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# Integration of indoor air quality in the life cycle assessment of buildings

Rachna Bhoonah

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**Integration of indoor air quality to the life cycle  
assessment of buildings**

**Intégration de la qualité de l'air intérieur dans  
l'analyse de cycle de vie des bâtiments**

Soutenu par

**Rachna Bhoonah**

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Procédés**

Composition du jury :

<b>Olivier, JOLLIET</b>	<i>Président</i>
Professeur Université Technique du Danemark	
<b>Cécile, BULLE</b>	<i>Rapporteuse</i>
Professeure Université du Québec à Montréal	
<b>Adélaïde, FERAILLE</b>	<i>Rapporteuse</i>
Professeure Adjoint Ecole des Ponts ParisTech	
<b>Christophe, GOBIN</b>	<i>Examineur</i>
Directeur de recherche Vinci Construction	
<b>Charlotte, ROUX</b>	<i>Examinatrice</i>
Ingénieure de Recherche ARMINES	
<b>Patrick, SCHALBART</b>	<i>Examineur</i>
Responsable Scientifique ARMINES	
<b>Bruno, PEUPORTIER</b>	<i>Directeur de thèse</i>
Directeur de recherche Mines Paris PSL	



### Intégration de la qualité de l'air intérieur dans l'analyse de cycle de vie des bâtiments

L'objectif de cette thèse est de prendre en compte les impacts liés à la qualité de l'air intérieur dans l'analyse de cycle de vie des bâtiments. Passant plus de 85% de notre temps dans des espaces intérieurs, nous sommes directement exposé(e)s aux polluants qui y sont présents, notamment les composés organiques volatils (COV) et les particules fines ( $PM_{2.5}$ ). Ces substances peuvent mener à divers effets notoires sur la santé, y compris des cancers, problèmes de développement ou de reproduction ou maladies cardiopulmonaires. Les seuils recommandés, en termes de concentration d'exposition, par l'OMS (Organisation Mondiale de la Santé) sont souvent dépassés, surtout dans les bâtiments ayant des volumes et des taux de renouvellement d'air relativement faibles.

Afin de lier la qualité de l'air (QAI) et l'analyse de cycle de vie (ACV), cette thèse propose une méthode de calcul d'impacts de la QAI en tenant compte du parcours des polluants : émission par de différentes sources, concentration dans l'air intérieur, assimilation dans le corps humain, et impacts. Les émissions de COV par des matériaux sont calculées avec un modèle basé sur le bilan massique des substances. Une méthode de calibrage du modèle est développée, en utilisant des mesures afin de préciser la valeur des paramètres incertains. Les émissions de COV et  $PM_{2.5}$  par les occupants et leurs activités sont collectées dans la littérature. Ces facteurs d'émission sont intégrés à l'outil INCA-Indoor, qui calcule les concentrations des polluants dans l'air. En fonction du scénario d'occupation (et donc de la présence des occupants dans les pièces), l'assimilation par la respiration, l'ingestion et le contact cutané est ensuite calculée. Les impacts (en DALYs, années de vie en bonne santé perdues) sont évalués avec les facteurs d'effet issus du modèle USEtox pour les COV et du modèle intégré d'exposition-réponse (IER) du *Global Burden of Disease* pour les  $PM_{2.5}$ .

Les impacts QAI sont additionnés aux impacts *end-point* de l'ACV, évalués avec la même unité. Il est cependant possible de les distinguer dans des résultats détaillés. L'applicabilité de la méthode proposée est démontrée sur des études de cas, où nous observons que les  $PM_{2.5}$  (intérieures et extérieures) sont les principaux contributeurs aux impacts de la QAI (40 à 94% des impacts totaux). Une augmentation du taux de ventilation permet d'évacuer plus de polluants, mais induit une hausse potentielle de consommation d'énergie pour le chauffage, dont les impacts sont aussi considérés par cette méthode. Des taux optimaux de ventilation réduisant les impacts totaux du bâtiment sont ainsi identifiés. Ceux-ci sont différents en fonction de l'usage des pièces, et plus élevés pour des pièces ayant de fortes sources de PM comme la cuisine. Dans le cas étudié, une ventilation double-flux avec échangeur de chaleur et filtre de particules réduit de 56% les impacts sur tout le cycle de vie du bâtiment.

L'intégration de la QAI dans l'ACV du bâtiment permet, dès la phase amont d'un projet de construction, d'estimer les impacts potentiels de la QAI. En d'autres mots, elle permet de faire des choix

de matériaux de construction et de revêtements, de dimensionner la ventilation ou les ouvertures et de fixer leurs orientations. Elle peut contribuer à mettre à jour les réglementations en matière de santé publique en fixant des valeurs cibles de ventilation dans différents secteurs, tout en évitant le déplacement des impacts vers d'autres étapes du cycle de vie.

## Integration of indoor air quality into the life cycle assessment of buildings

The main objective of this thesis is to account for impacts of indoor air quality in life cycle assessment of buildings. Spending more than 85% of our time indoors, we are directly exposed to indoor air pollutants. The two categories treated in this study are volatile organic compounds (VOCs) and fine particulate matter (PM<sub>2.5</sub>). Exposure to these substances increase the risk of diseases, including cancer, developmental, reproductive or cardiopulmonary diseases. Recommended exposure concentration limits by the World Health Organisation (WHO) are often exceeded, especially in buildings where the air volume and renewal rates are relatively small.

To link indoor air quality (IAQ) and life cycle assessment (LCA), a comprehensive framework is developed to calculate IAQ impacts, considering the full pollutant pathway: emissions by different sources, indoor air concentration, intake by humans and impacts. VOC emissions from materials are calculated using a mass-balance model. A method was developed to calibrate the emission model using measured data in order to fix uncertain parameters. Occupant and activity VOC and PM<sub>2.5</sub> emissions are obtained from literature. The emission rates are integrated to the INCA-Indoor model which calculates the air concentrations of substances. According to the occupancy scenario (thus the presence of occupants in the rooms), intake by inhalation, ingestion, direct dermal contact and gaseous dermal uptake are calculated. Health impacts (in DALYs, Disability Adjusted Life Years) are evaluated using effect factors from USEtox for VOCs and the integrated exposure-response model (IER) of the *Global Burden of Disease* for PM<sub>2.5</sub>.

IAQ impacts are added to LCA end-point impacts, calculated in the same unit. It is however possible to separate them in detailed results. The applicability of the suggested framework is demonstrated on case studies, where we note that PM<sub>2.5</sub> (indoor and outdoor) are the main contributors to IAQ impacts (40 to 94% of total impacts). An increase in ventilation rates allows to evacuate pollutants, but induces a potential rise in heat consumption, whose impacts are considered in the model. Optimal ventilation rates leading to an overall decrease in impacts are thus identified. These are different according to the use of the rooms, and are higher for rooms with strong PM sources, such as the kitchen. In the performed case study, double-flow ventilation with heat exchanger and particle filters lead to an overall 56% decrease in impacts of the building.

The integration of IAQ into buildings LCA allows, from the design phase of a construction project, to estimate impacts of IAQ. It helps in decision-making, namely for the choice of construction materials or surface finishes, to dimension adequate ventilation rates and openings and fix their orientations. It can also help to update regulations for public health, by fixing target ventilation rates in different sectors, avoiding the transfer of impacts to other stages of the life cycle.



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Symbol	Definition
ABM	Agent-based modelling
ALRI	Acute lower respiratory disease
ANSES	Agence nationale de sécurité sanitaire de l'alimentation
BIM	Building information modelling
BTEX	benzene, toluene, ethylbenzene and xylene
CFD	Computational fluid dynamics
CO	Carbon monoxide
CO <sub>2</sub>	Carbon dioxide
COPD	Chronic obstructive pulmonary diseases
DALY	Disability-Adjusted Life Years
ED50	Estimated lifetime dose having 50% chance of causing cancer
FU	Functional unit
GBD	Global burden of disease
HO	High occupancy
IAP	Indoor air pollutants
IAQ	Indoor air quality
IER	Integrated exposure-response
IHD	Ischaemic heart disease
INSEE	Institut National de la Statistique et des Etudes Economiques
LCA	Life cycle assessment
LCIA	Life cycle impact assessment
LO	Low occupancy
NO <sub>x</sub>	Nitrogen oxides
O <sub>3</sub>	Ozone
OECD	Organisation for Economic Co-operation and Development
PM	Particulate matter
PM <sub>2.5</sub>	Particulate matter of diameter under 2.5 µm
PVC	Polyvinyl chloride
SBS	Sick building syndrome
SOA	Secondary organic aerosol
SVOC	Semi-volatile organic compound
VOC	Volatile organic compound
WHO	World Health Organisation

Symbol	Definition	Unit
$\Delta t_{\text{exposure}}$	Duration of exposure	s
$\nabla H_v$	Enthalpy of vaporisation	J/mol
ACH	Air changes per hour	1/h
BMR	Basal metabolic rate	(MF/day)
BR	Breathing rate	$\text{m}^3/\text{s}$
$C_0$	Theoretical minimum exposure level	$\mu\text{g}/\text{m}^3$
CF	Characterisation factor	$\mu\text{DALY}/\mu\text{g}_{\text{emitted}}$
$C_{\text{in}}$	Indoor air concentration	$\mu\text{g}/\text{m}^3$
$C_{\text{in,inc}}$	Increment in indoor concentration	$\mu\text{g}/\text{m}^3$
$C_{\text{out}}$	Outdoor air concentration	$\mu\text{g}/\text{m}^3$
$D_m$	Material diffusion coefficient	$\text{m}^2/\text{s}$
DR	Particle deposition rate	1/s
EF	Effect factor	$\mu\text{DALY}/\mu\text{g}_{\text{intake}}$
$e_{\text{MF0}}$	Sum of RMSLE	-
$ER_{\text{activity}}$	Activity emission rate	$\mu\text{g}/\text{s}$
$F_{\text{emission}}$	Emission factor	$\text{kg}_{\text{emitted}}/\text{kg}_{\text{burnt}}$
FF	Fate factor	$\text{kg}_{\text{in compartment}}/\text{kg}_{\text{emitted}}$
HI	Health impacts	DALY
$HV_{\text{wood}}$	Net heating value of wood	MJ/kg
iF	Intake fraction	$\mu\text{g}_{\text{intake}}/\mu\text{g}_{\text{emitted}}$
ILR	Individual lifetime risk	death/person/lifetime
IS	Impact score	$\mu\text{DALY}$
$K_{\text{ma}}$	Material-air partition coefficient	-
$K_{\text{oa}}$	Octanol-air partition coefficient	-
MALE	Mean absolute log error	-
$MF_0$	Initial mass fraction	$\text{kg}_{\text{substance}}/\text{kg}_{\text{material}}$
$M_{i,r}$	Annual mortality in region r due to disease i	deaths/year
MR	Metabolic rate	met
$M_s$	Mass of substance inside the material	kg
MW	Molecular mass	g/mol
$M_{\text{wood}}$	Mass of wood	kg
P	Pressure	kPa
$P_{\text{heat}}$	Average heating power	W
POP	Number of occupants	-
RMSE	Root mean squared error	$\mu\text{g}/\text{m}^3$
RMSLE	Root mean squared log error	-
RR	Relative risk of developing a disease	-
SF	Severity factor	DALY/case
$SF_{i,r}$	Severity factor specific to region r and disease i	DALY/death
T	Absolute temperature	K
$V_{\text{room}}$	Room volume	$\text{m}^3$
XF	Exposure factor	$\text{kg}_{\text{intake}}/\text{kg}_{\text{in compartment}}$



This thesis is mainly based on the following works, which have either been published, or are in preparation:

- Bhoonah R., Maury-Micolier A., and Jolliet O. 2023. ‘Integrated Empirical and Modelled Determination of the Human Health Impacts of Building Material VOCs’. *Building and Environment*, June, 110523. <https://doi.org/10.1016/j.buildenv.2023.110523>. (**Published**)
- Bhoonah R., Maury-Micolier A., Jolliet O. and Fantke P. 2023. ‘Fine particulate matter exposure and health impacts from indoor activities’. (**Accepted by Indoor Air**)
- Bhoonah R., Mendez M., Maury-Micolier A. 2023. ‘Human health impacts and indoor chemical reactions of VOCs from cleaning products and occupants’. (**Submitted to Atmospheric Environment**)
- Bhoonah R., Roux C. Peuportier B. 2023. ‘Integration of IAQ into the LCA of buildings: an aid to decision-making’. (**In preparation, targeted journal: International Journal of Life Cycle Assessment**)

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- Bhoonah R., Maury-Micolier A., Schalbart P., Peuportier B. and Jolliet O. ‘Intégration de la qualité de l’air intérieur dans l’analyse de cycle de vie des bâtiments’, IBPSA France, Châlons-en-Champagne May 2022
- Bhoonah R., Schalbart P. and Peuportier B. ‘Construire et mesurer la performance environnementale’, Université du Lab Vinci, Paris November 2021



# Introduction

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## 1. Definitions and a brief history

Through history, humans have migrated towards more and more indoor living, to a point where we are now called “The Indoor Generation” (YouGov 2018). The main purpose of indoor environments is to provide shelter against outdoor conditions such as rain, strong winds and cold temperatures, as it was the case for early caves and huts. They have further evolved to also provide shelter for intimacy, privacy and security. Our homes were adapted for our comfort and divided into several sections, for instance with the introduction of indoor kitchens, toilets or individual bedrooms. Indoor environments now additionally serve a wide variety of uses, ranging from work or study, to relaxation or leisure. Our work places slowly moved from outdoor farms and fields to indoor workspaces. Multiple objects were developed for our distraction, e.g. televisions, computers, video or board games and musical instruments, further encouraging humans to spend more time indoors – presently about 85% of our time (ANSES 2010).

We went a long way in creating a comfortable environment. We devised means of heating, starting with indoor open fires with central roof opening, before creating chimneys, and started using insulation materials from the mid-20<sup>th</sup> century. Glazed openings were added to increase natural lighting and ventilation. Decorative elements such as wallpaper, paint and stylish furniture were developed for our aesthetical and visual pleasure. With concern for “clean” environments, cleaning products and detergents were created. Air fresheners, incense sticks and scented candles were also created for olfactory pleasure and to mask undesirable odours. Several of these components, however, have resulted in poorer air quality.

Greeks and Romans were aware of air pollution, especially in crowded cities or mines (Hippocrates, 460–377 BC), while consciousness of indoor environments started to grow in the mid-19<sup>th</sup> century, especially with the appearance of tuberculosis (Sundell 2004). In the 19<sup>th</sup> century, several professors in hygiene (Max von Pettenkofer and Herman Rietschel in Germany, Elias Heyman in Sweden) studied the presence of CO<sub>2</sub> and other organic substances in air. They set the recommended CO<sub>2</sub> concentration limit of 1000 ppm still used today. A minimum ventilation rate of 2 L/s per person was first recommended in 1836 by Thomas Tredgold (mining engineer), followed by 15 L/s by ASHRAE (American Society of Heating, Refrigerating and Air-conditioning Engineers) in 1895. Florence Nightingale (1820-1910), known as the founder of modern nursing, wrote about the role of fresh air and the link with health and hygiene of patients.

Until the late 20<sup>th</sup> century, the main indicator for indoor air quality was occupant perception, which were thought to indicate potential health risks, and the main pollutant source was considered to be exhaled air (Fanger 1988; Fanger et al. 1988). Other pollution sources such as materials, computers and the effect of ventilation and humidity on indoor air were addressed about a decade later (Fang et al. 1999; Wargocki et al. 1999). Issues related to radon started to arise in the late 1960s, formaldehyde and

the sick building syndrome (SBS) in early 1970s and allergies in the late 1990's. IAQ and ventilation have been of particular attention during the pandemic (2020-2023). However, viruses are only one of the types of pollutants that exist indoors. Others, including particulate matter (PM) or volatile organic compounds (VOCs), can have adverse long-term effects on health.

## 2. Observed effects of IAQ

Effects of poor IAQ can be short-term. For instance, the sick building syndrome (SBS) refers to health symptoms or low comfort levels related to the indoor atmospheric environment (including air pollution, electromagnetic radiation or other unknown causes). The symptoms can be both psychological and physiological, e.g. lethargy, irritability, dry throat, headache or chest pain (Norhidayah et al. 2013). Odours, temperature and humidity are markers of IAQ, indicating lack or presence of ventilation. However, human's perceptions can be biased and vary across individuals, especially because we do not all detect odours in a similar way and some pollutants are odourless.

