exposure	10.9	1.5	2.5	5.2	20.1
	DALYs (10^3)/a	5,784	857	1,285	2,784	10,710
	Damage costs (10^6)/a	493,535	73,116	109,675	273,628	913,954
Gens (2012)	Exposure	4.5	2.2	7.1	3.5	17.2
	DALYs (10^3)/a	3,239	1,504	3,875	2,438	11,020
	Damage costs (10^6)/a	208,634	98,557	249,035	159,545	715,768

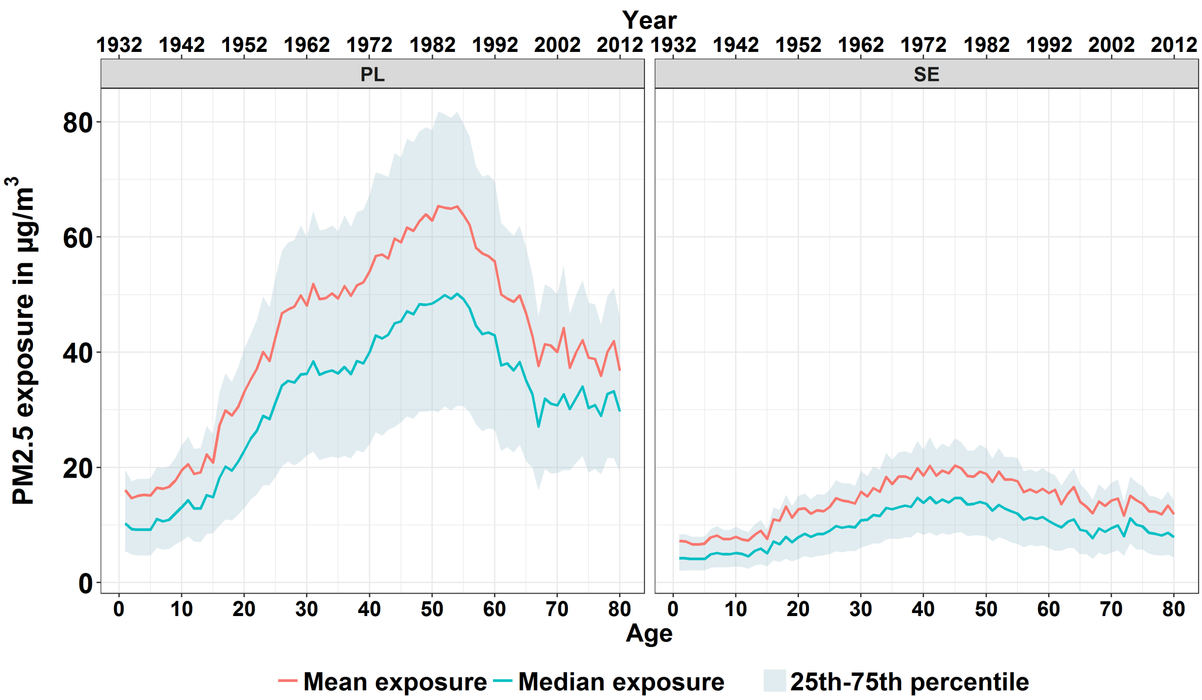

Figure 5-39: Temporal course of the lifelong exposure to PM2.5 for an 80-year-old male from Poland and Sweden. The red line, blue line and grey ribbon represent the mean, median and range of 25th to 75th percentile of the exposure level.

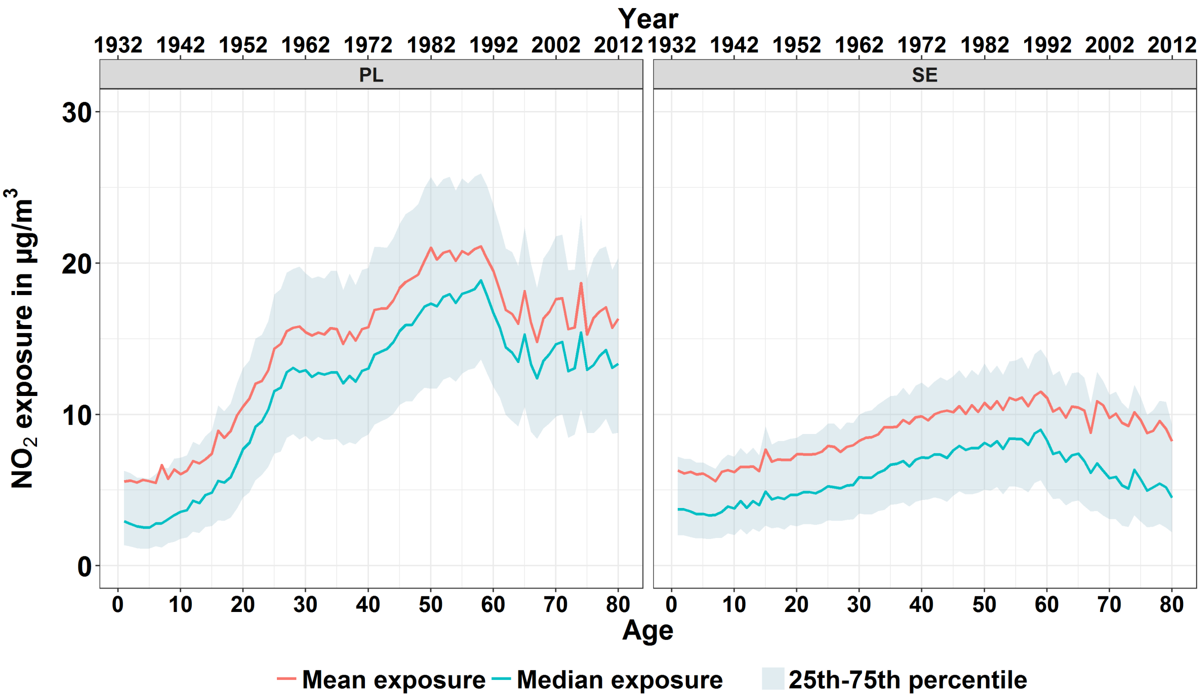

Figure 5-40: Temporal course of the lifelong exposure to NO_2 for an 80-year-old male from Poland and Sweden. The red line, blue line and grey ribbon represent the mean, median and range of 25^{th} to 75^{th} percentile of the exposure level.

For a European aged 80 the average exposure over lifetime to PM2.5 and NO_2 was 23.86 (95% CI: 2.95-81.86) and 13.49 (95% CI: 1.36-43.84) µg m^{-3}. Figure 5-41 and Figure 5-42 show the temporal courses of the lifelong exposure to both pollutants stratified by source for an 80-year-old European. As shown by the figures, the dominant contributions were made by the outdoor concentration (47% for PM2.5 and 67% for NO_2). A trend of continuous increase of exposure stemming from cooking can be observed, which can be explained by the elevated cooking time for the elder subgroup. The annual average exposure to PM2.5 and NO_2 resulted in YOLL of 3.53×10^{-2} (95% CI: 4.79×10^{-3}-1.17×10^{-1}) and 4.76×10^{-3} (95% CI: 0.00-3.03×10^{-2}) per year of exposure respectively. The health outcomes, DALYs and damage costs of other endpoints are listed in Table 5-6. Again a factor of 0.67 was applied to the health impacts brought by NO_2 (see Section 5.5).

It must be noted that the health impacts brought by the exposure to both pollutants should not be exactly the same at different phases of life. As pointed out by Bates (1995), Simoni et al. (2015) and Song et al. (2017), children, pregnant women and the elderly are more vulnerable to the exposures of toxic substances. Due to the data limitation, however, no difference has been made between the life stages. This should be improved in the future work if more information can be accessed.