Long-term adverse effects of bad indoor air quality result from the exposure to certain volatile organic compounds (VOCs), such as formaldehyde passively emitted by wood composites or cleaning products, that can lead to serious health impacts, including the risk of cancer or heart failure (Y. Huang, Ho, Ho, Lee, Yu, et al. 2011; Zheng et al. 2020; Ran et al. 2018). Other harmful VOCs, such as limonene, are actively used by occupants since they are added to air fresheners or candles for their smell.

Another indoor pollutant that represents a serious health risk is particulate matter (PM), mainly generated by fuel combustion. In developing countries using solid fuels indoors, it was observed that the most important cause of deaths for children under five years of age was acute respiratory infections, representing 64 % of deaths (Smith and Mehta 2003).

## 3. Standards and regulations

In an effort to reduce the impacts of bad IAQ, several standards have been set regarding different pollutants. In France, following a decree passed in 2011, products sold in the country have to possess a regulatory label that indicates the emission rates of VOCs (decree of 19 april, 2011). The French National Health and Food Security Agency (ANSES) identified 13 substances of concern according to their toxicological profile, and proposes a list of Indoor Air Quality Guideline Values (IAQGs) (ANSES 2013a). Guidelines have also been developed to rate buildings. Some examples are: Home Quality Mark (HQM, 2015), developed by the Building Research Establishment (BRE), the LEEDv4 Homes Design and Construction (2016) and the WELL Building Standard on air quality. Onsite performance tests were carried out to measure the air concentration of different pollutants, including VOCs, PM, CO and O<sub>3</sub>, in order to provide ratings.

However, even if these efforts help to classify existing buildings and materials (which can be chosen according to their label), such regulations and classifications concern the concentrations of

pollutants indoors, while each pollutant can have different effects on health. A unique basis for the evaluation of their impacts is hence needed to avoid the transfer of impacts from a life cycle point of view. Certain choices that allow to lower IAQ impacts, can, on the other hand, increase impacts of other stages of the building's life cycle or other environmental pathways. For instance, a good ventilation allows to reduce the concentration of indoor pollutants, but may lead to a higher heating demand, and hence higher impact related to energy consumption. Certain materials are not sources of volatile pollutants, but could have high impacts linked to their raw material extraction or manufacturing.

#### 4. Life cycle assessment

Having a life cycle perspective can help to avoid the transfer of impacts from one stage of a building's life cycle to another, right at the design phase. This precautionary approach is, for instance, applied by life cycle assessment (LCA) practitioners using tools that evaluate the environmental performance of products and buildings before their construction or renovation, based on LCA methods and life cycle inventory databases. Impacts are calculated on different environmental indicators, falling under three broad categories: human health, biodiversity and natural resources. The life cycle stages considered are generally: 1) extraction, fabrication and transport of materials, 2) construction, 3) operation, 4) renovation and 5) end-of-life.

However, current LCA methods do not consider IAQ impacts. Several efforts have been made to model different aspects of IAQ such as emission, occupant intake or health impacts. Up to date, no holistic framework is available to consider all contributions to IAQ in the design phase. A framework that answers such design questions by combining LCA and IAQ tools in an integrated approach could help in decision making in the construction sector in order to help reduce impacts of all stages of a building's life cycle.

#### 5. Thesis Outlines

The main objective of this thesis is: *to develop a framework for the evaluation of IAQ health impacts and their integration into building LCA.*

The different chapters and main questions addressed are summarised as follows:

##### Chapter 1: State of the art

In the first chapter, a review of studies treating life cycle assessment (LCA) and indoor air quality (IAQ) is presented. We discuss the pollutant pathway, which begins from the identification of key pollutants present indoors, their sources, their emission, the exposure of occupants, their intake, and, eventually, their impacts on health. For each step of the pathway, different existing models are identified. We particularly focus on studies that cover multiple steps of the pathway. We conclude by proposing a method to integrate IAQ to LCA, and identify main knowledge gaps to achieve this objective.

## Chapter 2: Emission model and health impacts of VOCs from materials

Materials are identified as important sources of chemicals indoors. Though emission data is available, they are only valid at specific times and conditions, while emission models contain unknowns and uncertainties. A calibration method is developed that extrapolates measured air concentration data of volatile organic compounds (VOCs) emitted from materials using a multilayered emission model. The framework is validated on two monolayered materials and its applicability to a multilayered material is assessed. Using the complete concentration profile, the intake and health damages are calculated for the two materials. Sensitivity tests allow to assess the effect of model parameters on short, medium and long-term impacts.

## Chapter 3: Health impacts of VOCs from occupants and indoor activities

Studies show that occupants and indoor activities such as cleaning with detergents and air fresheners are sources of VOCs. A framework is proposed to calculate their consequent health damage and applied to a case study. VOC emission data for cleaning and the use of air freshener are introduced in the INCA-Indoor model in order to evaluate corresponding air concentrations. Resulting health impacts for occupants are calculated in an example case of an office. The effect of indoor air chemistry is assessed, and we conclude that chemical reactions forming VOCs or secondary organic aerosols should not be neglected.

## Chapter 4: Health impacts of fine particulate matter from indoor activities

From literature,  $PM_{2.5}$  emission data and the size-distribution of indoor PM is obtained. These elements are integrated to the INCA-Indoor dynamic model in order to calculate air concentrations. A parametric one-compartment model is updated to consider deposition of particles penetrating from outdoors and results are compared with the dynamic model. In order to facilitate the integration of IAQ damages of different activities into LCA, the characterisation factors of different common indoor activities emitting  $PM_{2.5}$  are derived for different air change rates and occupancy, using the GBD IER non-linear model.

## Chapter 5: General framework integrating IAQ impacts to building LCA: application to a case study

A general framework is developed combining material, occupant and activity emissions based on the previous chapters, and LCA results. The applicability of this framework is demonstrated on a tertiary building and a residential building, with activity scenarios defined according to room use (office, kitchen, meeting room or living room). Material and activity VOC emissions,  $PM_{2.5}$  emissions and resuspensions from activity and outdoor  $PM_{2.5}$  concentrations are integrated into INCA-Indoor. Health damages are calculated based on simulated concentrations (considering indoor chemistry) and occupancy scenarios. Different mechanical ventilation rates are tested for each room under study and

the optimal trade-off is identified. For the home case study, the effects of position and number of open windows on the air change rates are evaluated. Three different energy sources for heating are compared. Gains of a combination of double-flow ventilation fitted with filters is evaluated.

We conclude this thesis with a summary of our findings from each chapter, a discussion on the possible applications of our work and suggestion of perspectives.





# 1. State of the art

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Dans ce chapitre, l'état de l'art sur le sujet de la qualité de l'air intérieur (QAI) et son intégration dans l'analyse de cycle de vie (ACV) des bâtiments est présenté. La QAI est un enjeu majeur de santé humaine sur lequel il est possible d'agir dès la conception des bâtiments, or elle n'est pas prise en compte dans les méthodes actuelles d'écoconception basées sur l'ACV. Son intégration dans l'ACV, à travers un outil d'aide à la décision aiderait à réduire les impacts sur tout le cycle de vie du bâtiment et éviter les transferts d'impacts. Par exemple, une bonne ventilation permet d'évacuer des polluants de l'air intérieur et réduire les effets sur la santé des occupants. Toutefois, une augmentation du renouvellement d'air pourrait accroître la consommation d'énergie pour le ventilateur et le chauffage. Un taux optimal de ventilation pourrait être évalué avec un tel outil associant ACV et QAI.

Dans cet état de l'art, différents types de polluants présents dans l'air intérieur sont présentés. Le devenir des polluants, de l'émission par différentes sources aux effets sur la santé, est étudié par la suite. Différents données et modèles décrivant chaque étape sont abordés : 1) données d'émission de particules fines et composés organiques volatils (COV) par des activités, 2) modèles d'émission de COV par des matériaux et paramètres physicochimiques, 3) simulation de concentrations de polluants dans l'air basée sur des modèles de mouvement d'air, 4) calcul de la quantité de polluants assimilée par les différentes voies : inhalation, ingestion, contact cutané, absorption cutanée et 5) modèles de calcul d'impacts des COV et particules fines sur la santé avec une unité commune à l'ACV : le DALY (Disability-adjusted life years). Des modèles traitant l'intégration de la QAI dans l'ACV du bâtiment sont évoqués, mais ceux-ci ne traitent que certaines parties de la chaîne entre émissions et impacts. La nécessité d'un modèle intégrant toutes les étapes mentionnées précédemment est identifiée, ainsi que les verrous scientifiques, dans le but de réaliser l'écoconception des bâtiments avec un modèle d'aide à la décision qui intégrerait la QAI à l'ACV.

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## Summary

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This chapter presents a review of methods that have been developed for the calculation of health impacts in life cycle assessment (LCA) and indoor air quality (IAQ), in particular pollutant emission rates and air concentrations based on building characteristics and occupant behaviour, and the response of humans to different types of pollutants.

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## 1. Indoor air Quality

### 1.1. The indoor environment

Humans spend a high fraction of their time indoors: 85 to 90% (ANSES 2010; U.S. Environmental Protection Agency 1989), and are exposed to indoor air which can contain pollutants. The indoor environment includes buildings that serve a wide variety of uses, such as residential, office, commercial or leisure. These environments often contain particulate or gaseous substances known as indoor air pollutants (IAPs) that affect occupant health. IAPs can penetrate indoors from outdoors (emitted for e.g. by vehicles, factories or volcanoes) through unfiltered ventilation. They can also be directly emitted indoors by occupants, activities, or materials.

Because there is strong evidence linking IAPs to negative impacts on human health (WHO 2012; Wang et al. 2013; Mandin 2020; Karr et al. 2021; Ali et al. 2021; Torkmahalleh et al. 2021; L. Huang et al. 2022a; Mainka and Fantke 2022), different actions aim at mitigating this pollution, such as monitoring or modelling. For both of them, indoor pollutant concentrations are generally measured or predicted. Concentrations are, however, not indicators of the actual impacts of pollutants on health. These impacts both depend on the exposure of occupants (duration/frequency of presence and intake rate) and the toxicity of the substance. For instance, in building design, different scenarios could lead to the increase or decrease of different substances. If their impacts on health are unknown, optimal solutions cannot be devised.

Furthermore, in the case of monitoring, the main disadvantage is that measurements of indoor pollutants are taken during the building's lifespan: it does not follow a precautionary approach. Some design choices can influence IAQ impacts and can help to avoid them before the building's construction. For instance, low-emitting materials can be chosen and ventilation can be adequately dimensioned to evacuate pollutants (Poirier et al. 2021). However, reasoning with a life cycle perspective, solutions reducing IAQ impacts could lead to the transfer of impacts. For instance, low-emitting materials or paint might have higher impacts at its extraction/fabrication or at its end-of-life. Materials with lower chemical emissions can also have lower insulation properties and thus lead to higher heat consumption to maintain comfort indoor temperatures. In addition, higher air renewal rates that allow to evacuate indoor pollutants could increase energy consumption for ventilation, heating (in winter) and cooling (in summer).

In order to avoid this transfer and reduce overall building impacts on human health, the entire building life cycle should be considered (Orosa et al. 2020). This life cycle perspective is considered in building design using Life Cycle Assessment (LCA), which is a design-making tool allowing to evaluate environmental and health impacts of a building. It also allows to compare different variants of the same building, considering possible transfers of impacts, as an aim to reduce overall impacts throughout its life cycle.

## 1.2. LCA: a tool to evaluate the environmental and health burden of buildings

Buildings have an important contribution to global environmental impacts, given the scale of their stock, the important amounts of materials required for construction, and operational energy consumption. The building and construction sector represents 30% of the final energy use and 27% of carbon dioxide (CO<sub>2</sub>) emissions worldwide according to the International Energy Agency (2022), while built-up areas keep growing (Haščič and Mackie 2018). Each process involves the input and output of substances to and from the environment, causing the depletion of resources, and harm to living organisms. In the context of rising conscience regarding environmental issues, several methods have been developed as an effort to reduce environmental impacts of buildings, such as material or substance flow analysis, environmental risk assessment and Life Cycle Assessment (LCA).

LCA is a multi-criteria and multi-step environmental assessment method that considers the different stages in the life cycle of a product or service. In the case of a building, these stages are: *raw materials extraction, fabrication of products, construction, operation, renovation* and *end-of-life*. At each stage, the energy and material use are considered and their impacts are evaluated according to a set of environmental indicators, categorised in three main areas of protection (resources, ecosystem quality and human health) and two areas of concern (water and carbon), defined by Impact World+ (Bulle et al. 2019). This method is useful to identify the key contributors to environmental impacts, such as energy, equipment, materials or processes. It allows impacts to be prevented right from the design phase, preliminary to the building's life cycle. The first step of an LCA, according to principles and framework defined in ISO 14040 (Arvanitoyannis 2008), is the definition of objectives. These ultimately determine how the method is applied. A *functional unit* is then defined, including: a function (e.g. dwelling, tertiary), a quantity (e.g. surface area), quality (for e.g. comfort of occupants: temperature, brightness, noise) and duration (e.g. 1 year or the building's whole life cycle). Once a functional unit has been defined, comparative LCA can be used as a decision-making tool, for choosing between different variants serving the same function. [Figure 1-1](#) represents the life cycle of a building with the different life cycle stages.



*Figure 1-1: Life cycle of a building*

In the *materials fabrication* stage, impacts related to the extraction of raw materials and their transformation are calculated. Current building LCA tools consider energy and water consumption, waste production and the transport of occupants during the *operation* stage (which is the longest phase of the building's life cycle). It usually represents the highest share of impacts for most indicators, due to water consumption and energy consumption for water and space heating (Ramesh et al. 2010; Peuportier et al. 2013; Hoxha et al. 2017), and depends on the occupancy. At the *renovation* stage, equipment can be replaced, and reparations may be needed, requiring material and energy inputs. There can also be renovations leading to a reduction in heat consumption on the long term and consequent environmental impacts, such as the addition of more insulation or heat recovery ventilation. Finally, at *end-of-life*, a building is deconstructed or demolished, and the waste is either landfilled, incinerated or recycled/reused. The demolition process, transport to waste management sites and different waste treatment processes require energy and infrastructures that directly or indirectly emit pollutants (such as carbon dioxide and particulate matter) into the atmosphere.

With the help of LCA for ecodesign, decisions leading to least environmental impacts considering all stages of the building's life cycle can be made. These include the choice of construction materials, type and thickness of insulation, ventilation type and rate, size and orientation of openings. These choices have different impacts which can be direct (fluxes linked to material consumption or energy for construction) or indirect (on the long term: energy required for lighting, heating and cooling).

The impacts of buildings (or any other human activity) can be global or local. An example of a global impact is the emission of greenhouse gases contributing to global warming. Local impact refers to damage on a regional, continental or local scale (Ross and Evans 2002) where resources are consumed or substances are emitted. For example, emissions from industrial facilities into surrounding environment directly affect populations on a local scale (Eom et al. 2018) and water use has different impacts based on the country or region's reserves (Pfister 2015). The spatialisation of impacts in LCA is thus widely studied (Hauschild 2006; Aissani 2008; Nitschelm et al. 2016) and recommended by LCA specialists (Humbert et al. 2011; Patouillard et al. 2018; J. Li et al. 2021). On an even smaller scale, some materials and indoor activities can emit pollutants into air, and consequently affect the health of building occupants. This aspect is generally not considered by current LCA tools.

### 1.3. Accounting for IAQ impacts in LCA

IAQ's contribution to total health damages of a building's life cycle could be important: up to 12% (Collinge et al. 2013) or more than 50% with thick insulation (Micolier 2019) and about 156  $\mu$ DALY/year (83 min/year) for preschool children exposed to PM<sub>2.5</sub> and metals in kindergartens (Mainka and Fantke 2022). In order to reduce impacts of buildings on their whole life cycle, considering occupant exposure to pollutants during the *operation* stage, IAQ impacts have to be integrated to LCA. For this, IAQ impacts have to be quantified with similar indicators and integrated into building LCA as a decision-making aid.

#### 1.3.1. Existing models and knowledge gaps

Different studies have aimed at integrating IAQ into LCA. For example, Humbert et al. (2011) proposes a framework to evaluate health impacts of particulate matter (PM). The study summarises literature-based intake fraction values for different contexts: urban/rural, indoor/outdoor, but did not include health impact values. The pollutant pathway is illustrated in [Figure 1-2](#), based on Humbert et al. (2011) and the USEtox framework (Fantke et al. 2017).



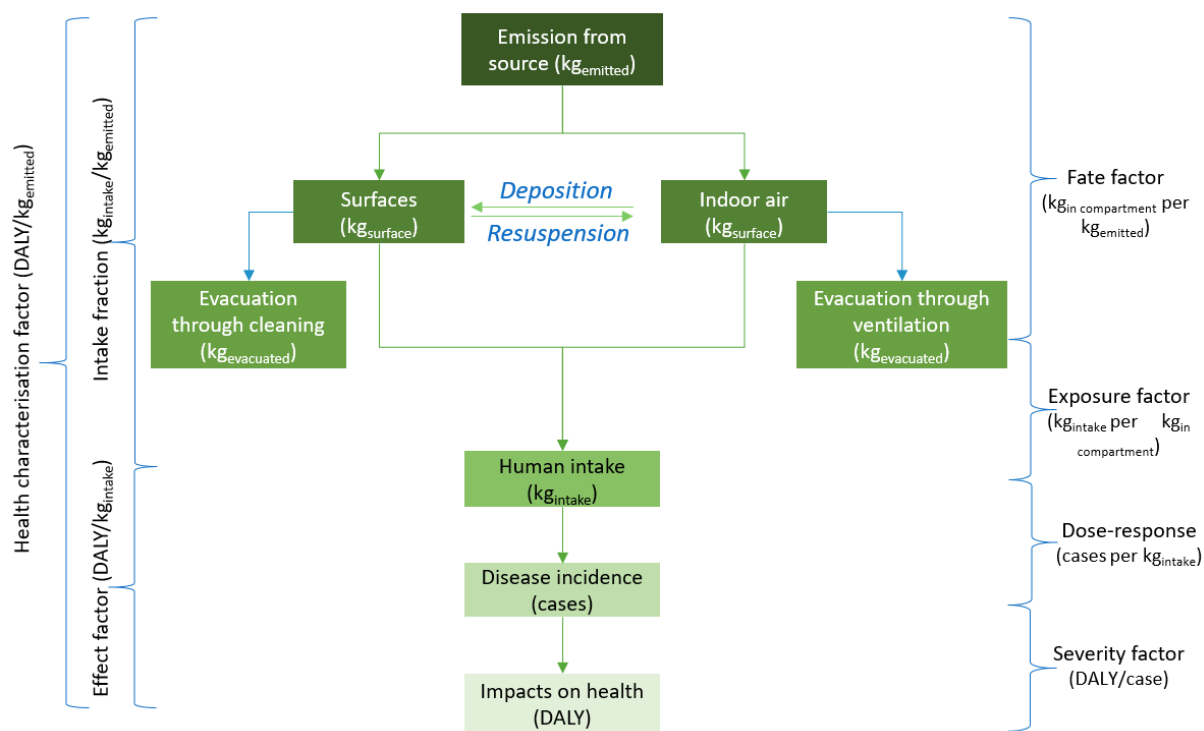


Figure 1-2: Pollutant pathway from emission to impact adapted from the PM emission to damage framework proposed by Humbert et al. (2011) and USEtox (Fantke et al. 2017)

Pollutants are emitted by different indoor sources or penetrate from outdoors (also considered as a source). They are mixed in indoor air and particles also get deposited on surfaces before potentially being resuspended or evacuated by ventilation or cleaning. The pollutant's fate factor ( $\text{kg}_{\text{in compartment}}/\text{kg}_{\text{emitted}}$ ) determines its presence in different compartments (indoor air, outdoor air and surfaces). Occupant exposure to pollutants are calculated by the exposure factor ( $\text{kg}_{\text{intake}}$  per  $\text{kg}_{\text{in compartment}}$ ). The disease incidence is calculated according to the dose-response factor (cases per  $\text{kg}_{\text{intake}}$ ) and impacts are calculated using the severity factor (DALY/case). The framework, however, does not include models that treat emission of pollutants from different sources.

Studies focusing on source emissions include the development of material VOC emission models (Yan, Zhang, and Wang 2009; Guo 2013; L. Huang et al. 2020), the simulation of indoor air concentrations based on emission rates (Hellweg et al. 2009; Fantke et al. 2017; Yang et al. 2022) and measurement of particulate matter emission rates from different activities (He et al. 2004; 2010; Licina, Tian, and Nazaroff 2017; Aquilina and Camilleri 2022). Other studies have focused on the intake of pollutants and their effects on health (Burnett et al. 2014; GBD 2015 Risk Factors Collaborators 2016; Fantke et al. 2017). Air concentration measurements of  $\text{PM}_{2.5}$  and metals have been coupled with effect factors in order to evaluate health damages (Mainka and Fantke 2022). Table 1-1 summarises the different studies and the steps of the pollutant pathway that are addressed by each.

Table 1-1: Summary of studies treating different steps of the pathway linking IAQ to LCA

Reference	Source	Emission	Concentrations	Intake	Impacts
Yan et al. 2009; Guo 2013; L. Huang et al. 2020	Materials	Model (VOCs)	Calculation of indoor air concentrations and material concentrations	-	-
He et al. 2004; 2010; Licina et al. 2017; Aquilina and Camilleri 2022	Activities	Measurements (PM <sub>2.5</sub> )	Measurements	-	-
Hellweg et al. 2009; Humbert et al. 2011; Fantke, Jolliet, et al. 2017; Yang et al. 2022	Indoor and outdoor	-	-	Intake fractions	-
Burnett et al. 2014; GBD 2015 Risk Factors Collaborators 2016; Fantke et al. 2017	Ambient and indoor PM <sub>2.5</sub>	-	Global measured ambient concentrations	Average intake	Non-linear relationship derived between exposure concentration and effect factor
Mainka and Fantke 2022	-	-	Measured indoor PM <sub>2.5</sub> and metal concentrations	Calculated	Calculated
Micolier (2019)	Insulation materials	Model (VOCs)	Calculation of indoor air concentrations and material concentrations	Calculated	Calculated

In order to integrate IAQ at the design phase, concentrations, exposure and health damages have to be modelled, and linked to the building's LCA-related damages. From Table 1-1, we note that studies have aimed at treating different pollutant sources: materials, activities and outdoor air, and two main pollutant categories: VOCs and PM<sub>2.5</sub>. Some studies are parametric, considering average values, while others are dynamic (i.e. emissions, concentration or exposure can vary as a function of time). Most studies do not treat the whole pathway, except for Micolier (2019). However, no study was found treating different pollutant sources together (material, occupants, activities and outdoor air) through the whole pathway from emission to impact.

### 1.3.2. Proposed solutions

Conventional LCA methods do not currently evaluate IAQ health impacts, and no holistic approach is available linking IAQ to LCA. IAQ simulation tools are often dissociated with LCA and results are not comparable (health impacts v/s simulated or measured pollutant concentrations). Our study aims at integrating IAQ to LCA. Two types of methods can be used for this integration: inventory-based method and simulation-based method. In the first one, emissions are directly linked to inventories,

similar to current LCA tools: 1) users can indicate materials, air change rates, occupancy rates and activities (if specific activities are known) or a building category according to its use (for pre-defined standard scenarios, e.g. classroom, office, kitchen...), 2) the relevant indoor emissions (fluxes) are obtained from the activity or material inventories and, 3) based on the pollutant impact pathway, characterisation factors are calculated as a function of standard parameters or those defined by the user and coupled with emission data to evaluate health impacts. It has the advantage of being simple to use, can be readily integrated into LCA tools and requires low calculation times. However, temporal fluctuations might not be fully represented.

The second model is based on simulations to obtain full emission, concentration and exposure profiles that integrate the dynamic nature of the building's occupancy. Simulations are valid for the case under study, can be more time-consuming, require more input data and background knowledge from the user and have higher calculation times. However, they are more adapted to study specific building designs and compare with variants.

This study aims at proposing a framework that can be applied to the two models, both relying on the pollutant pathway from emission to impact summarised in [Figure 1-2](#), adapted from Humbert et al. (2011) and Fantke et al. (2017). The different models discussed in section 1.3.1 existing in literature that treat one or several steps of the pathway can be linked in order to achieve the integration of IAQ into building LCA. In order to develop the full framework, these models and additional studies providing relevant information on IAQ and human health are described in details in the next section. The main objective is to identify pollutants with known and quantified effects on human health, models that can answer to our goal and that can be linked, and other parameters that were reported as relevant to IAQ and health.

## 2. Calculation of IAQ impacts: the pollutant pathway

### 2.1. Types of pollutants and their sources

To calculate impacts of IAQ, common indoor air pollutants (IAPs) are identified. They are either biological or non-biological, organic or inorganic. Some of these pollutants are presented in [Figure 1-3](#) and organised by their nature and particle diameter. It is to be noted that no information is given on their dangerousness in this figure.

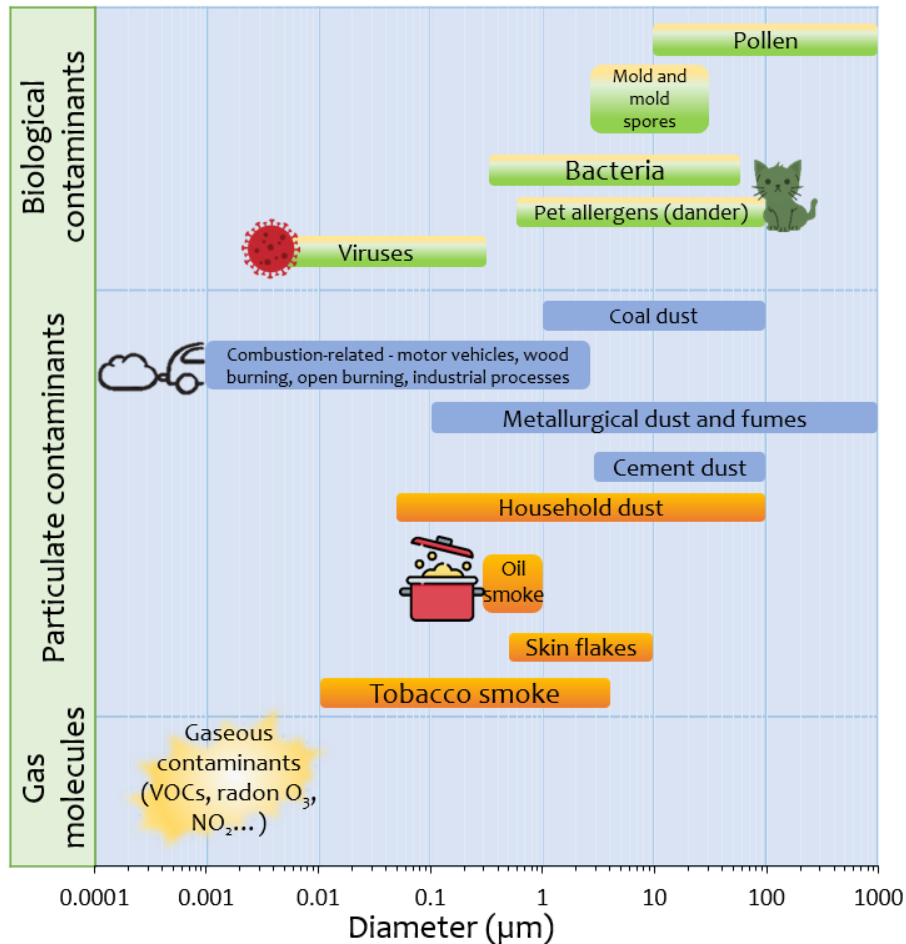


Figure 1-3: Some indoor air pollutants and some sources, classified by their nature (biological or non-biological) and particle diameter: created by author, data from Engineering toolbox (2005)

The presence of biological contaminants is strongly influenced by occupants: having pets, frequency of cleaning, presence of sick persons or lack of maintenance (for e.g. unrepaired leakage that increases dampness and mould occurrence). Non-biological contaminants, in the form of particles or gases, are both influenced by user behaviour and building design.

Particles present in air are called particulate matter and are classified by their diameters into different categories: PM<sub>10</sub> (coarse, diameter ≤ 10 μm), PM<sub>2.5</sub> (fine, diameter ≤ 2.5 μm) and PM<sub>0.1</sub> (ultrafine, diameter ≤ 0.1 μm). Fine particles develop mainly through gas transformation or combustion and are composed of nitrates, sulphates, elementary carbon, organic compounds, and trace elements, including metals, while coarser particles are usually formed by the mechanical degradation of large particles, typically consisting of dust, fibres, metal, pollen, and bacteria.

Gaseous pollutants include volatile organic compounds (VOCs), semi-volatile organic compounds (SVOCs), carbon monoxide (CO), ozone (O<sub>3</sub>), nitrogen dioxide (NO<sub>2</sub>) and radon. VOCs are defined as any organic compound containing at least one carbon and one hydrogen atom in its molecular

structure and having a boiling-point range between the limits of 50 °C to 260 °C (World Health Organisation 1989). They are identified by the odours they emanate and the adverse health effects they cause, particularly the irritation of eyes, nose or throat and high toxicity at certain concentrations (A. J. Li, Pal, and Kannan 2021; L. Huang et al. 2022a; Halios et al. 2022). One of the most widespread VOC is formaldehyde (CH<sub>2</sub>O), characterised by its pungent smell. SVOCs are distinguished from VOCs by their higher molecular weight and boiling points, hence slower emissions.

Different particulate and gaseous pollutants and their sources are presented in [Table 1-2](#) (ADEME 2015; Halios et al. 2022)

*Table 1-2: Sources of some indoor chemicals (ADEME 2015; Halios et al. 2022)*

<b>Indoor pollutants</b>	<b>Some sources</b>
VOCs (formaldehyde, benzene, toluene, xylene, styrene, octane, trichloroethylene...)	Construction or decoration materials, furniture, detergents, cooking, smoking, particle board, insulation foams, carpet, textiles, glues, paint and varnish, ink, insecticides, cosmetics, deodorants, air fresheners, hair sprays
Glycol ethers	Some water-based paints, ink, varnish, glue, detergents, thinners, cosmetics
Carbon monoxide (CO)	Equipment for heating and hot water production through combustion that are not efficient or well-maintained, tobacco smoke
Nitrogen oxides (NO <sub>x</sub> )	Combustion (gas cooker, gas water heater, wood heater, tobacco smoke)
Sulphur dioxide (SO <sub>2</sub> )	Coal combustion, oil
Pesticides	Chemicals in wood treatment, products for plants and animal treatments, insecticides
Particulate matter (PM)	Indoor: cooking, wood stoves, chemical transformations Outdoors: industrial processes, vehicles, construction, volcanoes, deserts

Main sources of gaseous pollutants include construction and decoration materials and their treatment or surface finishes, which depend on the building design. Occupants also contribute to their presence indoors by various activities such as cooking, cleaning and using deodorants. Particulate matter either penetrates from outdoors or is produced indoors by activities or chemical transformations. Ventilation is hence an important factor that can eliminate pollutants emitted indoors and potentially introduce outdoor pollutants such as particulate matter.

## 2.2. Emission of pollutants

To evaluate the exposure of occupants to indoor pollutants, their emission from different sources have to be modelled. When specific emission models are not available, emission rates (mass emitted per unit of time) can be obtained in different databases, such as: SOPHIE (Sources of Pollution for a Healthy and comfortable Indoor Environment) (Bluysseri, De Oliveira Fernandes, and Molina 2000) MEDB-IAQ (Material Emission DataBase and Single-Zone IAQ Simulation program) (J. Zhang et al. 2010), BUMA (prioritisation of Building Materials as indoor pollution sources) (Bartzis et al. 2009) and the U.S. Environmental Protection Agency's source-ranking database (US EPA 2015). The PANDORA database (Abadie et Blondeau 2011) aggregates most of the existing databases and studies on indoor gaseous and particulate pollutant emissions, with a total of 370 sources and 5406 emission rates (Abadie and Blondeau 2011) separated into three categories: 1) materials, 2) occupants and their activities and 3) cleaning products and air fresheners.