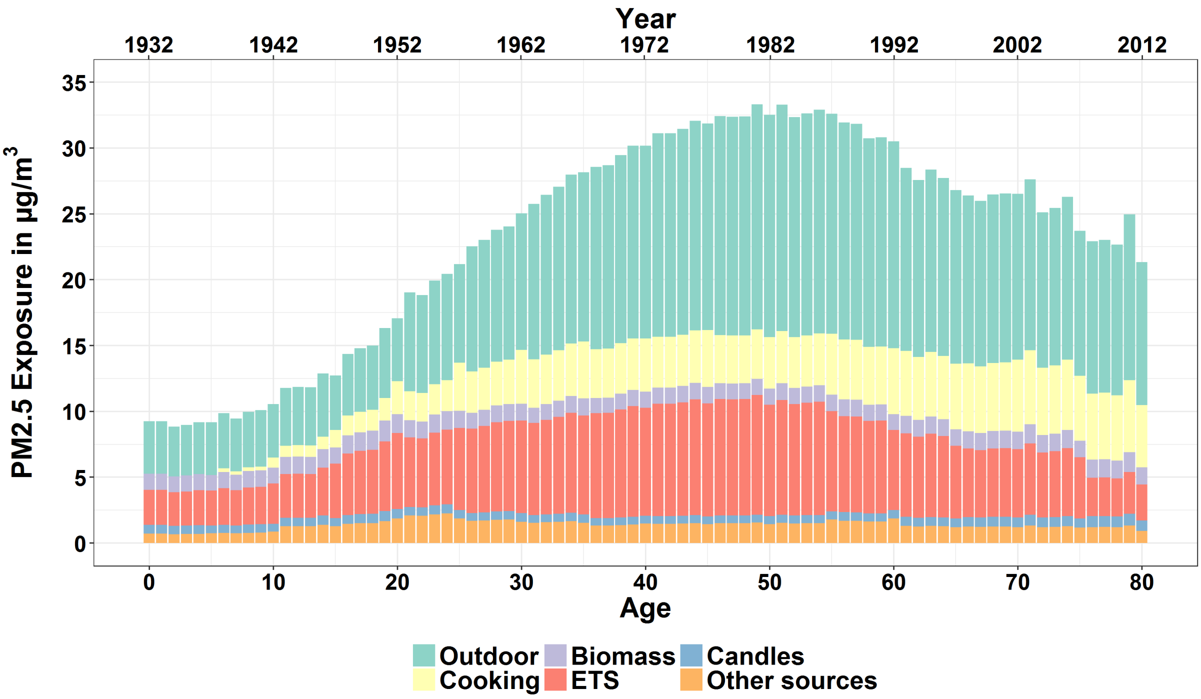

Figure 5-41: Temporal course of the average lifelong exposure to PM2.5 for an 80-year-old European by source, including infiltration from outdoors, biomass, candles, cooking, ETS and other sources.

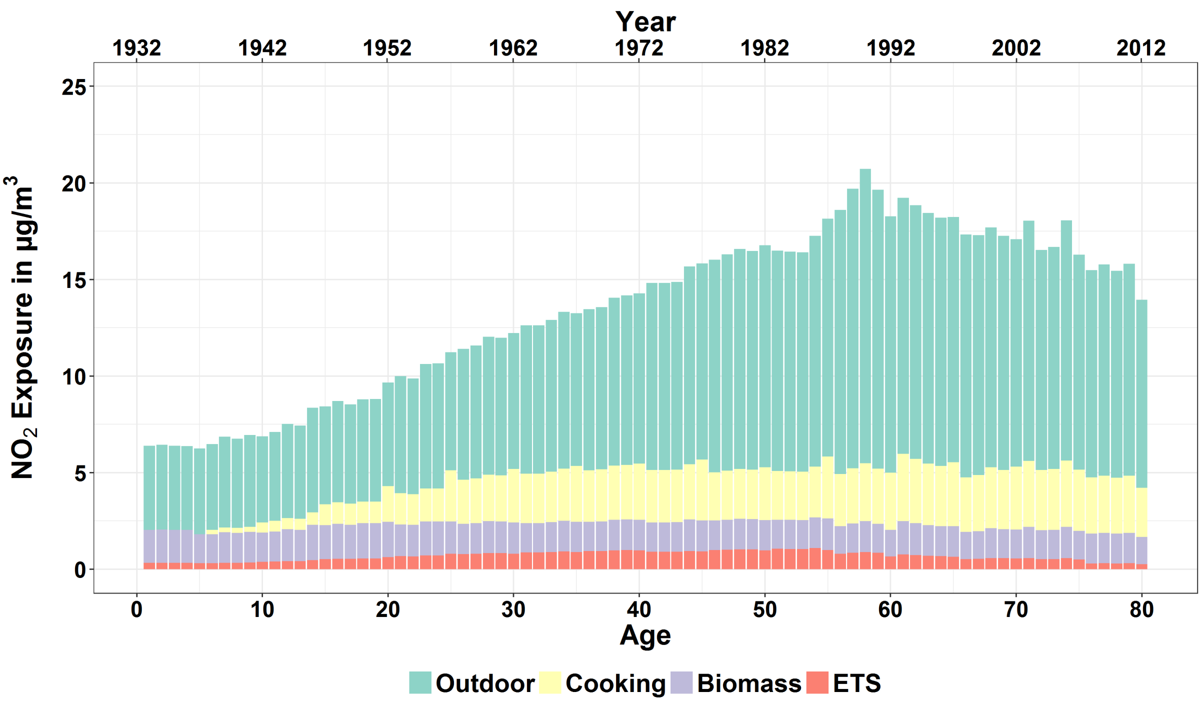

Figure 5-42: Temporal course of the average lifelong exposure to NO_2 for an 80-year-old European by source, including infiltration from outdoors, biomass, cooking and ETS.

Table 5-6: Health impacts, DALYS and damage costs of the individual annual average exposure to PM2.5 and NO_2. The information for the abbreviations of the health endpoints is presented in Table A-4.

Endpoint	Cases		DALYs		Damage costs (€)	
	Mean	95% CI	Mean	95% CI	Mean	95% CI
YOLL_PM2.5	3.53×10^{-2}	4.79×10^{-3}-1.17×10^{-1}	3.53×10^{-2}	4.79×10^{-3}-1.17×10^{-1}	2.12×10^{3}	1.63×10^{1}-8.27×10^{3}
WLD	6.75×10^{-1}	5.71×10^{-2}-2.38	1.83×10^{-4}	1.55×10^{-5}-6.44×10^{-4}	2.98×10^{2}	2.52×10^{1}-1.05×10^{3}
MRAD	1.95	1.89×10^{-1}-6.76	3.73×10^{-4}	3.62×10^{-5}-1.30×10^{-3}	1.11×10^{2}	1.07×10^{1}-3.85×10^{2}
RAD	3.03	2.96×10^{-1}-1.05×10^{1}	8.23×10^{-4}	8.04×10^{-5}-2.84×10^{-3}	5.89×10^{2}	5.75×10^{1}-2.03×10^{3}
IM	9.85×10^{-5}	0.00-4.33×10^{-4}	7.88×10^{-3}	0.00-3.46×10^{-2}	4.42×10^{2}	0.00-1.94×10^{3}
CB	4.12×10^{-3}	1.85×10^{-4}-1.58×10^{-2}	4.08×10^{-3}	1.83×10^{-4}-1.56×10^{-2}	2.72×10^{2}	1.18×10^{1}-1.05×10^{3}
CHA	1.84×10^{-4}	8.31×10^{-6}-7.01×10^{-4}	4.97×10^{-6}	2.24×10^{-7}-1.89×10^{-5}	5.50×10^{-1}	2.15×10^{-2}-2.20
RHA	2.42×10^{-4}	1.19×10^{-5}-8.83×10^{-4}	5.90×10^{-6}	2.90×10^{-7}-2.15×10^{-5}	7.25×10^{-1}	3.05×10^{-2}-2.79
BUC	3.97×10^{-2}	0.00-4.21×10^{-1}	2.39×10^{-5}	0.00-2.54×10^{-4}	3.18	0.00-3.38×10^{1}