Emissions related to occupants (activities or direct emissions), are depend on occupant presence and behaviour. Emissions related to materials or furniture are continuous and passive. Both emission types are time-dependent and influenced by external conditions such as temperature and ventilation rates, but the identified databases do not take into account user behaviour or the time-dependency of material emissions. Different models have aimed at representing variations in occupant and material emissions and they are going to be discussed in the next sections.

### 2.2.1. Occupant sources

Humans are often the major source of indoor pollution, with about 2.5 times more particulate matter when the building is occupied (Licina et al. 2016) and contributing to 57% of total VOC emissions in densely-populated environments (Tang et al. 2016).

#### Direct emissions

Occupants emit pollutants from their skin and clothes. The skin can emit about 64 different VOCs, mainly related to the diet, use of perfumes and cosmetics, smoking and metabolic and cutaneous processes that cause the degradation and formation of certain substances (Mochalski et al. 2014). Different substances, including lead, arsenic and pesticides (Diemel et al. 1981; Butte 2004), deposit on clothes, shoes and hair in outdoor environments and are likely to be reemitted indoors (McDonagh and Byrne 2014). Clothes could also be responsible for tetrachloroethylene, 1,1,1-trichloroethane and chloroform emissions due to dry-cleaning or washing machines (Namieśnik et al. 1992). Another occupant source of VOC is exhaled air, with an average of 49 of these compounds detected in exhalation samples from 117 healthy human subjects (Kruza and Carslaw 2019a; Mochalski et al. 2014; Sun, He, and Yang 2017). These emissions are passive since breathing cannot be controlled, but still highly influenced by individual physical characteristics, namely, age groups, gender, smoking habits, level of

activity and even race/ethnicity (Braun 2015; Persily and de Jonge 2017; Sun, He, and Yang 2017). Higher activity levels tend to generate more CO<sub>2</sub> emissions (Persily and de Jonge 2017).

Emissions hence depend on several factors such as individual metabolism, activity level, exposure to outdoor pollutants, garment fabric and laundry practices (Zhao 2013). These direct emissions from skin, clothes or breath create a “personal cloud” effect whereby the air surrounding occupants contain important concentrations of particulate or gaseous pollutants (Licina, Tian, and Nazaroff 2017; Licina et al. 2019; Kruza and Carslaw 2019a; Bekö et al. 2020). Furthermore, occupants can also influence the presence of substances through their intake and different chemical reactions that they can induce (Kruza and Carslaw 2019a). Emissions from the skin and breath of occupants could be of concern to human health, though little knowledge exists on the subject at present.

### Activities

Occupant activities can have considerable effects on IAQ and occupant health, especially when they involve combustion. These activities highly dependent on socio-economic and cultural characteristics. For example, many homes are equipped with cooking or heating wood stoves while these are of high risks for occupants, particularly young children (Smith et al. 2011; Alves et al. 2011; Lekkas 2013; Aberilla et al. 2020). Candles and incense sticks were also identified as a major source of active VOC and particulate emissions (Wallace 2006; Aquilina and Camilleri 2022; Halios et al. 2022), representing 19% of VOC emissions in European buildings. Other fragrant sources of pollutants include air fresheners, cleaning detergents and deodorants (William W. Nazaroff and Weschler 2004; Y. Huang, Ho, Ho, Lee, Gao, et al. 2011; Halios et al. 2022). Particles present in air deposit on surfaces, and are resuspended by different activities such as walking, vacuuming or dancing (Wallace 2006; Corsi et al. 2008; Lewis et al. 2018; Rohadi et al. 2020). According to Wallace (2006), the resuspension rate was of considerable magnitude and concentrations increased by up to 80 µg/m<sup>3</sup> for particles of diameter 2.5 µm to 5 µm.

#### 2.2.2. Material sources

Materials have been identified as important sources of chemicals indoors and their compositions and emissions have been widely studied (Wallace et al. 1987; Yan et al. 2009; Friar and Vittori 2017; L. Huang et al. 2022a). Indoor materials are used for construction, decoration or furnishing. Different materials are used in building construction, depending on the building’s use, aesthetic tastes, local standards, climate, and cultural practices. They generally include bricks, concrete, wood, metal structures and insulation. Indoor surfaces are often made of gypsum board covered with surface finishes such as paint or wallpaper. Flooring types include tiles, linoleum, parquet or carpet while windows and door frames can be made of different material combinations such as metal, wood or plastic. Materials used to make furniture are numerous: metal, wood or wood composites and plastics with finishes such as paint or varnish.

Each material contains several volatile substances (or chemicals) that can be emitted into air. This emission depends on both material and chemical properties. According to the technical specifications of the building, different performances must be met, such as thermal or acoustic insulation, mechanical strength, aesthetics, water-resistance, transparency or opacity. These properties are met by combining materials and substances together and the complexity of material types and their compositions influence emission dynamics. Measured emission can be found in different studies and databases (Abadie and Blondeau 2011; Maupetit et al. 2017) or directly from material manufacturers, in particular due to increasing concern regarding chemical release of materials. This has encouraged the creation of regulations and certifications, such as regulatory labelling scheme in France which requires manufacturers to measure VOC emissions of construction, wall and flooring materials (decree of 19 april, 2011) or material certifications (e.g. Emissioncode®, Indoor Air Comfort Gold or Blue Angel).

Material emission data are obtained from chamber measurements of VOC concentrations in air. Materials are placed in chambers with controlled ventilation rates and temperature with only one face emitting substances, the other being covered with an impermeable material. Chamber walls are made of very low absorption or emission materials such as glass or stainless steel, in order not to interfere with the test sample. The limit of these data is that they are valid for the given chamber conditions and not occupied buildings. Furthermore, they are only available and valid at a few specific points in time, during a restricted timespan of a few days to a few weeks, though emissions are time-dependent. Indeed, emission rates generally increase to a peak before decreasing gradually since the amount of substance present inside the material decreases. For instance, it was found that formaldehyde concentrations were higher in newer buildings (Brown et al. 2015) and that the entirety of ethylbenzene from a 90 m<sup>2</sup> dwelling was released within a year (Micolier 2019). Furthermore, peak emissions can occur at an early or later stage, depending on the chemical and material properties, external conditions such as temperature and pressure, and the non-uniformity in the presence of substances inside the material (Haghighat and De Bellis 1998; Meininghaus, Gunnarsen, and Knudsen 2000).

The presence of different material layers can also influence the emission since outer layers act as a buffer to emissions from deeper layers. Substances are sorbed into the outer levels, before eventually being emitted by the external layer on a larger time frame. For instance, it was found in a particular study that substances from lower layers of linoleum and PVC floorings were not emitted at all during the 204-days study (Wilke, Jann, and Brödner 2004).

Emission models were developed to predict time-dependent material VOC emissions and some consider the effect of different layers of materials on emission rates (Haghighat and Huang 2003; L. Z. Zhang and Niu 2004; Yan, Zhang, and Wang 2009; Yuan Yao et al. 2011; Guo 2013; Xiong et al. 2019). These models are based on the principle of mass-balance, both influenced by external parameters (such as ventilation rate, intake by occupants, temperature, or relative humidity) and properties specific to the chemical and the material. These properties include the diffusion coefficient of the material ( $D_m$ ) which



describes the movement of volatile substances through the material due to difference in concentrations and the material-air partition coefficient ( $K_{ma}$ ) that is the ratio of concentrations of the substance in the material to that in air. In other words,  $K_{ma}$  describes the partitioning effect between the material and air. These two coefficients depend on both material and chemical properties and are unique to each material-chemical combination.

Coefficients have been computed from large datasets and are available in literature (L. Huang, Fantke, et al. 2017; L. Huang and Jolliet 2019). Another important factor is the composition of the material since the initial mass of substance inside the material influences the emission profile. Some material compositions can be found in the Pharos database (Friar and Vittori 2017). Instead of using single material-chemical specific coefficients and masses, an approach to treat multi-layered material emissions consists in modelling single initial concentration and material-air partition coefficient for all layers based on the properties of the different materials (Yuan Yao et al. 2011) while another one derives dimensionless correlation factors (Xiong et al. 2019). However, the number of derived coefficients are limited and do not allow to account for variations in building material compositions.

### 2.3. Exposure concentrations

Once pollutants are emitted indoors, occupants are exposed to them through different pathways: inhalation, ingestion, dermal contact and gaseous dermal uptake. The intake of pollutants can primarily occur from inhalation or gaseous dermal uptake (Micolier 2019), since these pathways are passive and do not necessitate direct contact, unlike the other two pathways. They both depend on the concentration of pollutants in indoor air, influenced by emission and evacuation.

Pollutant evacuation from air is driven by ventilation and deposition, but also includes cleaning, filtration in some cases, or intake by occupants. The ventilation of a room causes air renewal and the dilution of pollutants into larger volumes of air from outdoors. Models that calculate indoor air pollutant concentrations are based on mass balance: the mass of emitted pollutants equals to the sum of the masses of pollutants evacuated and remaining in indoor air. Based on the different evacuation routes described by Fantke et al. (2017), the daily average concentration of a pollutant in a room is expressed in equation (1-1). Additional routes can be considered, especially for particles: deposition rate and filtration of recirculated air (Fantke et al. 2017).

$$C_{in} = C_{out} + \frac{ER_{activity}}{V_{room} \times \left(ACH + \frac{POP \times BR}{V_{room}}\right)} \quad (1-1)$$

Where  $C_{in}$  is the indoor pollutant concentration ( $\mu\text{g}/\text{m}^3$ ),  $C_{out}$  is its outdoor concentration ( $\mu\text{g}/\text{m}^3$ ),  $ER_{activity}$  is the emission rate of the activity ( $\mu\text{g}/\text{h}$ ),  $V_{room}$  is the room volume ( $\text{m}^3$ ),  $ACH$  is the air change rate (1/h),  $POP$  is the number of occupants (-) and  $BR$  is the average breathing rate of an occupant ( $\text{m}^3/\text{h}$ ). This parametric model allows us to identify some key parameters

determining occupant exposure to pollutants. Along with these parameters, indoor air chemistry can also play an important role in the presence of IAPs (Hodas et al. 2016). These parameters are going to be discussed in the next sections (except for emission which has already been discussed previously).

### 2.3.1. Ventilation, outdoor pollution, and filters

Ventilation in buildings can be mechanical or natural. With mechanical ventilation, there can be either or both supply fans driving air into the building and exhaust fans driving air out of the building. Since the fan speed can be modified, ventilation rates can be controlled. Natural ventilation includes air inlets, windows, doors, and infiltrations through cracks, leaks or closed window and door frames. There also exist mix-mode ventilations that combine natural and mechanical ventilations. A high variability can be observed in air change rates, *ACH* (vol/h) based on building characteristics (e.g. air tightness or size and positions of openings), meteorological and climatic conditions, noise and insulation (affecting indoor temperatures and hence occupant behaviour regarding openings). Occupant sensitivity to several of these parameters such as noise and temperature influences the time and duration of window and door opening and hence influence ventilation rates. Measured ventilation rates reported in literature vary from 0.2 vol/h in highly airtight buildings to 23 vol/h in the West region of the US (US EPA 2011; Rosenbaum et al. 2015). Ventilation types also vary widely across different geographic locations. For example, 100% of houses are naturally ventilated in Bulgaria, compared to 52% in Finland (Litiu 2012). Furthermore, the French regulation on air renewal defines minimum ventilation rates according to building use and the number of rooms (Légifrance 1982).

High ventilation rates can help to evacuate pollutants, but also introduce outdoor pollutants indoors. In the case of particulate matter, the concentrations can be high in dense cities or close to a desert or construction site. In very polluted cities, especially large cities with populations above 3 million, natural ventilation could simply increase exposure to particulate matter (Apte et al. 2012). When mechanical ventilation is available, filters can be installed to reduce the particulate matter content in air penetrating from outdoors or circulating from other rooms if recirculation is possible. Filters are classified into different categories defined by EN779 for the European Union and ASHRAE 52.2 in the US according to their efficiency and the particle size that they can filter. VOCs can also be filtered from air using carbon filters (Khararoodi, Haghghat, and Lee 2023) or by gas-to-particle transformation through air cleaners (Krugly et al. 2022).

Different types of models can be used to simulate air flows, and hence pollutant concentrations. For mechanical ventilation, fixed rates or scenarios can be modelled, while rates are highly variable with natural ventilation. Natural air flows depend on pressure differences, as described by Bernoulli's principle, and temperature differences, creating thermal draft. They are evaluated by different types of models:

- single-compartment models (monozone, represented by one well-mixed node) which can be represented by a simple mass-balance with relevant parameters as in equation (1-1)
- multiple-compartment models (multizone, each zone represented by one well-mixed node)
- zonal models (different zones inside one compartment)
- computational fluid dynamics (CFD, based on Navier-Stokes equations of mass, energy, and momentum conservation) where the fluid volume under study is represented by a mesh.

Given that buildings generally contain different rooms separated by walls, single-compartment models can only provide gross estimates, but can be useful for quick estimates and to test more complex models' results. CFD simulations have the highest precision and can be used to study microenvironments or complex building geometries. For example, higher CO infiltrations into houses from gas furnaces were found with CFD simulation where the precise location of the furnace was indicated, as compared to multizone modelling where there was no defined location (Szczepanik and Scislo 2021). Another study shows that vertical variations influence the measured concentration of particles indoors (Ainiwaer et al. 2022). However, higher precision implies greater simulation times (Tan and Glicksman 2005), especially for CFD and often require expert user input (Q. Chen 2009). Nodal modelling has similar advantages and disadvantages, with generally less dense meshing (Q. Chen 2009).

Being adapted to low air speeds in relatively large volumes of air, multizone models can be used for buildings. It was reported that air in a room was well-mixed three minutes after the injection of a substance (Thatcher et al. 2002). A higher degree of precision can be achieved by considering interactions between zones: infiltrations and openings can transfer pollutants from one zone to another and materials at the interface of two zones, such as partitions, can emit substances in two rooms simultaneously. Adequate emission models need to be used to consider emissions from two faces of the same material (Hu et al. 2007; Yan, Zhang, and Wang 2009; Guo 2013). Multizone models include COMIS, CONTAM, MZAP, BREEZE and COMFIE. The five of them showed good concordance for same input parameters in a case study (Trocmé 2009).

The best trade-off between time and precision has to be selected according to the use intended. Multizone models could provide the desired level of precision according to the scope of this study and, due to their lower complexity and time consumption, encourage broader use of ecodesign tools.

### 2.3.2. Particle deposition

Particle deposition on surfaces is affected by gravitational attraction, interception, impaction and Brownian motion. Gravitation, interception and impaction have greater effects on heavier particles, whereas lighter ones are mostly affected by Brownian diffusion.

The deposition on horizontal surfaces is caused by gravitation and Brownian diffusion while Fick's law and turbophoresis are responsible for the deposition on vertical surfaces (K. Lai and Nazaroff

2000). Models and observations conclude that deposition is highly affected by the area available for deposition, room volume, particle diameter and the speed of particles (K. Lai and Nazaroff 2000; Thatcher et al. 2002). For instance, a 1.3 to 2.4 times higher deposition was observed in a 35 m<sup>2</sup> room with the presence of furniture, increasing the area available for deposition by 12 m<sup>2</sup> (Thatcher et al. 2002). A higher increase in deposition was observed for smaller particles since heavier particles were readily deposited in the unfurnished room by gravitational sedimentation. Figure 1-4 shows the effect of particle size on the rate of deposition for different levels of furnishing and different air speeds.

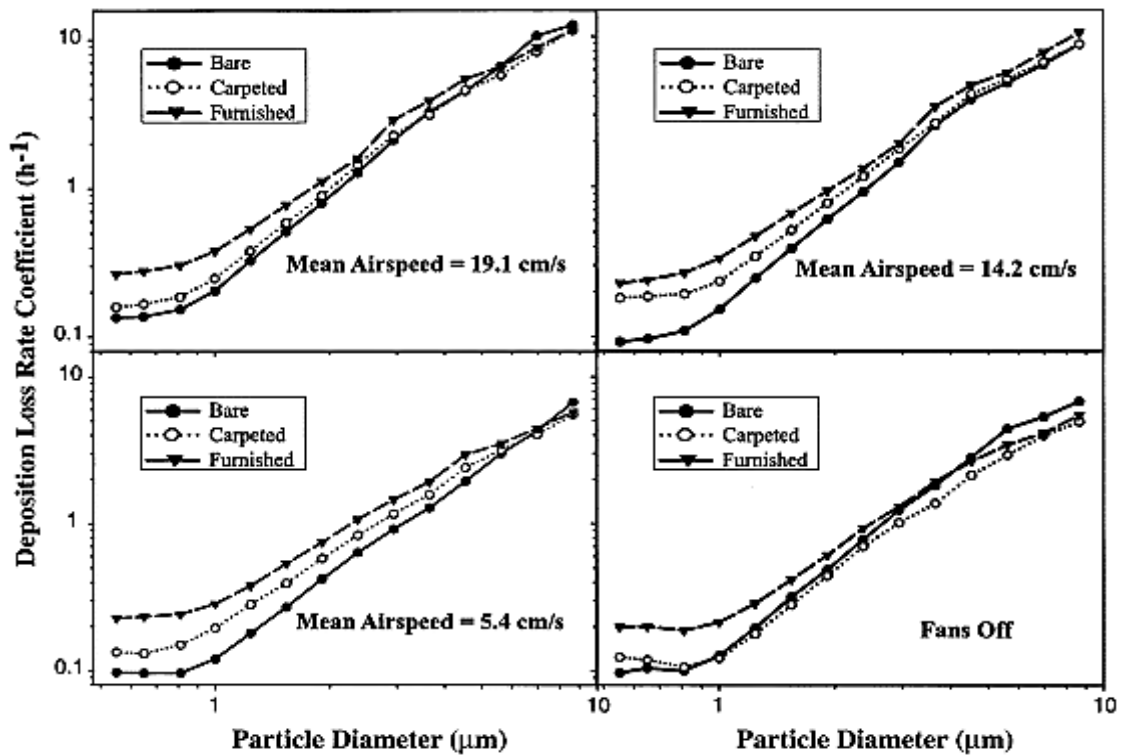


Figure 1-4: Particle deposition loss-rate coefficients as a function of particle size at each mean core air speed for three furnishing levels (Thatcher et al. 2002)

For smaller particles with diameters in the range of 0.1 to 1 μm, a deviation between calculated and experimental results were found since, at these diameters, the shape affects deposition and the particles can no longer be approximated as spheres (K. Lai et Nazaroff 2000).

### 2.3.3. Indoor air chemistry

Particles and gases present indoors (penetrating from outdoors or emitted indoors) undergo phase changes and chemical reactions that can break down existing substances, generate new ones or coagulate to form larger particles. These reactions are named “indoor air chemistry”, which differentiate from reactions occurring outdoors due to lower temperature fluctuations and direct sunlight, the absence of rain, and much higher surface-to-volume ratios and concentrations of organic compounds (Weschler and Carslaw 2018). Main outdoor pollutants include ozone (O<sub>3</sub>) and nitrogen oxides (NO<sub>x</sub>) which can react with VOCs to produce new radicals. For instance, five of the VOCs emitted by an air freshener

were degraded and caused the formation of particulate matter (secondary organic aerosols, SOA) in the presence of O<sub>3</sub> (Liu et al. 2004). The presence of secondary pollutants are directly linked to some symptoms (Nazaroff and Weschler 2004; Weschler 2004; N. Carslaw and Wolkoff 2006) and produce dangerous substances such as formaldehyde and acetaldehyde (Liu et al. 2004; Mendez et al. 2015).

Models have been developed to simulate these reactions (Sørensen and Weschler 2002; Nicola Carslaw 2007; Carter 2010; Mendez et al. 2015). For instance, the INCA Indoor model (Mendez et al. 2015) which is based on the SAPRC-07 model (Carter 2010) takes into account the different physical and chemical reactions, including photolysis, coagulation, condensation and nucleation processes. The following is a brief description of the main indoor air reactions:

- Photolysis is the degradation of a substance into other chemicals with the action of light.
- Coagulation occurs when particles of different diameters and species are attracted to each other to form a mass of different substances, guided by Brownian diffusion for low velocities, corresponding to indoor air.
- Condensation occurs when vapour condenses on, or evaporates from, a particle, making it change in size, respectively becoming bigger or smaller. It is hence the change between gaseous and liquid states and depends on external conditions such as pressure and temperature.
- Nucleation is the formation of new particles of diameter of the order of nanometers, with a new physicochemical structure. In the case of air, particles are produced by gas-to-particle conversion. It affects the total particle number and size distribution (Pratsinis 1988; Curtius 2006).

However, discrepancies still remain between some predictions and measurements since indoor air chemistry remains “important, ubiquitous, and complex” (Charles J. Weschler and Carslaw 2018).

#### 2.3.4. Occupant behaviour

Despite good knowledge of all physical parameters and robust models to calculate indoor concentrations and intake of pollutants, the human factor can be unpredictable and introduce high levels of uncertainty. Being in interaction with the indoor environment, occupants can considerably influence IAQ. Different factors influence the way humans are affected by pollutants, classified as *intraindividual* (variability within a person, for example reactions to the intake of pollutants) or *interindividual* (variability due to use patterns or housing characteristics) (Kvasnicka et al. 2020), while some factors can be classified as both. Intraindividual variabilities are linked to age, sex, race/ethnicity, levels of activity (Braun 2015; Persily and de Jonge 2017) but also their sensitivity to odours and temperatures which can affect the frequency of window opening and hence the evacuation or introduction of pollutants. Occupant habits, related to housing characteristics (interindividual) or personal taste (intraindividual), induce different levels or frequencies of emission and evacuation such as cooking with

or without a kitchen hood, lighting of candles, cleaning, and dusting. The frequency of hand-washing or showering can also affect the exposure to pollutants, in particular SVOCs (Kvasnicka et al. 2020).

Several studies have focused on occupant behaviour (Haldi and Robinson 2011; Bonte et al. 2013; Vorger, Schalbart, and Peuportier 2014; O'Brien and Gunay 2014; Schalbart, Vorger, and Peuportier 2021), and on the link between socioeconomic and lifestyle factors and IAQ (Brown et al. 2015). For instance, the income level, the number of occupants and the presence of a garage were linked to BTEX (benzene, toluene, ethylbenzene and xylene) emissions (Brown et al. 2015).

Occupant behaviour modelling hence seems crucial to precisely model their exposure to pollutants. Two main types of models have been identified: stochastic models and agent-based models (ABM). Stochastic models use a probabilistic approach to determine activity scenarios. Using activity timetables, the probability of engaging in a certain activity at a given time can be calculated in order to determine likely actions at a given time. Socio-demographic characteristics have also been used, coupled with activity scenarios from the “*Enquête Emploi du Temps*” of the *Institut National de la Statistique et des Etudes Economiques* (INSEE) in France, to determine main parameters affecting actions related to occupant comfort, such as the opening of windows (Vorger, Schalbart, and Peuportier 2014; Schalbart, Vorger, and Peuportier 2021). The main parameters identified were household size, use of solar protection, exterior temperature, ventilation and age of the occupant.

ABM is based on artificial intelligence to simulate agent (occupant) behaviour. The advantages of ABM are mostly related to the ability to simulate a large number of agents in shorter time periods, with the possibility of integrating feedback loops (for e.g. changing electricity consumption for a particular month based on electricity bills at the end of the previous month) (Micolier 2019). However, models are complex and less relevant for simple models with few agents, such as building occupant behaviour. Furthermore, the models have to be validated by field data and the validation process can be time-consuming and costly.

## 2.4. Health damages

As presented in the pollutant pathway, once substances are emitted in air, they are taken in by occupants. Health impacts are thus dependent on the exposure of occupants to these pollutants and the toxicity of the substances. Most of the intake takes place indoors, even for outdoor pollution since people spend the highest fraction of their time in indoor environments: intake fractions are 2 to 5 orders of magnitude higher indoors (Fantke et al. 2017). The different substances present in indoor air can lead to several health complications. For instance, toxic substances such as VOCs are mutagenic, genotoxic, neurotoxic and carcinogenic, and linked to respiratory illnesses, leukaemia, birth defects, neurocognitive impairment and cancer (A. J. Li, Pal, and Kannan 2021). Chemical/toxicological profiles are created for different chemicals in order to characterise their toxicity and the adverse health effects on humans and

biodiversity, such as PHAROS (Healthy Building Network 2000) and ToxValDB (US EPA and Richard 2018).

Other harmful gases include CO, which has 200 times higher affinity than O<sub>2</sub>, hence attaching itself to haemoglobin instead of oxygen and leading to asphyxiation and NO<sub>2</sub>, retained in the lungs upon inhalation and dissolving into the tissues, depositing in the alveoli, leading to cardiopulmonary effects, respiratory problems and intensified allergic responses, mainly for children.

As for particulate matter, it has been estimated that PM<sub>2.5</sub> was the fifth highest global mortality risk factor, being responsible for 4.2 million deaths in 2015 according to the Global Burden of Diseases (GBD) study (Cohen et al. 2017a). The GBD study aims at interpreting observational health data and attributing them to different risk factors, thus estimating the number of years lived with disabilities and years of life lost due to different diseases linked to each risk factor, namely ambient and household PM<sub>2.5</sub> pollution (including high exposure to PM from active and passive cigarette smoke and indoor combustion for heating or cooking). PM<sub>2.5</sub> concentrations were modelled using a global atmospheric chemistry transport model, linked to ground-based PM<sub>2.5</sub> measures in this study. Once inhaled, particles deposit on the respiratory tracts and can lead to cardio-pulmonary complications. Diseases that are partially attributed to the intake of PM<sub>2.5</sub> are: (COPD), ischemic heart disease (IHD), acute lower respiratory diseases (ALRI), cerebrovascular (stroke) and lung cancer (LC) (Burnett et al. 2014).

World Health Organization's (WHO) guideline indicates that yearly annual exposures should not exceed 5 µg.m<sup>-3</sup> and 24-hour averages should not exceed 15 µg.m<sup>-3</sup> more than 3 to 4 days per year (WHO 2021), however, indoor PM<sub>2.5</sub> concentrations were found to be higher than the guideline values in 96% of French schools (Mandin 2020) and in several studies related to daily indoor activities (Ferro, Kopperud, and Hildemann 2004; Corsi, Siegel, and Chiang 2008; Pagels et al. 2009; Buonanno, Morawska, and Stabile 2009).

The next sections present the different possible intake pathways, the parameters defining intake quantities and health impact calculation methods.

#### 2.4.1. Exposure and intake of pollutants

The different pathways through which occupants can assimilate pollutants are inhalation, direct dermal contact, gaseous dermal uptake and ingestion. Inhalation was found to be the main exposure pathway in buildings, followed by dermal intake, while dust ingestion was negligible (Huang et al. 2022), but are particularly relevant for young children (Kvasnicka et al. 2020) who might have more frequent hand-to-mouth contact. The exposure factor,  $XF$  (kg<sub>intake</sub>/s per kg<sub>in compartment</sub>) has been derived for each pathway by (Micolier 2019) based on L. Huang et al. (2017) and Rosenbaum et al. (2007): inhalation *inh*, gaseous dermal uptake *derm,gas*, ingestion *ing* and dermal contact *derm,cont* and are presented in the equations (1-2), (1-3), (1-4) and (1-5):

$$XF_{inh} = \frac{POP_{ad} \cdot BR_{ad} \cdot f_{time,ad} + POP_{ch} \cdot BR_{ch} \cdot f_{time,ch}}{V_{compartment}} \quad (1-2)$$

$$XF_{derm,gas} = \frac{K_{p,gas}(t)(POP_{ad}A_{skin,gas,ad}f_{time,ad} + POP_{ch}A_{skin,gas,ch}f_{time,ch})}{V_{compartment}} \quad (1-3)$$

$$XF_{ing} = \frac{\frac{f_{dust}}{K_{md}(t)\rho_{dust}}(POP_{ad} \cdot ingR_{ad} \cdot f_{time,ad} + POP_{ch} \cdot ingR_{ch} \cdot f_{time,ch})}{V_{compartment}} \quad (1-4)$$

$$XF_{derm,cont} = \frac{\frac{K_{p,aq}(t)}{K_{mw}(t)}(POP_{ad} \cdot FQ_{ad} \cdot f_{time,ad}A_{contact,ad} + POP_{ch} \cdot FQ_{ch} \cdot f_{time,ch}A_{contact,ch})}{V_{compartment}} \quad (1-5)$$

$POP$  (-) represents the number of occupants,  $BR$  ( $m^3/s$ ) the inhalation rate,  $ingR$  ( $kg/s$ ) the ingestion rate,  $V_{compartment}$  ( $m^3$ ) the volume of the compartment,  $A_{skin,gas}$  ( $m^2$ ) the skin gaseous uptake surface,  $A_{contact}$  ( $m^2$ ) the skin surface in contact with the floor,  $f_{time}$  (-) the fraction of time spent indoor,  $FQ$  ( $s^{-1}$ ) the frequency of dermal contact with the floor and the subscripts  $ch$  and  $ad$  stand for the children and the adults respectively.  $f_{dust}$  (-) is the fraction of ingested dust that is from the considered building material,  $K_{md}$  (-) the material dust partition coefficient,  $\rho_{dust}$  ( $kg/m^3$ ) the dust density,  $K_{p,gas}$  ( $m/s$ ) gaseous-skin permeation coefficient,  $K_{p,aq}$  ( $m/s$ ) the skin permeation coefficient via aqueous solution,  $K_{mw}$  (-) the material water partition coefficient. The exposure rate, which is the volume to which a population is exposed through the different pathways per unit of time, can reflect different building uses with parameters  $POP$ ,  $f_{time}$  and  $FQ$ , differentiated for adults and children.

With the  $XF$ , we can evaluate the product intake fraction  $iF$  ( $kg_{in}/kg_{emitted}$ ), which is the fraction of emitted substance that is assimilated (Jolliet et al. 2015), using the fate factor  $FF$  ( $kg_{in\ compartment}/kg_{emitted}$ ) (Fantke et al. 2017). It can be calculated for known intake and emission rates, but also has the advantage of being emission-independent, only defining the competition between exposure factors and pollutant removal rates, as represented by equation (1-6).

$$iF = \frac{iR}{ER} = XF \times FF \quad (1-6)$$

$ER$  represents the emission rate ( $\mu g/s$ ) and  $iR$  the intake rate ( $\mu g/s$ ).  $XF$  is dependent on population characteristics such as breathing rates and  $FF$  is dependent on compartment characteristics, such as air renewal rate and deposition rate in the case of indoor environments.



In the case of inhalation, a distinction can be made between intake fraction and uptake fraction ( $uF$ ), defined by Humbert (2009) as the part of the substance that penetrates the different regions of the breathing tract and represented by equation (1-7).

$$uF = \frac{\sum_{people,time} \text{uptake of pollutant by an individual}}{\text{mass released into the environment}} = f_{dep,tot} \times iF \quad (1-7)$$

The total fraction of deposited particles is represented by  $f_{dep,tot}$ . The respiratory tract is responsible for the inhalation, filtration and conditioning of air that goes into the lungs. Air, and the substances it contains, is partially absorbed and the rest is exhaled. The particles of a certain diameter can be filtered in the respiratory tract. The bigger ones are filtered earlier on in the tract, while the smaller ones, mainly gaseous particles, can be absorbed.

The respiratory tract is presented in Figure 1-5, where it is separated into *generations* – sections of the respiratory tract divided from the trachea to the last order of terminal bronchioles. Air is inhaled through the nose or mouth, then goes to the pharynx (throat), the larynx (voice box) and the trachea where dirt and dust are trapped (generation 0). The next generations conduct air towards the gas exchange regions (generations 22 and 23), where gases are taken in by the body further down the tract, mainly through alveoli.