Continued on next page

Table 5-6 – *Continued from previous page*

Endpoint	Cases		DALYs		Damage costs (€)	
	Mean	95% CI	Mean	95% CI	Mean	95% CI
BUA	4.93×10^{-1}	0.00-2.47	2.97×10^{-4}	0.00-1.49×10^{-3}	3.94×10^{1}	0.00-1.98×10^{2}
LRSC	3.64	0.00-1.47×10^{1}	9.88×10^{-4}	0.00-3.98×10^{-3}	2.08×10^{2}	0.00-8.35×10^{2}
LRSA	2.04	9.46×10^{-2}-7.82	5.55×10^{-4}	2.57×10^{-5}-2.12×10^{-3}	1.17×10^{2}	5.39-4.46×10^{2}
YOLL_NO_2	4.76×10^{-3}	0.00-3.03×10^{-2}	4.76×10^{-3}	0.00-3.03×10^{-2}	2.85×10^{2}	0.00-2.03×10^{3}
PBSC	1.27	1.25×10^{-1}-5.16	7.64×10^{-4}	7.52×10^{-5}-3.11×10^{-3}	1.01×10^{2}	9.97-4.13×10^{2}

5.7 Validation

As described in Section 3.7.3, the observing data from EXPOLIS are employed in this thesis to validate the parameters of mass-balance model for dwellings. The results of the model simulation are available as probability density functions since a probabilistic exposure model has been established (see Chapter 3). In this thesis, the estimation is deemed with "good performance" if the monitoring indoor concentration is within the range of 25th to 75th percentile of the predicting outcome. After multiple rounds of data collection, fusion and adjustment, the model achieved satisfactory outcome for both pollutants. The performance of the model for PM2.5 and NO_2 is displayed by Figure 5-43 and Figure 5-44 respectively. The blue dot represents the measurement that is within the corresponding 25th to 75th percentile of the modelling range, while the red dot is beyond that range. 70% (258 out of 370) and 65% (172 out of 265) of the dots for PM2.5 and NO_2 are within the acceptable range.

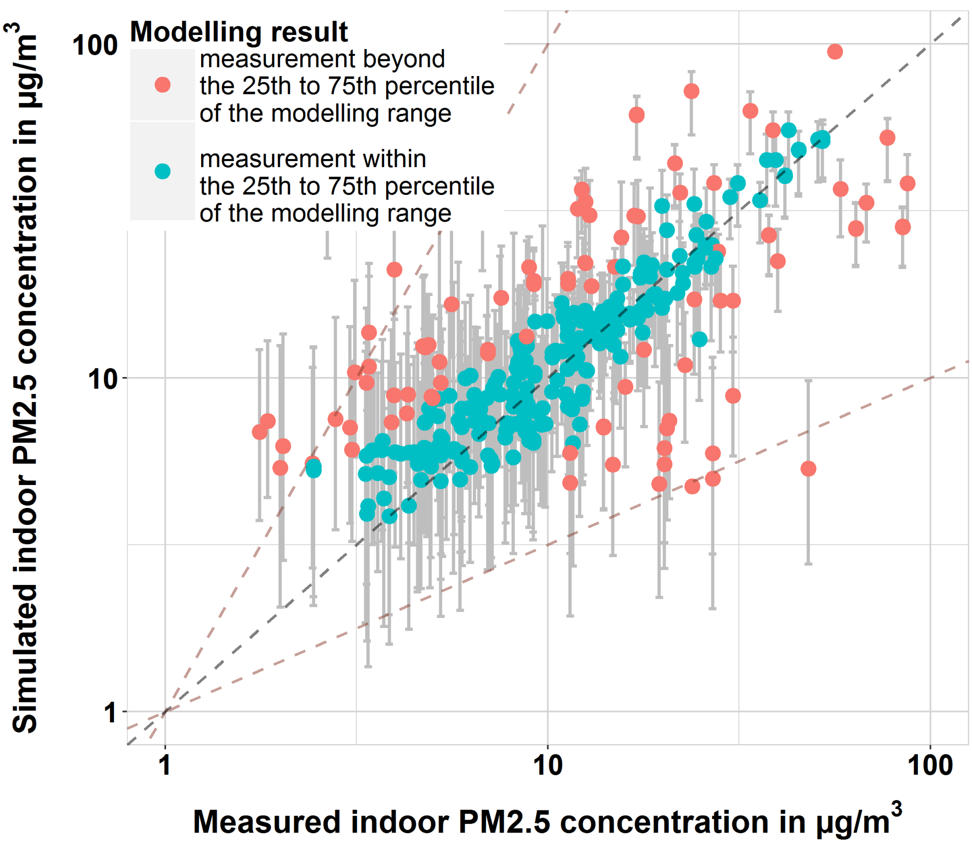

Figure 5-43: Validation for PM2.5. The blue dot represents the measurement that is within the corresponding 25th to 75th percentile of the modelling range, while the red dot is beyond that range. For PM2.5 70% of the dots (258 out of 370) are within the range.

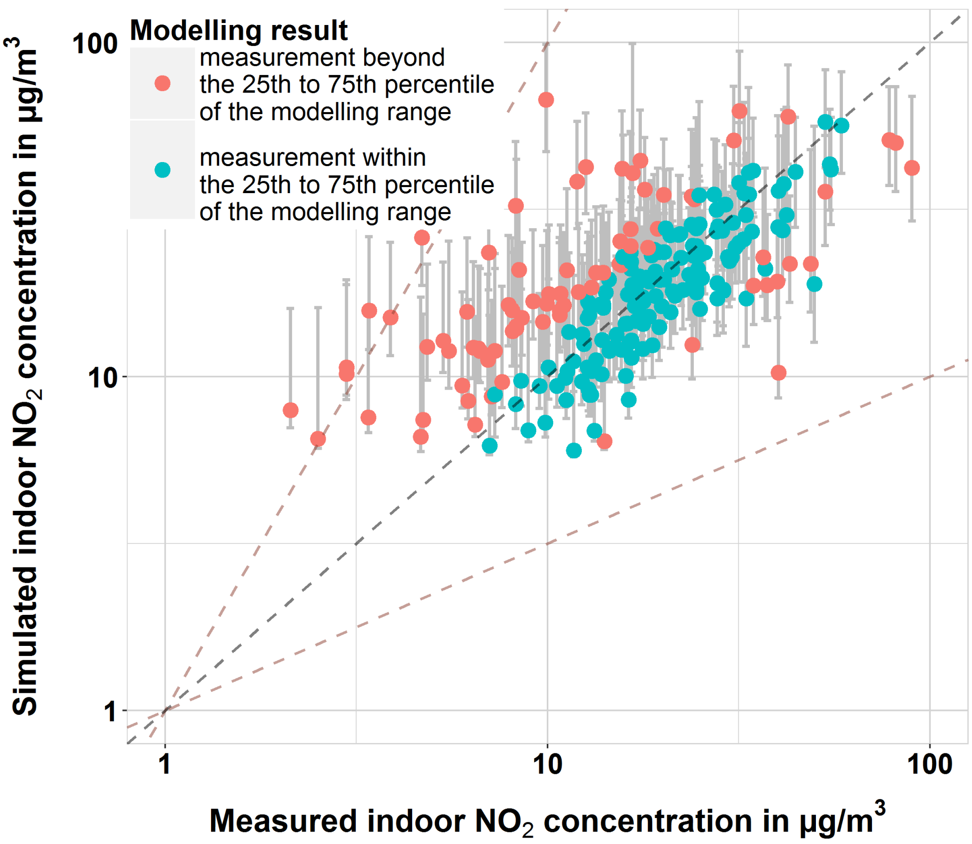

Figure 5-44: Validation for NO_2. The blue dot represents the measurement that is within the corresponding 25th to 75th percentile of the modelling range, while the red dot is beyond that range. For NO_2 65% of the dots (172 out of 265) are within the range.

5.8 Comparison with other studies

Except for the validation with monitoring data, the results of the exposure modelling developed in this thesis are compared with other studies as well.

5.8.1 PM2.5

Table 5-7 displays the comparison of the result for EU population in 2010 between this thesis and Gens (2012). The overall exposure simulated by the author is higher due to the notably heavier exposure stemming from the outdoor air. In Gens' thesis, the outdoor concentration fields from EMEP/MSC-W were directly adopted. By contrast, the author of this thesis employed the concentration map adjusted by EIONET interpolation method. This modification is essential since the CTM models usually underestimate the PM2.5 concentration (see Section 3.2).

As for the indoor sources, a noticeable gap is observed by the exposure from ETS. In this thesis, the implementation of regulations to moderate the ETS exposure is considered (see Section 4.3.1). Thus, the simulated outcome is less than 40% of the value from Gens.

Table 5-7: Comparison of the personal exposure to PM2.5 by source between this thesis and Gens (2012) for EU population in 2010 (µg m^{-3}).

	This thesis	Gens (2012)
Outdoor air	12.8	5.8
Only indoor air sources	9.2	12.8
ETS	2.8	7.6
Total	22.0	18.6

Without the consideration of smoking bans, the model developed in this thesis generates an average value of 7.0 µg m^{-3}. This value, however, coincides well with the result demonstrated by Gens (2012) as 7.6 µg m^{-3}, given the large uncertainties accompanied to both models. For other indoor sources, no evident difference has been found (6.4 compared to 5.2 µg m^{-3}).

The model results of PM2.5 are also compared with some measurement data of exposure from existing literatures. Brunekreef et al. (2005) monitored the personal, indoor and outdoor concentrations to PM2.5 for elderly people with cardiovascular disease living in Amsterdam, the Netherlands and in Helsinki, Finland, between 1998 and 1999. Table 5-8 shows the comparison of the results between this thesis and Brunekreef et al. (2005). It is noteworthy that the total exposure discussed here excluded the contribution from biomass and candles according to the setting of their study. The result of the model shows a relatively high agreement with the measurement data from Amsterdam, especially for the group exposed to ETS indoors. The simulated exposure is slightly heavier for group “All” and “without ETS”, which is mainly due to the higher outdoor concentrations modelled compared to the measurement values. As for the samples from Helsinki, an overestimation has been observed by the model result for the group “with ETS”. This can be attributed to the extremely low cigarette consumption of the Finnish participants (2 cigarettes per day compared to the national average of 16.5).

Molnár et al. (2005) investigated the personal exposures during the winter of 2003 in a Swedish town where wood burning for domestic space heating was common. Samples from the wood-burning households and a reference group, who lived in homes with electrical heating or heat pumps, were selected. Both groups of subjects were non-smokers and they were not exposed to ETS during the sampling period. Table 5-9 lists the comparison between the model result from this thesis and the measurement data. The model outcome reveals a high agreement with the monitoring data recorded for both groups.

5.8.2 NO_2

For NO_2, the model results from Dimitroulopoulou et al. (2001) are taken as a reference. They developed a dynamic multi-compartment computer model to determine the indoor NO_2 concentrations as a function of outdoor concentrations, indoor emission rates and

Table 5-8: Comparison of the personal exposure to PM2.5 between this thesis and Brunekreef et al. (2005) for elderly people in Amsterdam and Helsinki, 1999 (µg m^{-3}).

		Amsterdam			Helsinki		
		25%	Median	75%	25%	Median	75%
Brunekreef et al. (2005)	All	8.9	13.6	23.1	6.9	9.2	13.0
	without ETS	8.2	11.7	16.9	6.9	9.1	12.8
	with ETS	19.5	34.8	67.4	7.3	13.1	18.0
This thesis	All	13.0	17.0	22.1	5.7	7.9	11.9
	without ETS	10.7	13.8	17.5	4.9	6.9	10.1
	with ETS	22.5	34.3	55.0	15.1	27.9	54.2

Table 5-9: Comparison of the personal exposure to PM2.5 between this thesis and Molnár et al. (2005) for Swedish samples in 2003 (µg m^{-3}).

	This thesis			Molnár et al. (2005)		
	Min	Median	Max	Min	Median	Max
Wood burners	4.4	19.8	79.2	5.3	18.0	59
Reference group	2.4	11.4	56.3	5.8	12.0	46.0

building characteristics. This study has been conducted from 1993 to 1996 for a representative homemaker, schoolchild and office worker in Lullington Heath, Birmingham, Leeds and London respectively. Table 5-10 displays the reported ranges of the mean values for the British study, as well as the mean and 95% CI of exposure simulated in this thesis. Following the setting for their model, only cooking as the indoor source has been involved. As shown by the table, the exposure ranges generated by this thesis fit quite well with those from Dimitroulopoulou et al. (2001). However, it is difficult to make a conclusion for the mean values since no sample weight was given by their model.

Table 5-10: Comparison of the personal exposure to NO_2 by source between this thesis and Dimitroulopoulou et al. (2001) for the UK samples in 1995. The reported ranges of the mean values for the British study, as well as the mean and 95% CI of exposure in this thesis are presented (µg m^{-3}).

	This thesis	Dimitroulopoulou et al. (2001)
Outdoor air	18.2 (8.0-32.4)	9.4-41.4
Only indoor air sources	2.2 (0.0-21.7)	0.0-32.3
Total	20.3 (8.6-42.7)	9.4-50.76

For the United Kingdom, another study was conducted by Kornartit et al. (2010) to measure the personal exposure during the winter of 2000 and summer of 2001 respectively. The NO_2 exposure level was reported as 26.3 (±2.8) μg m^{-3} for summer and 17.8 (±4.5) μg m^{-3} for winter. The annual average exposure simulated in this thesis is 21.4 (95% CI: 8.9-47.5) μg m^{-3}. The monitoring data accord quite well with the modelling values considering the large uncertainties attached.

Personal NO_2 exposure measurements were achieved by Piechocki-Minguy et al. (2006) in greater Lille, France in 2001 (Table 5-11). For the corresponding year, the average exposure simulated by this thesis was 21.7 (95% CI: 6.1-44.4). The modelling results match well with the values for summer. However, for winter the simulated data are lower, especially for working days. As pointed out by Piechocki-Minguy et al. (2006), the use of gas stoves and gas heating in winter was a driving factor for the sharp increase of NO_2 personal exposure. As stated in Section 4.3.2.1, the impact of gas stove on NO_2 exposure has been taken into account. Due to the limitation of data, nevertheless, emission from gas heating has not been incorporated in this thesis, which should be improved with respect to further research.

Table 5-11: Comparison of the personal exposure to NO_2 between this thesis and Piechocki-Minguy et al. (2006) for the French samples in 2001 (μg m^{-3}).

	This thesis		Piechocki-Minguy et al. (2006)	
	Mean	95% CI	Mean	Range
Summer working day	21.7	6.1-44.4	22	12-32
Summer weekend			17	8-50
Winter working day			38	24-53
Winter weekend			29	14-61

6 Summary, conclusions and outlook

6.1 Summary and conclusions

A wide range of adverse impacts of air pollutants have been documented worldwide. In spite of actions that have been taken within the last few decades, Europe is still facing the challenges of air pollution, especially for PM2.5 and NO_2. According to WHO (2013b), more than 80% of the population in EU live in urban areas where the PM level is over the recommended value of the WHO Air Quality Guidelines. As for NO_2, the annual limit value is widely exceeded across Europe as well (EEA 2018).