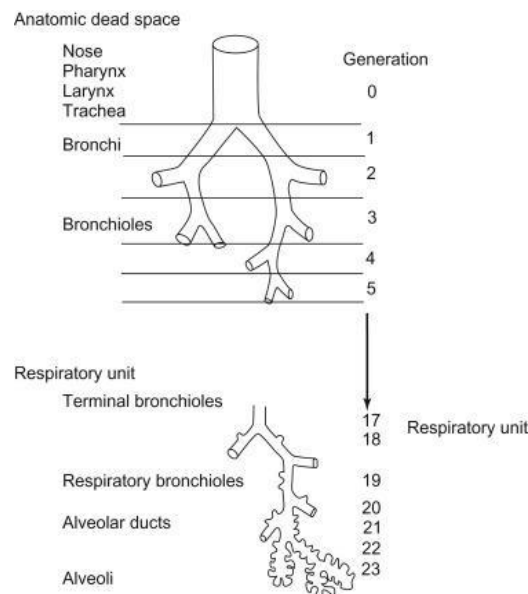


Figure 1-5: Diagram of the respiratory tract (Feher 2017)

Once intake or uptake fractions have been calculated, the total intake can be calculated as a function of the concentration and duration of exposure. The health effects can be obtained using the total intake and effect factors.

### 2.4.2. Effect factors and health damage

Based on dose-response curves representing the response to a certain amount of substance in the body, effect factors have been derived. In current LCA methods, characterisation factors,  $CF$  (DALY/kg<sub>emitted</sub>), are more commonly used. They are the product of the intake fraction (kg<sub>in</sub>/kg<sub>emitted</sub>) and the effect factor  $EF$  (DALY/kg<sub>in</sub>) and represent the health impacts per unit intake of the substance. Two models will be discussed, treating 1) chemical toxicity and 2) mortality due to particulate matter. The models allow to derive effect factors with the same indicator as LCA health damage endpoint categories: the DALY (disability-adjusted life years). The QALY (quality-adjusted life years) is another metric that can be added to define the quality of life of occupants by integrating the well-being of occupants but is not commonly used in LCA. The total impact score  $IS$  (DALY) is calculated by the product of the effect factor (DALY/kg<sub>in</sub>) and the cumulative mass intake  $M_{in}$  (kg<sub>in</sub>).

$$IS = EF \times M_{in} \quad (1-8)$$

#### Chemical toxicity

The underlying model behind human toxicity characterisation factors for Impact World+ LCA method and chemical data for ReCiPe is USEtox (Fantke et al. 2017). Under the aegis of the UNEP/SETAC Life Cycle Initiative, USEtox has been developed with the aim of harmonising different characterisation models in LCA. The health impacts on humans are represented by the potential cases of cancer or non-cancer diseases per unit intake, which lead to the end-point DALY unit with the following relationships: one cancer case corresponds to 11.5 DALY, while one non-cancer case corresponds to 2.7 DALY (M. Huijbregts, Rombouts, and Ragas 2004; M. A. J. Huijbregts et al. 2005). Effect factors are considered to be linear. Coupled with concentration models, the USEtox model allows to evaluate the exposure and intake of toxic chemicals, in particular VOCs, as a function of occupant and building characteristics.

#### Particulate matter health damage

Health effects due to PM and particularly PM<sub>2.5</sub> are calculated by several existing LCIA methods. However, following the Basel Guidance Workshop guidelines (Fantke et al. 2015), approaches should involve higher degrees of spatialisation. Furthermore, current methods consider exposure-response curves to be linear, while different studies have concluded otherwise (Pope, Ezzati, and Dockery 2009; H. Chen et al. 2013; Stafoggia et al. 2013; Nasari et al. 2016), especially at very high concentrations where linear models give unlikely response. For example, smokers inhale large quantities of PM but do not always die young, indicating the body's ability to avoid or repair smoking-related damage (Levine and Crimmins 2014). An integrated exposure-response model has been derived based on data from studies on ambient air pollution, exposure to solid household cooking fuel and active/passive smoking to evaluate the fraction of mortality that can be attributed to PM<sub>2.5</sub>. The model

relies on mortality rates of the five identified diseases: COPD, IHD, ALRI, stroke and LC, as well as the ambient PM<sub>2.5</sub> concentration of the country or region under study. These data can be obtained from the Global Burden of Diseases study (GBD Global Burden of Disease Collaborative Network 2019) which aims at attributing deaths to 87 risk factors. Ambient particulate matter and household air pollution are among the ten highest contributors.

One limit of the model is that we consider average compositions of PM, while these can vary considerably according to their sources and contain different elements or compounds, including heavy metals, fatty acids (especially during cooking) and carbon (Kleeman, Schauer, and Cass 2000; Pagels et al. 2009; Abdullahi, Delgado-Saborit, and Harrison 2013; Rohra et al. 2018; Mainka and Fantke 2022). Furthermore, mortality data is based on outdoor ambient pollution, while occupants are mostly exposed to indoor concentrations.

### 3. Integration of IAQ into LCA for building design

#### 3.1. Optimal solutions and trade-offs

As seen previously, with a life cycle perspective, design choices that allow to reduce IAQ impacts can, on the contrary, increase impacts of other contributors over the life cycle of a building. For example, increasing ventilation rates to evacuate pollutants can lead to higher impacts related to heating. Micolier (2019) considers the whole pollutant pathway from material chemical emissions to their impacts on health, coupled with health damage related to the building's LCA. Particular attention is paid to insulation thickness (thicker insulation can lower energy need but emit more VOCs) and trade-offs between ventilation and energy use for heating. Furthermore, depending on the heat source, increased heating needs can also lead to more IAQ-related impacts, in particular for wood stoves since more combustion is required, leading to more emissions. Higher heat consumption from increased ventilation can be compensated by implementing more insulation, but, besides impacts on the construction stage, health damages on the operation stage can also increase due to higher VOC emissions from thicker material layers. Thus, the integration of IAQ impacts to building LCA can help to identify solutions that give the best trade-off, i.e. solutions that allow to lower impacts over the building's whole life cycle.

#### 3.2. Tools that treat both LCA and IAQ assessment

Different models treating the pollutant pathway exist in literature, but, in order to be considered in building projects, tools that facilitate decision making and that are user-friendly can be proposed. Some programs allow for the simulation of energy needs, the calculation of environmental impacts with LCA (One Click LCA, Tally, ClimaWin 2020, ArchiWIZARD RE2020, THERMBIM), and the simulation of indoor air concentrations (CONTAM; IA-QUEST). A review of some tools that have aimed at integrating IAQ to other stages of the building life cycle into a single indicator are discussed. Only few tools allow to treat both IAQ and LCA. They are either rating schemes or simulations. A

particular software that allows to for energy, LCA and IAQ calculations on the same interface is introduced: Pleiades.

### 3.2.1. Building ratings

The Swedish environmental tool (Malmqvist et al. 2011) uses indicators covering air quality, energy use and noise levels to allocate “Gold”, “Silver” or “Bronze” ratings to buildings. LEED (Leadership in Energy and Environmental Design) and BREEAM (Building Research Establishment Environmental Assessment Method) are similar tools that give ratings to buildings. The disadvantage of such tools lies in their qualitative nature and the absence of focus on sources of concern. Though simple to understand, these ratings cannot be used for eco-design where multi-criteria assessments are necessary.

### 3.2.2. Pleiades

The Pleiades software developed by IZUBA Energies integrates three important modules for the integration of IAQ to building LCA: thermal simulation (COMFIE), building LCA (EQUER) and IAQ (INDALO).

#### COMFIE

The COMFIE model was developed by the Center for Energy Efficiency of Systems (CES) at Mines Paris PSL (Peuportier and Blanc Sommereux 1990), for the dynamic thermal simulation of the building envelope. Pleiades allows to create the model with a Building Information Modelling (BIM) module. Based on 1) building characteristics (shape, orientation, materials and openings), 2) equipment (air inlets or mechanical ventilation), 3) occupants (comfort temperatures, occupation scenarios), and 4) location (defining meteorological data), the energy demands of the building can be simulated. The occupancy and ventilation scenarios of each zone (or room) has to be defined. The opening scenarios of doors and windows can also be indicated and the time step for the scenarios set to one hour (if a window is open for a few minutes, the scenario can only consider an average hourly opening rate). The ventilation system of the building has to be defined and the units of the different zones have to be assigned (inlets and outlets). Pollutant-controlled ventilation can be installed, namely humidity, CO<sub>2</sub> and total volatile organic compounds (TVOCs).

#### EQUER

The dynamic simulation parameters (materials used and relevant quantities) and results (energy needs) are coupled with the ecoinvent database and LCA methods, for instance Impact World+ (Bulle et al. 2019) and ReCiPe2016 (M. A. J. Huijbregts et al. 2017) to calculate the building’s environmental impacts. The duration of use and occupant-related parameters can be defined, such as hot and cold water use, daily transport and waste production. Heat and electricity mixes can be modified and additional equipment such as solar panels, ventilation systems or furnaces can be included, together with their

expected replacement frequency. Results are obtained on all indicators considered by the LCA methods, differentiated by life cycle stages, zones or elements of the building (for e.g. foundation, façade, roof and equipment).

### INDALO

INDALO is the IAQ module linked to Pleiades. INCA-Indoor (Mendez et al. 2015), its underlying model, allows to simulate indoor air concentrations of a large number of pollutants from indoor or outdoor sources, taking into account the physiochemical phenomena of over 900 chemical species and 650 VOCs, as well as CO<sub>2</sub>, O<sub>3</sub>, NO<sub>x</sub> and particulate matter. Information on the building's design, ventilation and occupancy is collected from the BIM model. The deposition of particles is also considered based on their size, available surface areas and air speed.

Internal pollutant sources include occupants for CO<sub>2</sub> and humidity, windows, doors and surface finishes (such as paint, carpet or parquet floor) for VOCs and pollutants from outdoor sources such as PM, O<sub>3</sub> and NO<sub>2</sub> enter from unfiltered natural or mechanical ventilation. Outdoor pollution and meteorological data are collected based on the geographical location of the building. The age of the materials (1 month or >1 year) has to be indicated, as well as the week of the year during which the simulation takes place. The age affects the emission rates of the pollutants and the week number affects meteorological conditions and outdoor pollution levels.

The calculation is carried out using CONTAM (Dols and Polidoro 2018) model by taking into account several factors, namely:

- VOC emission rates that are collected from the PANDORA database,
- CO<sub>2</sub> and H<sub>2</sub>O emissions from occupants, defined in INCA-Indoor,
- Air renewal rate defined by the user for mechanical ventilation or, for natural ventilation, calculated from meteorological conditions, infiltration rates and sizes and orientations of openings,
- Dispersion of pollutants transported by these airflows from indoors to outdoors and vice versa,
- Adsorption and desorption of the pollutants to and from building materials, filtration, and deposition to building surfaces,
- Chemical reactions that occur between chemicals.

Results indicate dynamic indoor air concentrations of pollutants, including CO<sub>2</sub>, VOCs, PM<sub>10</sub>, PM<sub>2.5</sub> and PM<sub>1</sub> and TVOC over a duration of one week.

## 4. Conclusion

This chapter has summarised important steps in the calculation of IAQ impacts on occupant health so that they can be integrated into building LCA. For this, two methods can be developed. The

first one is inventory-based, integrating IAQ emissions in LCA inventories coupled with the IAQ impact pathway in order to derive characterisation factors (similar to current LCA methods). The second method is simulation based, where specific case studies are simulated to obtain time-dependent concentration curves, hence reflecting the dynamic nature of building occupancy.

Existing models and previous work on the emission-impact pathway, important for both model categories, have been discussed. The main observation is that most studies focus on part or parts of the pollutant pathway. One study treats the whole pathway, but only for VOC emissions from insulation materials. However, from this literature review, we have noted that that materials, activities and occupants are important sources of indoor pollutants and have to be included when calculating IAQ impacts. We have also identified two main pollutant categories that are commonly present indoors, and that have quantified effects on health: VOCs and PM<sub>2.5</sub>.

Besides emission rates, air concentrations are affected by ventilation, driven by several parameters such as the presence of mechanical ventilation, size and frequency of openings, meteorological conditions, deposition on surfaces, intake and indoor air chemistry. These parameters, which can vary throughout the duration of the study, should also be included in IAQ evaluations by considering their dynamic nature. Considering the level of precision required, multizone models are most suitable for this study to evaluate indoor air concentrations since they present the best trade-off between desired level of precision and simulation times and have the advantage of being relatively less complex to use, as compared to monozone models and computational fluid dynamics models.

In order to reach the objective of this thesis, i.e. to integrate indoor air quality to the life cycle assessment of buildings, a link between different existing models identified in this chapter and, in some cases, their adaptation to indoor contexts are required. Given the large number of models and parameters, only relevant ones will be chosen. They are presented in the next section in [Figure 1-6](#). [Figure 1-7](#) presents the two types of methods identified for the integration of IAQ to LCA: inventory-based and simulation-based.

## 4.1. Methods overview

### 4.1.1. Models

Based on this literature review, the models that were identified at each step of the pollutant pathway are summarised in [Figure 1-6](#). These models will be adapted to the context of this study, i.e. indoor pollutant health impacts (calculated in DALYs, end-point indicator in LCA), and linked to follow the pathway.

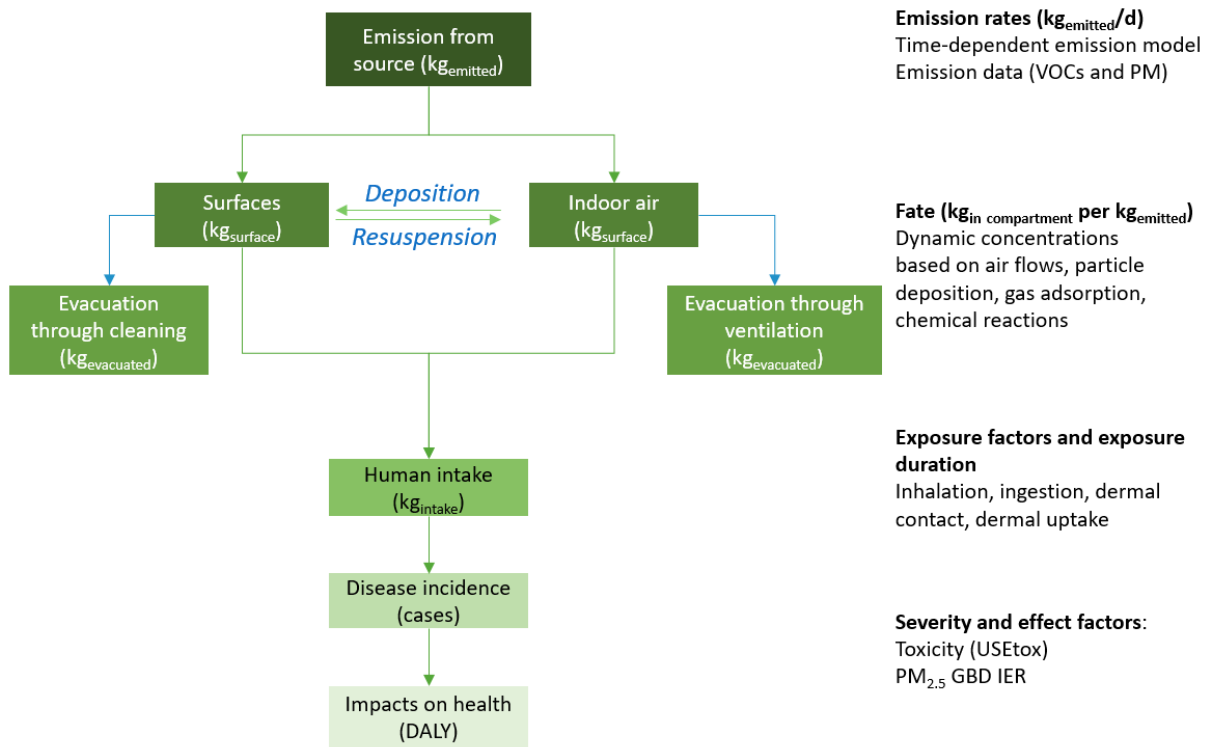


Figure 1-6: Models and data that can be used to address knowledge gaps at each step of the pollutant pathway

#### 4.1.2. Methods

From the pollutant pathway, two methods that aim at meeting the objective by (a) providing a framework similar to current LCA methods and (b) providing case-by-case solutions based on simulations are presented in Figure 1-7.