Up to now most of the studies have emphasised the use of concentration response functions (CRFs) to evaluate the health outcomes due to air pollutants. The CRFs are derived from epidemiological studies and correlate the health impacts with the pollutant concentration taken from the outdoor monitoring stations. These stations measure the background concentration with hourly to annual resolution. However, these measurement data could not reflect the real concentration that an individual is exposed to, since people usually spend most of the time indoors. Additionally, people are impacted by the sources generated from the indoor environment.

Moreover, the existing epidemiological studies are constrained to a certain small period (e.g. one year). However, it should be stressed that the chronic health outcomes are caused by the accumulated exposure through the whole lifetime.

Last but not least, the environmental inequalities have been universally documented (Deguen and Zmirou-Navier 2010; Gelotte 2017; Nguyen and Marshall 2018). This means, some vulnerable subgroups (e.g. people with low income) are likely to be burdened with heavier exposure. Thus, the influence of socio-economic status on exposure to PM2.5 and NO_2 should be addressed.

Considering the problems stated above, this thesis developed a methodology to simulate the long-term exposure to PM2.5 and NO_2 of European population subgroups that were characterised by age, gender, region, socio-economic status and behavioural habits. The exposure is defined here as the average concentration of pollutant in the air inhaled by a person. To estimate the lifelong exposure, a probabilistic model for simulating the concentration of pollutants in each micro-environment, including outdoor, home, office, school and transportation has been developed. For the outdoor environment, large efforts were made to yield the concentration fields since the 1930s. For the indoor micro-environment, a mass-balance model was applied. The contribution of indoor sources, such as cooking, smoking, wood burning, candle/incense and other sources was evaluated. For transportation, a micro-environment factor was employed to simulate the concentration while travelling.

Several subsequent applications were made with this model: the combination with a life course trajectory model to determine the lifelong exposure at the individual level, the health impact assessment with adjusted exposure response functions (ERFs) and the aggregation and monetary valuation with DALY factors and monetary values.

The methodology developed in this thesis enables the simulation of exposure of individuals and population subgroups to both PM2.5 and NO_2 at European level. Moreover, this method greatly extends the temporal scale of the exposure modelling even back to the 1930s.

The newly developed methodology supports the evaluation of exposure for different subgroups and explains the influence of socio-economic factors and behavioural habits on personal exposure.

The exposure model is combined with a life course trajectory model to simulate the lifelong exposure and health impacts at the individual level. This is the first time that the method of sequence analysis has been applied in the field of environmental science.

The exposure simulated with Monte Carlo sampling shows a log-normal distribution. Similar results have been documented by Moschandreas and Saksena (2002) and Burke (2005). The geometric mean exposure to PM2.5 and NO_2 for Europeans in 2015 was 17.0 and 14.0 µg m^{-3} respectively with a geometric standard deviation of 1.7 and 1.6. For PM2.5 the temporal course of the annual average exposure at European level showed a steady trend of increase since the 1950s from 19.0 (95% CI: 3.3-55.7) µg m^{-3} to a maximum of 37.2 (95% CI: 9.2-113.8) µg m^{-3} in the 1980s. The exposure turned to decline gradually afterwards until 2015 to 20.1 (95% CI: 5.8-51.2) µg m^{-3}. A similar trend was observed for the NO_2 exposure. The total exposure increased since the 1950s from 10.4 (95% CI: 0.9-36.8) µg m^{-3} to a maximum in 1990 of 21.4 (95% CI: 6.3-51.8) µg m^{-3}. Afterwards the exposure to NO_2 kept on decreasing to 15.5 (95% CI: 4.8-36.8) µg m^{-3} in 2015.

Generally, the outdoor air was a key factor in the overall exposure, especially for the peak years (55% for PM2.5 and 78% for NO_2). This is confirmed by the strong correlation between the outdoor concentration and the total exposure for both pollutants (r=0.78 for PM2.5 and r=0.81 for NO_2).

For both pollutants, the drop of exposure was strongly related to the decline of the ambient concentration, which was stemming from the implementation of a series of emission reduction policies for PM2.5 and its precursors (NO_x, NH_3 and SO_2) since the 1970s. The decline in emissions has been confirmed by the EDGAR emission data (Crippa et al. 2018).

Regarding the indoor sources of PM2.5, smoking had a considerable impact on the overall exposure for all time periods, especially for the peak year 1985 with 8.8 µg m^{-3}. The exposure to ETS also revealed a similar temporal course as the total exposure. This value dropped dramatically to only 2.5 µg m^{-3} in 2015 thanks to the measures taken for tobacco control, including smoking bans and high taxation of tobacco products. The contribution of ETS to the total NO_2 exposure is much less evident.

Cooking (especially frying) is another essential source for both pollutants, especially for the earlier time periods. Yet, the exposure kept on decreasing owing to the drop of time spent on cooking. Moreover, wood burning was another notable factor for the total exposure, especially for NO_2. The contributions from other indoor sources (candles/incense sticks and resuspension) were relatively low.

Not surprisingly, a large variance of the exposure to both pollutants among different countries has been found. Generally Eastern European countries ranked in the top regarding the exposure level for both pollutants, which can be explained by the higher background concentration, higher prevalence of indoor smoking, longer time spent on cooking and smaller dwelling size compared to the countries from other regions. In contrast, the inhabitants of Northern European countries were generally least affected by both pollutants.

It is important to note that the exposure to NO_2 was relatively high in Italy and some Northwestern countries, including Germany and United Kingdom. This was due to the fact that the overall background concentrations in these countries were relatively high. Moreover, most of the people in these countries were likely to reside in urban or suburban areas, where the outdoor concentration level was remarkably elevated as a consequence of the urban transportation.

The influences of socio-demographic factors, including gender, household income level, degree of urbanisation of living area, civil status, employment status and education level on exposure to both pollutants have been examined at European level as well. With respect to gender, apparent differences have been observed by the exposure due to ETS and cooking. Generally, the exposure stemming from cooking for women was at least as twice high as for men. In earlier time periods, the ratio between the two groups reached a factor of five. In contrast, the exposure due to ETS for men was heavier than for women as a consequence of higher prevalence of smokers and daily cigarette consumption.

Concerning the income level, a clear trend can be observed for both pollutants. The highest exposure levels were found at the population with the lowest income level. This conclusion was mainly caused by the smaller dwelling size of the low-income group. Even though the smaller room size is irrelevant to the exposure from infiltration, it hinders the dilution of the indoor sources. Besides, the subgroup with lower income tended to spend more time on cooking. This factor is also responsible for the increase of exposure for low-income people.

People living in cities experienced higher level of exposure to both pollutants than people in rural areas. As discussed above, the average background concentration was higher in cities owing to urban traffic. Moreover, the smaller average dwelling size of the urban citizens is another driving factor for the elevation of the exposure due to indoor sources.

The comparison between married and single persons showed different trends. Regarding PM2.5, the total exposure was higher for singles for most of the time periods. Even though couples had higher exposure due to cooking as a result of longer cooking time, the exposure from other indoor sources was lower owing to the larger dwelling size. However, the exposure of couples to NO_2 was constantly higher than that of singles due to the dominant influence of cooking.

For both pollutants the exposure for the unemployed was larger than for the employed. First of all, the employed were likely to spend less time on cooking. Moreover, the unemployed were the group showing the highest prevalence of smokers. Last but not least, smaller room sizes increased the total concentration caused by the indoor sources.

As for the education level, it is shown that the well-educated people were less likely to be influenced by the exposure to both pollutants. The reasons behind were the lower cooking time, lower smoker prevalence and larger room size.

The influence of the ventilation system is discussed in this thesis as well. In this thesis, buildings were categorised into four types as “natural”, “retrofitted”, “mechanical” and “AHU”. The lowest level of exposure to both pollutants has been observed by “AHU” buildings. This type of building was insulated and equipped with an air handling unit, which was able to recirculate the air through the filters. The installation of AHU reduced the exposure originated from both outdoor and indoor sources. For “mechanical” buildings, the air exchange rates are often reported as “under required values” despite the operation of extract or supply fans. This increased the exposure due to indoor sources, however, the installed filters effectively minimised the influence of infiltration. Hence, the introduction of a well-functioning mechanical ventilation with filters and heat recovery system is recommended. The outcome became complicated for “retrofitted” buildings, which were usually insulated but without additional mechanical ventilation or filtering system. The insulation raised the exposure due to indoor sources. In spite of the short-term problem raised, the insulation is still deemed as one of the most advantageous measures to reduce the overall energy consumption and weaken the influence of the outdoor air pollutant in a long-term.

The author calculated the health impacts, DALYs and damage costs due to the PM2.5 and NO_2 exposure for the total population in EU27+2 countries for 2015. The total DALYs reached 1.22×10^7 (95% CI: 3.61×10^6-2.95×10^7), and the total damage costs amounted to 1.01×10^{12} (95% CI: 3.46×10^{11}-2.37×10^{12}) €, within which 88.1% of the DALYs and 90.8% of the damage costs were stemming from PM2.5. Within the different health endpoints, the years of life lost (YOLL) due to PM2.5 were responsible for 71.8% and 52.0% of the total DALYs and damage costs.

With the support of the life course trajectory model, an assessment of the lifelong exposure to both pollutants has also been carried out. Large difference have been observed among different countries. For example, the average PM2.5 exposure over lifetime burdened by an 80-year-old Polish man was 42.22 (95% CI: 3.38-153.58) $\mu g\ m^{-3}$, while for a Swedish man of the same age the exposure amounted to only 14.03 (95% CI: 1.33-54.14) $\mu g\ m^{-3}$. With respect to NO_2, the exposure of this Polish man reached 12.55 (95% CI: 0.75-39.06) $\mu g\ m^{-3}$, whereas for the Swedish man this value was only 8.10 (95% CI: 0.62-33.79) $\mu g\ m^{-3}$. Generally for a European aged 80, the average exposure over lifetime to PM2.5 and NO_2 was 23.86 (95% CI: 2.95-81.86) and 13.49 (95% CI: 1.36-43.84) $\mu g\ m^{-3}$ respectively. The exposure to both pollutants led to YOLL of 3.53×10^{-2} (95% CI: 4.79×10^{-3}-1.17×10^{-1}) and 4.76×10^{-3} (95% CI: 0.00-3.03×10^{-2}) per year of exposure, i.e. an average loss of life expectancy of 0.42 and 0.06 month respectively per life year exposed.

To improve the exposure model for the indoor environment, measurement data from the EXPOLIS study have been applied for validation. For PM2.5 and NO_2, 70% and 65% of the measurement samples were falling within the range of the 25^{th} to 75^{th} percentile of the simulated outcome.

The simulated results of the exposure modelling have been compared with the data from existing models. Even though the source stratification showed a large difference for PM2.5, the total exposure was very similar to the result generated by Gens (2012). The comparison with the model for NO_2 from Dimitroulopoulou et al. (2001), as well as the measurement data from Brunekreef et al. (2005), Molnár et al. (2005), Piechocki-Minguy et al. (2006) and Kornartit et al. (2010) also showed a relatively high agreement.

The methodology framework developed within the scope of this thesis incorporated not only the influence of outdoor air, but also that of the indoor sources (e.g. cooking and smoking) on the overall exposure to PM2.5 and NO_2. In the meanwhile, it took the impacts of reduction measures (e.g. smoking bans and cooking hoods) into account. Last but not least, the inclusion of lifelong exposure modelling brought new ideas to the assessment of health impacts and environmental policies.

6.2 Outlook

Various future improvements of the methodology in this study are possible. Also a great potential is found in reducing the data gaps.

First of all, it would be advantageous if **more micro-environments**, such as restaurants, bars and stores could be additionally considered. According to Eurobarometer (2007), still over 70% of the European population reported exposure to tobacco smoke in restaurants, pubs, or bars in 2006. More data for these micro-environments would be necessary. Moreover, further research is needed to distinguish different **working places** in order to analyse the occupational exposure to both pollutants.

Some of the existing studies have argued that concentrations vary significantly between **modes of transport**. Thus, more work will need to be done to determine the micro-environment factor based on different transportation modes.

The **ventilation system** is one of the most important factors affecting the indoor pollutant concentration. In this thesis, only four categories, i.e. natural ventilation, natural ventilation with insulation, mechanical ventilation with or without air recirculation are covered due to the limit of data. In real life, the classifications of ventilation systems are more complicated. It is worthwhile to include more subcategories in the future work if more data can be attained.

It is worthwhile to increase the **temporal resolution** of the model input data. In this thesis, only the annual average of concentration fields was simulated. It could be helpful to have the pollutant concentration at seasonal, monthly, or even daily level. The temporal resolution of the parameters for the mass-balance model can be potentially improved as well. According to the data from Hänninen et al. (2011), a strong seasonal variance has been observed by the air exchange rate and the infiltration factor for PM2.5.

For the exposure due to **biomass**, the proportion of people exposed to wood burning was assumed to stay constant through all the time periods in this thesis. This could be modified if data for earlier years can be achieved. Furthermore, the impacts of different wood stove types and utilisation modes remain to be elucidated.

The issue of **other activities** is also an intriguing one which could be explored in the future research if information for more activities can be included.

Moreover, a better understanding of **gas heatings** needs to be developed with respect to the NO_2 exposure originated from the indoor sources, especially for the winter time.

More work will need to be done to distinguish the model input parameters for population with different **socio-economic status**. For example, people with higher socio-economic status are likely to live in dwellings equipped with advanced filtering systems or cooking hoods.

With respect to the **lifelong exposure**, it would be helpful if the life course trajectories for other socio-economic variables, e.g. income level, civil status, could be covered. Also the simulation of the lifelong exposure is based on the assumption that the person stays in the same region throughout their lifetime. It would be beneficial if information concerning relocations would become available.

Last but not least, it is found that children, pregnant women and the elderly are more vulnerable to the exposures of toxic substances. This should be improved in the future work if different weights can be assigned to the exposure at different **life stages** when estimating the health impacts caused by the air pollutants.

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A Appendix

Table A-1: MTUS 69 activity and 25 activity typologies (Fisher and Gershuny 2016).

69 categories	25 categories		
	Activity code	Description	Micro-environment
adult care	adult care	look after adults needing help or care	home indoor
cleaning	clean	cleaning, laundry, regular housework	home indoor
laundry, ironing, clothing repair			
other domestic work			
travel to or from work	commute	travel to/from work or education	travel/commute
education-related travel			
play computer games	computer	e-mail, web, program, computer games	home indoor
send e-mail, surf Internet, programming, computing			
home/vehicle maintenance/improvement, collect fuel	maintain	maintain home/vehicle, including collect fuel	home indoor
meals at work or school	eat/drink	meals or snacks	home indoor
meals or snacks in other places			
regular schooling, education	education	schooling, education, homework	school indoor
homework			
leisure course or other education or training			

Continued on next page

Table A-1 – *Continued from previous page*

69 categories	25 categories		
	Activity code	Description	Micro-environment
food preparation, cooking	food preparation	food preparation, cook, wash/put away dishes	home indoor
set table, wash/put away dishes			
gardening/forage (pick mushrooms), hunt/fish	garden	gardening/pick mushrooms	outdoor
general out-of-home leisure	go out	cinema/theatre, sport match, away from home leisure	other indoor
attend sporting event			
cinema, theatre, opera, concert			
other public event, venue			
restaurant, café, bar, pub			
party, reception, social event, gambling			
imputed time away from home			
other out-of-doors recreation			
teach child a skill, help with homework	intellectual child care	play/sports with, read/talk to child, help with homework	home indoor
read to, talk or play with child			
receive or visit friends	leisure	other free time leisure	home indoor
conversation (in person, phone)			
games (social or solitary), other in-home social			
general indoor leisure			