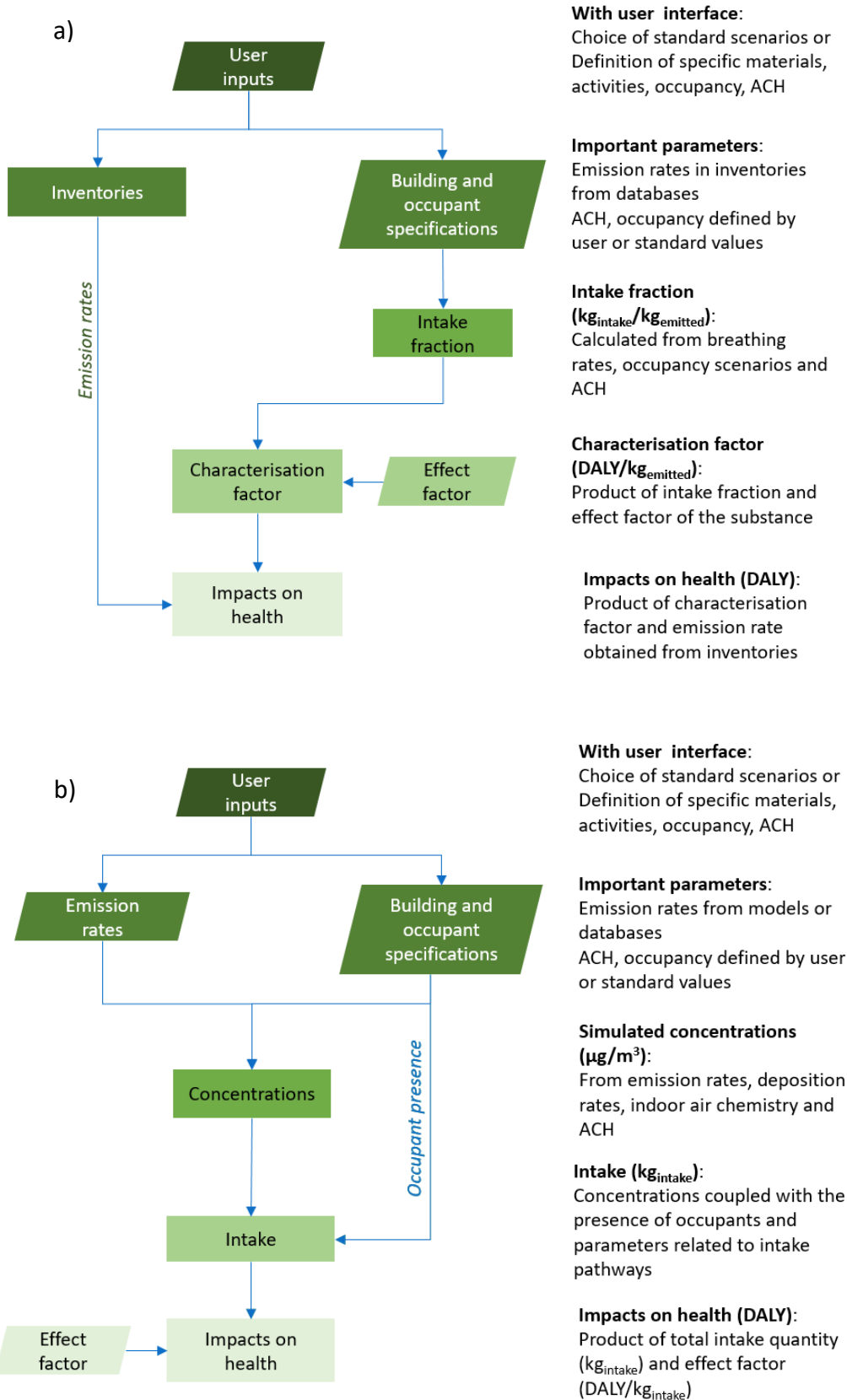


Figure 1-7: Schema representing a) method 1 for inventory-based IAQ impact assessment and b) method 2 for simulation-based assessment



Figure 1-7 (a) shows the method similar to current LCA with two additions: inputs specific to IAQ (either user defined or average values) and exposure calculation requiring a model adapted to specific indoor contexts while (b) shows the simulation-based method where user-defined scenarios determine indoor emission values that are linked to building and occupant specifications to obtain time-dependent concentration curves that can be used to calculate intake and health impacts.

## 4.2. Identification of knowledge gaps

In order to apply the proposed methods, some knowledge gaps at different steps of the pollutant pathway have to be addressed. As seen in this chapter, no study considers all identified pollutant sources, and only one study treats the whole pathway for insulation material emissions. Through this literature review, we identified models describing different steps that can be linked into one general framework.

We choose to base our study on the INCA-Indoor dynamic model, which calculates time-dependent indoor air pollutant concentrations based on emission rates and dynamic airflows for given building characteristics. It also has the advantage of integrating a number of important factors identified in this chapter, besides its dynamic nature: meteorological conditions and indoor-outdoor pollutant transfer according to the building's geographical location, indoor air chemistry and the deposition of particles. Furthermore, it has the advantage of being coupled to a BIM software (IZUBA ÉNERGIES 2001) that allows to conduct the LCA of the same building. Different knowledge gaps, linked to both INCA-Indoor and literature review, to drawing the complete pathway and link IAQ to LCA are described below.

### 4.2.1. Extrapolation of material emissions

Material emissions are time-dependent and measured emission data are only valid for specific times on a restricted time scale. Emission models seen in this literature review allow to predict the emission profile of VOCs from materials, but depend on parameters having high uncertainties. A more robust evaluation of the emission profile can be developed based on both emission data and models. Different layers of materials can create buffer effects and delay the emissions from the lower layers of materials. Building elements (floors, walls...) are rarely monolayered, hence, multilayered emission models are important to consider. Once dynamic VOC emission profiles are obtained, concentrations can be simulated according to the material's age in order to calculate resulting health impacts.

### 4.2.2. Occupant direct emissions

Several studies have measured occupant emissions of indoor pollutants, but the impact of these emissions on health have not been evaluated. INCA-Indoor only includes CO<sub>2</sub> and humidity, considering a standard value for each. However, we have seen that other pollutants, namely VOCs, are present in exhaled air and emitted from the skin. Furthermore, particulate matter deposited on skin and clothes from exterior environments can be resuspended indoors. Their relevance on indoor pollutant concentrations and final impacts on human health should be explored.

#### 4.2.3. Occupant activities (PM and VOCs)

As seen in literature, occupant activities can have an impact on the indoor pollutant concentrations. Activities identified include the use of detergents, candles, cooking and use of solid fuels such as wood for heating or cooking. The activity scenario is important to model as well. These are currently not included in INCA-Indoor, and resulting exposure and health impacts have not yet been calculated in the endpoint DALY unit.

#### 4.2.4. Indoor PM emissions

The INCA-Indoor model currently only treats PM penetrating indoors from outdoors. However, different studies have concluded that indoor sources, in particular occupant activities, can have a strong influence on PM concentration and on health. Their impacts on children health have been calculated as well based on measurements of indoor air concentrations. There is thus a need of using documented emission rates coupled with occupant activity patterns and effect factors to obtain the endpoint health impacts of different PM-emitting activities.

This study will thus link the identified models, with particular focus on the knowledge gaps, in order to propose a framework linking IAQ to building LCA, in order to help in decision-making in the construction sector.

#### 4.2.5. Impacts on human health

INCA-Indoor calculates indoor air concentrations of different pollutants, but the effect factor of each substance is different and concentrations cannot be directly integrated into LCA results. In order to make adequate design choices, we need a single unit of measure that can be used to assess the environmental cost or benefit of different options. The DALY is an appropriate unit and different models have been identified to assess the exposure-impact pathway.



# 2. Emission model and health impacts of VOCs from materials

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Des mesures de concentrations de composés organiques volatils (COV) émis par des matériaux ont été réalisées, souvent dans le cadre de l'étiquetage réglementaire des produits de construction. Ces données ne sont néanmoins valables que pour des instants spécifiques et les conditions des chambres de test. Dans cette étude, nous présentons un cadre permettant de les extrapoler, à l'aide d'un modèle d'émission basé sur le bilan massique et les propriétés des substances et du matériau. Ceci permet d'obtenir le profil complet d'exposition des occupants aux COV et les impacts sur leur santé.

Les paramètres du modèle (coefficient de diffusion  $D_m$  et coefficient de partition matériau-air  $K_{ma}$ ) sont calculés empiriquement et contiennent des incertitudes d'un facteur de 10 à 100. Le calibrage du modèle avec des données de mesure permet d'identifier ces coefficients, ainsi que de trouver la masse initiale  $M_0$ , souvent inconnue, de la substance dans le matériau. La méthode est testée sur deux matériaux : une plaque de plâtre et un parquet en bambou, et validée avec un facteur d'incertitude (racine de l'erreur logarithmique quadratique moyenne) de 1.1. Deux études exploratoires sont aussi présentées : l'application de cette méthode à des variations de taux de renouvellement d'air et à des matériaux multicouches.

Les coefficients optimaux et  $M_0$  sont appliqués à un cas d'étude d'un bureau de 15 m<sup>2</sup> avec des murs couverts de plâtre et un sol en bambou. Les impacts sont calculés sur trois échelles de temps : courte (10 jours), moyenne (100 jours) et longue (10000 jours), correspondant à 27  $\mu$ DALY, 64  $\mu$ DALY and 69  $\mu$ DALY respectivement pour trois occupants.

A travers une analyse de sensibilité, nous avons trouvé que la température, l'âge du matériau et son épaisseur, à masse initiale de substance égale, ont plus d'effet sur les impacts à court-terme alors que le taux de ventilation et l'épaisseur du matériau, à fraction massique égale, influencent plutôt les effets à long-terme. En permettant d'obtenir le profil d'exposition complet, cette méthode peut aider à réduire les impacts des environnements intérieurs à travers l'écoconception des bâtiments et des matériaux.

## Abstract

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Air concentration data of volatile organic compounds (VOCs) emitted from materials have been measured, often for material labelling, but are only valid for specific points in time, restricted timespans and test chamber conditions. The present study provides a framework to extrapolate these data using an emission model based on the mass-balance and the chemical-material properties that define the full emission/concentration profile and calculate health impacts related to the intake of released VOCs. Model parameters (diffusion coefficient,  $D_m$ , and material-air partition coefficient,  $K_{ma}$ ) have been computed from large datasets but contain relatively high uncertainties of a factor 10 to 100. Through the calibration of the emission model with measured data, the model parameters are fixed and the initial mass fractions of substances in the material, often unknown, can be calculated. The framework is tested on two materials with available air concentration data (gypsum board and bamboo flooring) and validated with an uncertainty factor of 1.1 (RMSLE). The calculated parameters are applied to a case study to obtain the full air concentration profile and evaluate health impacts of a 15 m<sup>2</sup> room with gypsum-covered walls and bamboo-covered floor over three time frames: short (10 days), medium (100 days) and long (10000 days), equivalent to 27  $\mu$ DALY, 64  $\mu$ DALY and 69  $\mu$ DALY respectively for three occupants. Through a sensitivity analysis, it was concluded that temperature, material age and material thickness for equal initial substance mass had higher effects on short-term health impacts while ventilation and material thickness with equal mass fractions had more important long-term effects. By providing full concentration and exposure profiles, this framework can help reducing impacts in indoor environments, through building or material eco-design.

---

## 1. Introduction

Materials have been identified as important sources of chemicals present in indoor environments (L. Huang et al. 2022a; Shaw 2004), but impacts related to their use (release of chemicals) are excluded from current LCA methods. In order to consider their local health impacts on building occupants and reduce health impacts right from the building's design, the full concentration and exposure profiles during the building's occupancy have to be studied. These profiles can be obtained from measurements (observations) or models (predictions).

A high number of studies have measured the emission of volatile organic compounds (VOCs) from specific materials (Wilke, Jann, and Brödner 2004; Won et al. 2003; Shaw 2004; James and Yang 2005). Furthermore, since 2011, it is mandatory for manufacturers to measure VOC emissions of construction, wall and flooring materials under the regulatory labelling scheme in France (decree of 19 april, 2011). However, measured data are only available and valid at a few specific points in time, during a restricted timespan of a few days to a few weeks. To obtain a complete emission profile and assess the cumulative intake of these VOCs, concentrations have to be extrapolated. Linear extrapolations cannot be applied since emissions themselves vary with the time dependent mass of the substance left in the material. Simple fit of decay exponential curves has also limited validity to reflect the potential increase of initial air concentration. Moreover, air test chamber measurements and emissions depend on the actual ventilation rate and surfaces to volume ratio, and therefore need to be scaled up to real building usage conditions (Xu and Zhang 2004; Rackes and Waring 2016; X. Zhou, Liu, and Liu 2018). Thus there is a need to couple measurements with more elaborated emission models in order to determine the emission and resulting concentration profiles as a function of time.

Emission models of VOCs from materials are based on the mass balance of the substance emitted from the material and depend on chemical, material and building parameters. Two main parameters characterize the combined influence of material and chemical properties and are unique to each material-chemical combination: 1) the diffusion of these substances through the material, determined by the diffusion coefficient,  $D_i$  and 2) the material-air partition coefficient,  $K_{ma}$ , which is the ratio at equilibrium of the concentration of the chemical at the surface of the material and in the boundary air layer at material surface. QPPRs (Quality Property-property Relationship) have been developed to compute material-specific properties such as  $D_m$  and  $K_{ma}$ , for multiple material types from chemical properties more commonly available such as molecular weight and the octanol-air partition coefficient,  $K_{oa}$  (Huang et al. 2017; Huang and Jolliet 2019). Developed from large datasets of about 1000 measurements, these QPPRs reference values can be used to simulate average indoor concentrations when no specific emission data are available, but are associated with relatively large uncertainties of a factor 10 to a 100. Another key parameter is the mass fraction of volatile substances in the material,  $MF_0$ , which determines the amount that can be emitted in air. Material compositions are

often unknown, except for the average values that can be found for some material categories in e.g. the Pharos database (Friar and Vittori 2017).

Measurements do not represent real-life occupied indoor environments and are only available at specific points in time, while models contain uncertainties and unknowns. The combination of measurements and emission models could take advantage of the respective strengths of these approaches and help predicting more precisely the evolution of indoor VOC concentrations. Furthermore, since VOCs can have serious adverse effects on health, it is beneficial to model intake and health damages on building occupants to make appropriate choices at the design phase. The present study therefore aims to propose and apply a framework combining measured emission data with a mass balance model of VOC emission from materials. More specifically, the different objectives are to:

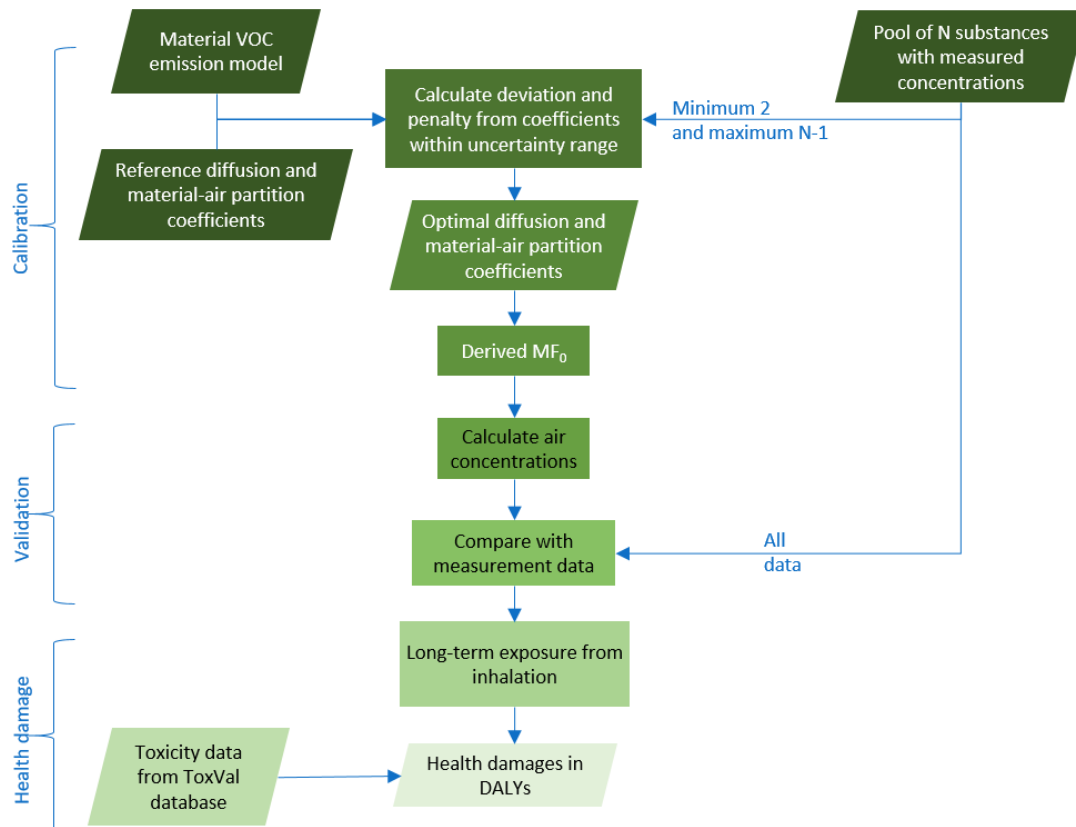
1. develop a framework using a multi-layered emission model for interpreting and extrapolating measured emission data from building materials.
2. perform a parameter optimization and evaluate the quality of the approach based on sets of chemicals in gypsum board and bamboo flooring.
3. determine long-term chemical intakes and related human health impacts and compare across chemicals.