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Table A-1 – *Continued from previous page*

69 categories	25 categories		
	Activity code	Description	Micro-environment
artistic or musical activity	leisure	other free time leisure	home indoor
written correspondence			
knit, crafts or hobbies			
relax, think, do nothing			
paid work-main job (not at home)	paid work	paid work and related activities	work indoor
paid work at home (main, second or other job)			
second or other job not at home			
unpaid work to generate household income			
travel as a part of work			
work breaks			
other time at workplace			
look for work			
pet care (other than walk dog)	pet care	pet care (including walk dogs)	outdoor
walk dogs			
physical or medical child care	physical child care	physical, medical, supervisory, routine child care	home indoor
supervise, accompany, other child care			
read	read	read	home indoor
imputed personal or household care	self care	wash, dress, care for self	home indoor

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Table A-1 – *Continued from previous page*

69 categories	25 categories		
	Activity code	Description	Micro-environment
wash, dress, care for self	self care	wash, dress, care for self	home indoor
purchase goods	shopping	purchase goods, consume services	other indoor
consume personal care services			
consume other services			
sleep and naps	sleep	sleep and naps	home indoor
imputed sleep			
general sport or exercise	sport/exercise	sport or exercise	outdoor
walking			
cycling			
no activity, recorded travel mode or change of location	travel	other travel	travel/commute
travel for voluntary/civic/religious activity			
child/adult care-related travel			
travel for shopping, personal or household care			
travelling for other purposes			
listen to music, iPod, CD, audio book	TV/radio	watch television, listen to radio	home indoor
listen to radio			
watch TV, DVD, including web streamed content			

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Table A-1 – *Continued from previous page*

69 categories	25 categories		
	Activity code	Description	Micro-environment
worship and religious activity	religion	worship, religion and prayer	other indoor
voluntary work, civic or organizational activity	voluntary activity	voluntary, civic, organisational activity	other indoor

Table A-2: Categories for employment status and education level (Schieberle et al. 2017).

Category	Description
1	Employee working full-time
2	Employee working part-time
3	Fulfilling domestic tasks and care responsibilities
4	In compulsory military community or service
5	In retirement or in early retirement or has given up business
6	Infancy
7	Lower-secondary or second stage of basic education (ISCED 2)
8	No education (during adolescence)
9	Other inactive person
10	Permanently disabled or/and unfit to work
11	Post-secondary non-tertiary education
12	Pre-primary education (ISCED 0)
13	Primary education or first stage of basic education (ISCED 1)
14	Pupil, student, further training, unpaid work experience
15	Self-employed working full-time (including family worker)
16	Self-employed working part-time (including family worker)
17	Unemployed
18	Upper-secondary education (ISCED 3)

Table A-3: Values of the air exchange rate for schools by country (h^{-1}) (Csobod et al. 2014).

Country	EUROSTAT country code	Min	Mode	Max
Austria	AT	0.22	0.36	0.67
Belgium	BE	0.58	0.64	0.72
Bulgaria	BG	0.13	0.21	0.27
Cyprus	CY	0.90	2.10	4.30
Czech Republic	CZ	1.01	1.37	2.47
Denmark	DK	0.22	0.36	0.67
Estonia	EE	0.22	0.30	0.53
Finland	FI	0.22	0.36	0.67
France	FR	0.10	0.26	0.60
Germany	DE	0.03	0.25	0.86
Greece	EL	0.06	0.32	1.59
Hungary	HU	0.14	0.38	0.64
Ireland	IE	0.22	0.36	0.67
Italy	IT	0.07	0.18	0.82
Latvia	LV	0.34	0.73	1.44
Lithuania	LT	0.93	1.40	2.81
Luxembourg	LU	0.22	0.36	0.67
Malta	MT	0.12	0.60	0.88
Netherlands	NL	0.22	0.36	0.67
Norway	NO	0.22	0.36	0.67
Poland	PL	0.10	0.39	0.62
Portugal	PT	0.05	0.17	0.38
Romania	RO	0.03	0.08	0.12
Slovakia	SK	0.22	0.41	1.84
Slovenia	SI	0.34	0.73	1.44
Spain	ES	0.22	0.60	1.37
Sweden	SE	0.22	0.36	0.67
Switzerland	CH	0.22	0.36	0.67
United Kingdom	UK	0.18	0.28	0.49

Table A-4: Overview of the impact functions.

Health impact	Pollutant	Age group	Risk fraction	Impact function
Bronchodilator usage adults (BUA)	PM2.5 (>2.4 μg m^{-3})	20+	0.102	14,600 (± 14,150) additional days of bronchodilator usage per 1μg m^{-3} increase in PM2.5 per 100,000 adults aged 20 and older with well-established asthma, per year
Bronchodilator usage children (BUC)	PM2.5 (>2.4 μg m^{-3})	5-14	0.144	2,200 (± 6,025) additional days of bronchodilator usage per 1μg m^{-3} increase in PM2.5 per 100,000 children aged 5-14 years meeting the PEACE study criteria. Per year.
Cardiac hospital admissions (CHA)	PM2.5 (>2.4 μg m^{-3})	All	1	0.65 (± 0.16) additional emergency cardiac hospital admissions per 1μg m^{-3} increase in PM2.5 per 100,000 people (all ages). Per year.
New cases of chronic bronchitis (CB)	PM2.5 (>2.4 μg m^{-3})	18+	0.9	14 (± 5.95) new cases of chronic bronchitis per 1μg m^{-3} increase in PM2.5 per 100,000 at-risk adults aged 18 and older. Per year.
Infant mortality (IM)	PM2.5 (>2.4 μg m^{-3})	1 month to 1 year	1	0.87 (± 0.29) additional infant deaths per 1μg m^{-3} increase in PM2.5 per 100,000 live births. Per year.

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Table A-4 – *Continued from previous page*

Health impact	Pollutant	Age group	Risk fraction	Impact function
Lower respiratory symptoms adults (LRSA)	PM2.5 (>2.4 μg m^{-3})	18+	0.3	20,800 (± 8,750) additional lower respiratory symptom days per 1μg m^{-3} increase in PM2.5 per 100,000 adults aged 18 and older with chronic respiratory symptoms. Per year.
Lower respiratory symptoms children (LRSC)	PM2.5 (>2.4 μg m^{-3})	5-14	1	29,000 (± 7,400) additional lower respiratory symptom days per 1μg m^{-3} increase in PM2.5 per 100,000 children aged 5-14. Per year.
Respiratory hospital admissions (RHA)	PM2.5 (>2.4 μg m^{-3})	All	1	0.86 (± 0.0625) additional emergency respiratory hospital admissions per 1μg m^{-3} increase in PM2.5 per 100,000 people (all ages). Per year.
Minor restricted activity days (MRAD)	PM2.5 (>2.4 μg m^{-3})	18-64	1	5,770 (± 545) additional MRAD per 1μg m^{-3} increase in PM2.5 per 100,000 adults aged 18-64 (general population). Per year.
Restricted activity days (RAD)	PM2.5 (>2.4 μg m^{-3})	18-64	1	9,020 (± 552.5) additional RAD per 1μg m^{-3} increase in PM2.5 per 100,000 adults aged 18-64 (general population). Per year.
Work loss days (WLD)	PM2.5 (>2.4 μg m^{-3})	18-64	1	2,070 (± 155) additional work loss days per 1μg m^{-3} increase in PM2.5 per 100,000 people aged 15-64 in the general population. Per year.

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Table A-4 – *Continued from previous page*

Health impact	Pollutant	Age group	Risk fraction	Impact function
Years of life lost (YOLL_PM2.5)	PM2.5 (>2.4 μg m^{-3})	30+	1	101.4 (± 7.5) additional YOLL per 1 μg m^{-3} increase in PM2.5 per 100,000 people aged 30 and older in the general population. Per year.
Years of life lost (YOLL_NO_2)	NO_2 (>20 μg m^{-3})	30+	1	93.0 (± 6.9) additional YOLL per 1 μg m^{-3} increase in NO_2 per 100,000 people aged 30 and older in the general population. Per year.
Prevalence of bronchitic symptoms in asthmatic children (PBSC)	NO_2	5-14	0.158	111,427 (± 2,128) additional bronchitic symptom days per 1 μg m^{-3} increase in NO_2 per 100,000 children aged 5-14. Per year.

Table A-5: Prevalence of stove type by country (Blesl et al. 2011).

Country	Electric Stove	Other stove (Primarily Gas stove)
Austria	0.55	0.45
Belgium	0.35	0.65
Bulgaria	0.99	0.01
Cyprus	1.00	0.00
Czech Republic	0.33	0.67
Denmark	0.91	0.09
Estonia	0.61	0.39
Finland	0.96	0.04
France	0.44	0.56
Germany	0.91	0.09
Greece	0.89	0.11
Hungary	0.38	0.62
Ireland	0.28	0.72
Italy	0.31	0.69
Latvia	0.13	0.87
Lithuania	0.55	0.45
Luxembourg	0.87	0.23
Malta	0.69	0.31
Netherlands	0.24	0.76
Norway	1.00	0.00
Poland	0.68	0.32
Portugal	0.02	0.98
Romania	0.04	0.96
Slovakia	0.12	0.88
Slovenia	0.61	0.39
Spain	0.48	0.52
Sweden	0.96	0.04
Switzerland	0.87	0.23
United Kingdom	0.48	0.52

Table A-6: Fraction of people exposed to biomass by country (Torfs et al. 2007; Gens 2012).

Country	Ratio
Austria	0.52
Belgium	0.04
Bulgaria	0.21
Cyprus	0.14
Czech Republic	0.11
Denmark	0.21
Estonia	0.65
Finland	0.45
France	0.27
Germany	0.10
Greece	0.18
Hungary	0.15
Ireland	0.02
Italy	0.06
Latvia	0.96
Lithuania	0.37
Luxembourg	0.07
Malta	0
Netherlands	0.03
Norway	0.33
Poland	0.23
Portugal	0.29
Romania	0.30
Slovakia	0
Slovenia	0.39
Spain	0.13
Sweden	0.23
Switzerland	0.01
United Kingdom	0.01

Table A-7: Area and height of classroom by country.

Country	Room area (m^2)			Room height (m)		
	Min	Mode	Max	Min	Mode	Max
Austria	30.0	37.5	50.0	2.5	3.0	3.5
Belgium	30.0	49.0	88.0	2.5	3.5	4.0
Bulgaria	40.0	50.0	120.0	2.8	3.0	3.5
Cyprus	24.0	40.0	62.0	2.9	3.0	3.5
Czech Republic	54.0	57.0	68.0	3.2	3.4	4.0
Denmark	60.0	60.0	60.0	2.8	2.8	2.8
Estonia	35.0	48.0	121.0	2.5	3.0	3.5
Finland	45.4	66.9	69.4	2.7	3.2	3.6
France	43.0	58.0	94.0	2.5	2.9	4.0
Germany	20.0	50.0	80.0	2.8	3.2	3.7
Greece	42.0	64.0	73.6	2.9	3.5	3.6
Hungary	35.0	56.0	76.0	2.7	3.1	4.5
Italy	25.0	42.0	80.0	3.0	3.5	4.0
Ireland	52.0	60.1	76.0	3.0	4.4	5.3
Latvia	25.8	64.3	150.0	2.8	3.0	3.3
Lithuania	25.8	64.3	150.0	2.8	3.0	3.3
Luxembourg	20.0	50.0	80.0	2.8	3.2	3.7
Malta	36.0	45.1	61.2	3.1	3.5	4.3
Netherlands	20.0	50.0	80.0	2.8	3.2	3.7
Norway	45.4	66.9	69.4	2.7	3.2	3.6
Poland	33.4	48.1	52.5	3.2	3.2	3.2
Portugal	30.0	50.1	56.0	2.7	3.5	4.9
Romania	27.0	48.6	68.0	3.2	3.5	3.8
Slovakia	48.0	84.0	120.0	3.0	3.3	4.2
Slovenia	24.0	55.5	72.0	3.0	3.6	5.0
Spain	30.0	50.1	56.0	2.7	3.5	4.9
Sweden	60.0	60.0	60.0	2.8	2.8	2.8
Switzerland	20.0	50.0	80.0	2.8	3.2	3.7

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Table A-7 – *Continued from previous page*

Country	Room area (m^2)			Room height (m)		
	Min	Mode	Max	Min	Mode	Max
United Kingdom	52.0	60.1	76.0	3.0	4.4	5.3

Table A-8: DALY weights and durations.

Health impact	Pollutant	Weight	Duration
Bronchodilator usage adults	PM2.5	0.22	0.00274
Bronchodilator usage children	PM2.5	0.22	0.00274
Cardiac hospital admissions	PM2.5	0.71	0.038
New cases of chronic bronchitis	PM2.5	0.099	10
Infant mortality	PM2.5	1	80
Lower respiratory symptoms adults	PM2.5	0.099	0.00274
Lower respiratory symptoms children	PM2.5	0.099	0.00274
Minor restricted activity days	PM2.5	0.07	0.00274
Restricted activity days	PM2.5	0.099	0.00274
Respiratory hospital admissions	PM2.5	0.64	0.038
Work loss days	PM2.5	0.099	0.00274
Years of life lost	PM2.5	1	1
Years of life lost	NO_2	1	1
Prevalence of bronchitic symptoms in asthmatic children	NO_2	0.22	0.00274

Table A-9: Monetary values for health impact endpoint.

Endpoint	Mean (€)	SD (€)
Bronchodilator usage adults	80	4
Bronchodilator usage children	80	4
Cardiac hospital admissions	2,990	847
New cases of chronic bronchitis	66,000	9,500
Infant mortality	4,485,731	168,000
Lower respiratory symptoms adults	57	0
Lower respiratory symptoms children	57	0
Respiratory hospital admissions	2,990	847
Minor restricted activity days	57	0
Restricted activity days	194	0
Work loss days	441	0
Years of life lost (PM2.5)	59,810	29,387
Years of life lost (NO_2)	59,810	29,387
Prevalence of bronchitic symptoms in asthmatic children	80	4

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Content

Numerous epidemiological studies have demonstrated the damaging influence of air pollutants on human health. However, the environmental health studies up to now use urban background concentrations in the ambient air to estimate the health risks, while the inhalation of toxic substances, i.e. the concentration of pollutants, where the exposed person breathes, is the relevant indicator for estimating the health impacts. The main objective of this thesis is to assess the long-term exposure to fine particles and nitrogen dioxide for different European subgroups that are characterised by certain features including age, gender, region and socio-economic status. The exposure simulation is realised by developing a probabilistic model that incorporates an air quality model for estimating the ambient pollutant concentration, a mass-balance model for assessing the concentration of indoor micro-environments and a life course trajectory model for predicting retrospectively the transition between socio-economic states. The results of the exposure modelling are subsequently incorporated with exposure response functions (ERFs), aggregation factors and monetary values to assess health impacts and damage costs.

The modelling results indicate that for European countries the outdoor air concentration is a key factor for the total exposure, especially for NO_2. Similar to the trend of the ambient pollutant concentration, the average exposure for both pollutants in Europe reached a peak between the 1980s and 1990s and showed a notable decrease afterwards due to the introduction of emission reduction policies and smoking bans. For PM2.5 the most important indoor sources include smoking and cooking, while for NO_2 cooking and wood burning are the significant contributory factors. Considerable variance of the exposure levels has been observed among different European countries for both pollutants. The results also reveal that the exposure and its source distribution are dependent on gender, smoking habit and socio-economic status including household income level, employment status, education level and civil status.

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