First, we will present the framework developed to calibrate an emission model using emission data. The framework will then be applied to two mono-layered materials: gypsum board and bamboo flooring. The applicability of this method to multilayered materials is going to be presented as an exploratory step and a particular case of emission data for the same material but with variations in air change rates (ACH) will be discussed. In a case study, we will evaluate the health damages related to the intake of VOCs emitted by the gypsum board and bamboo flooring. Finally, sensitivity tests will be conducted for ventilation, temperature and material age.

## 2. Materials and Methods

The works in this chapter are mainly based on Bhoonah et al. (2023). The driving input parameters of an emission model are adjusted using available measured concentrations from test chamber experiments to simulate continuous VOC emissions from materials. The framework is applied to two monolayered materials: gypsum board and bamboo flooring. The applicability to a multilayered material, PVC flooring, is tested. Particular cases of changing ACH is also discussed. The main steps are summarised in [Figure 2-1](#).





*Figure 2-1: Methodology for extrapolating VOC emission data using an emission model: calibration, application and validation on different materials with measured data and determination of health damages from exposure*

From a test chamber measurement pool of  $N$  substances emitted by the material under study, a minimum of 2 and maximum of  $N-1$  substances are selected and coupled with an existing emission model and reference coefficient values in order to estimate the optimal diffusion and material-air partition coefficients unique to the material that best fit the measurements. The initial mass fraction,  $MF_0$ , of each substance, specific to each material, that yield the lowest error of the predicted compared to the measured concentrations is then derived. The framework is validated by comparing the predicted concentrations to measured data for all  $N$  substances, and, distinctly, validation data which have not been used for calibration. Using the determined optimal parameters, the inhalation exposure to material VOCs can be determined by the model and combined with exposure-response data to evaluate corresponding health damages.

### 2.1. Measured air concentration data in test chamber

Measurements of air concentrations of different substances emitted by the material at different points in time are collected in literature and from material manufacturers. Since these measurements are made in test chambers, the following chamber characteristics are considered: 1) the chamber walls are made of very low absorption or emission materials such as glass or stainless steel, in order not to interfere with the tested material and 2) only one face of the material emits substances, the other being covered

by an impermeable material. The ventilation rate and temperature during measurements must also be known.

In this study, we used data on two mono-layered materials - gypsum board and bamboo flooring. Two other materials will be used in an exploratory step: plywood to assess applicability of the method on varying ACH (because measurements for this material were taken at three different ACH), and PVC flooring for its applicability to multilayered materials. Air concentration data ( $\mu\text{g}/\text{m}^3$ ) are available at 3 days ( $t_1$ ) and/or 28 days ( $t_2$ ) for gypsum board, bamboo flooring and plywood from the manufacturer's data sheet, with uncertainty range only available for PVC (22%). Emission rates ( $\mu\text{g}/\text{m}^2/\text{h}$ ) for plywood board were obtained from Maupetit et al. (2017) at 3 days, 28 days, 35 days and 42 days. The tests were carried out at 23 °C and a relative humidity of 50 %, with an air change rate of 0.5 vol.h<sup>-1</sup>, except for plywood board with ventilation rates of 0.063 m<sup>3</sup>/h up to day 28, 0.124 m<sup>3</sup>/h for days 29 to 35 and 0.039 m<sup>3</sup>/h for days 36 to 42. A summary of relevant information is presented in Table 2-1.

*Table 2-1: Main characteristics of the measurement of VOCs emissions from gypsum board, bamboo flooring and PVC flooring*

Parameter	Gypsum board	Bamboo flooring	PVC flooring	Plywood
Number of substances measured (at both $t_1$ and $t_2$ )	21 (5)	11 (3)	11 (5)	6 (6)
Thickness of material (mm)	12.5	38	3.25	15
Density (kg.m <sup>-3</sup> )	1150	1150	1000	550
Chamber volume (m <sup>-3</sup> )	0.123	0.250	0.119	0.051
Loading ratio (m <sup>2</sup> .m <sup>-3</sup> )	1	0.4	0.4	4.4

## 2.2. Emission model and main parameters of influence

The multi-layered material emission model used in this study has been developed by Yan et al. (2009), modified by Guo (2013), and further adjusted by (Micolier 2019). The material is divided into several layers. Each layer is considered to have a uniform substance concentration and is represented by a single node. The nodes are the indoor air node, boundary layer nodes, inner nodes inside the material and nodes across the interface, as represented in Figure 2-2. The emission rate is obtained from the mass balance of substances at different nodes within the material and in air, as a function of the initial mass fraction of the substance in the material,  $MF_0$  (kg<sub>substance</sub>/kg<sub>material</sub>).

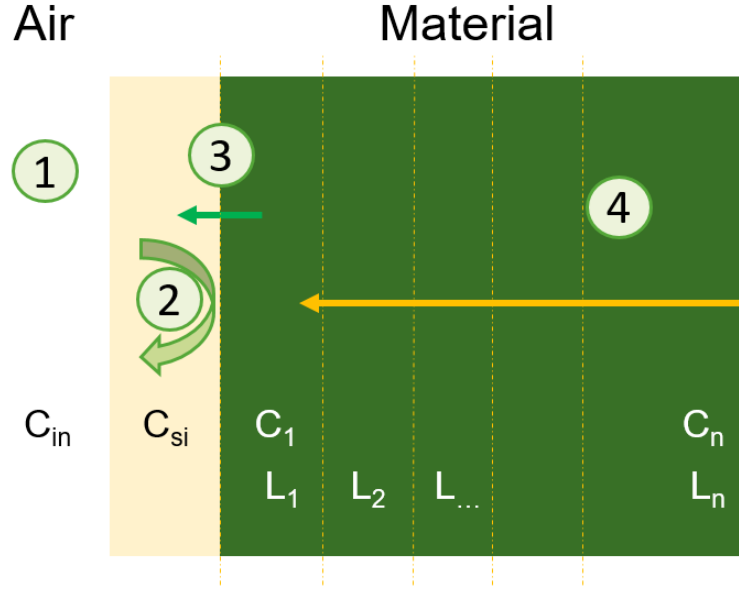


Figure 2-2: Different layers within the material, at the boundary and indoor air

Gaseous substances diffuse through layers  $L_1$  to  $L_n$  within the material (4), before reaching the interface between the material and air (3). Once emitted, at the boundary layer between indoor air and the material (2), convection drives the substance into indoor air (1). The underlying equations of the emission model is given in annex A1.1.

In this study, the main focus will be the two major parameters driving VOC emissions, i.e. the diffusion coefficient through the material  $D_m$  (m<sup>2</sup>/s), expressed in equation (2-1) and the material partition coefficients  $K_{ma}$  (-), expressed in equation (2-2). This emission can be limited by its diffusion through the material (D-limited) or by its partition from the material to the boundary layer (K-limited). These two parameters have been estimated from material and substance specific coefficients (Huang et al. 2017, Huang and Jolliet 2019) as follows:

$$D_m = 10^{6.39 + \frac{\tau - 3486}{T} - 2.49 \log_{10} MW + b} \quad (2-1)$$

$$K_{ma} = 10^{-0.38 + 0.63 \log_{10} K_{oa} + 0.96 \frac{1.37 \nabla H_v - 14}{2.303 R} \left( \frac{1}{T} - \frac{1}{298.15} \right) + \beta} \quad (2-2)$$

$MW$  (g/mol) is the molecular mass of the substance,  $R$  (J/mol) is the gas constant,  $T$  (K) is the absolute temperature of the room and  $b$ ,  $\tau$  (K) and  $\beta$  are material-specific coefficients,  $\nabla H_v$  (J/mol) the enthalpy of vaporisation and  $K_{oa}$  is the chemical's dimensionless octanol-air partition coefficient at 25 °C.

Coefficients  $b$  and  $\beta$  have several reference values as a function of the type of material independently of the considered chemical. These values have been obtained by Huang et al. (2017) and Huang and Jolliet (2019) based on the analysis of more than 1000 measured diffusion coefficients and

material-air partitioning coefficients, with high  $R^2$ , but still substantial remaining uncertainties: the standard error (SE) on  $\log_{10}D$  and  $\log_{10}K$  amounts to  $\pm 2.30$  and  $\pm 1.22$  respectively, corresponding to uncertainty factors of 200 and 17. It is therefore interesting to take advantage of the measured concentrations to calibrate these parameters within their uncertainty range to the specific considered material.

### 2.3. Selection of substances for main parameter calibration

In order to optimise the parameters  $b$  (material-specific coefficient for diffusion) and  $\beta$  (material-specific coefficient for partition), a calibration is realised for each material using data for part of  $N$  substances for which measurements at two points in time ( $t_1$  and  $t_2$ ) are known, the other substances (at least one) being kept for the validation step. Excluding one substance for validation (from  $N-1$ ), we select at least two and up to four substances for calibration, with the broadest range possible in their diffusion ( $D_m$ ) and partition coefficients to cover various emission dynamics: the substance having the highest  $D_m$  and highest  $K_{ma}$ , and the one with the lowest  $D_m$  and lowest  $K_{ma}$  are chosen. If more than one substance remains, up to two additional substances are selected: the one(s) having the highest absolute difference between  $D_m$  and  $K_{ma}$ . Any remaining substance is kept for validation. A detailed description of the selection process is given in annex A1.2.

### 2.4. Parameters calibration

$b$  and  $\beta$  are first varied within their range of uncertainty using a 50 x 50 grid, yielding 2500 combinations of  $D$  and  $K$ . For each of these combinations  $b-\beta$ , we perform the following iterative process: a) We first back-calculate the initial mass fraction  $MF_{0,b-\beta}$  of the substance  $s$  under study<sup>1</sup> that correctly predicts the concentration at time  $t_1$ ,  $C_{t_1}$ . b) We then evaluate for each  $b-\beta$  combination the deviation ( $d_{b-\beta_s}$ ) between the log of observed and predicted ratios of  $C_{t_1}/C_{t_2}$  calculated as:

$$d_{b-\beta_s} = \left| \left( \log_{10}(C_{t_1,m_s}) - \log_{10}(C_{t_2,m_s}) \right) - \left( \log_{10}(C_{t_1,p_{s,b-\beta}}) - \log_{10}(C_{t_2,p_{s,b-\beta}}) \right) \right| \quad (2-3)$$

c) We calculate for each  $b-\beta$  combination, the mean deviation ( $\bar{d}_{b-\beta}$ ) across all calibration substances

d) We also determine for each  $b-\beta$  combination a penalty for diverging from the reference  $D_m$  and  $K_m$  values, determined as follows:

---

<sup>1</sup> This step only serves the purpose of calibration. The actual mass fraction of the substance in the material is determined later based on all available measurements (see section below).

$$p_{b-\beta} = 0.1 \times (abs(\log_{10}(K_{ma,b-\beta}) - \log_{10}(K_{ma,ref})) + abs(\log_{10}(D_{m,b-\beta}) - \log_{10}(D_{m,ref}))) \quad (2-4)$$

The optimal  $b-\beta$  combination is selected as the one minimizing  $\varepsilon_{b-\beta_{tot}}$ , the sum of the obtained average deviation plus the penalty  $p_{b-\beta}$  :

$$\varepsilon_{b-\beta_{tot}} = \sum_{s=1}^n \bar{d}_{b-\beta_s} + p_{b-\beta_s} \quad (2-5)$$

## 2.5. Mass fraction and resulting air concentration

Since initial mass fractions of substances inside the measured materials ( $MF_0$ ) are usually unknown, we use the optimal parameters from the calibration step to determine  $MF_0$  as the mass that minimizes the root mean squared log error (RMSLE) on all concentrations for all times ( $i=1,n$ ) and substances ( $s=1,x$ ):

$$e_{MF_0} = \sqrt{\frac{1}{n} \sum_{i=1}^n \sum_{s=1}^x (\log_{10}(C_{t_i,m_s} + 1) - \log_{10}(C_{t_i,p_s,MF_0} + 1))^2} \quad (2-6)$$

We then apply the emission model from (Micolier 2019) to yield predicted air concentrations.

## 2.6. Validation

Predicted concentrations are compared to measurements for validation. Different error indicators are selected to represent different information. The percentage error represents the relative absolute difference between measured and predicted values, without differentiating between high or low values. We also calculate the root mean squared log error (RMSLE) and the mean absolute log error (MALE), both giving low significance to the actual value or scale of the error, as compared to the root mean squared error (RMSE) which increases when the scale of error increases (Tofallis 2015). This parameter is particularly important because we intend to calculate human health impacts, which do not only depend on the magnitude of concentrations, but also on the effect factor of the substance. Thus, we do not penalise substances with low concentrations since their effect factors could be high, leading to high impacts. Uncertainty factors are calculated, equal to  $10^{RMSLE}$  and  $10^{MALE}$ .

## 2.7. Health impacts calculation

Effect factors (cases/kg<sub>intake</sub>) are calculated for VOCs according to the ToxVal database (US EPA and Richard 2018). Health impacts  $HI$  (DALY) are calculated in DALYs (disability-adjusted life years), a measure of the number of healthy life years lost due to sickness or premature death recommended by the World Health Organisation (Murray et al. 1996), currently used as an endpoint

indicator in life cycle assessment (LCA). It is computed from the severity factor  $SF$  (DALY/case), effect factor  $EF_{disease}$  (cases/kg<sub>intake</sub>), product intake fraction  $PiF$  (kg<sub>intake</sub>/kg<sub>in product</sub>) and the initial mass of substance in the material  $M_0$  (kg<sub>in product</sub>).

$$\begin{aligned}
 HI = & (SF_{rep/dev} \times EF_{rep/dev} \\
 & + SF_{general non cancer} \times EF_{general non cancer} \\
 & + SF_{cancer} \times EF_{cancer}) \times PiF \times M_0
 \end{aligned} \tag{2-7}$$

With

$$PiF = \frac{XF \times V_{room} \int_{t=0}^T C_{in} dt}{M_0} \tag{2-8}$$

The effect factor  $EF_{disease}$  (cases/kg<sub>intake</sub>) and severity factors (DALY/cases) are obtained for: reproductive/ developmental non-cancer diseases (*rep/dev*), general non-cancer diseases and cancer. The calculation of the  $EF$  of a substance, obtained from its  $ED_{10}$  and  $ED_{50}$  (the lifetime dose per person that increases the probability of a cancer disease, a general non-cancer disease or a reproductive/developmental non-cancer disease by 10% or 50%) is detailed in annex A1.3 and values are given in Annex Table 1.  $XF$  (kg<sub>intake</sub>/s per kg<sub>in compartment</sub>) is the exposure factor representing the fraction of the air volume (and subsequently of substances, considering a uniform concentration in the room) that is taken in by the occupants per unit time, calculated using parameters recommended by the Exposure Factors Handbook of the Environmental Protection Agency (US EPA 2011) for each pathway: inhalation, gaseous dermal uptake, dust ingestion and direct dermal contact, using equations (1-2), (1-3), (1-4) and (1-5).

## 2.8. Case study

We selected as a case study a room with of 38.75 m<sup>2</sup> gypsum-covered walls and 15 m<sup>2</sup> bamboo flooring. The room is ventilated at 1 ACH (air changes per hour, in h<sup>-1</sup>) and occupied by 3 persons 8 hours per working day and indoor temperature is set at 20°C. Health impacts are calculated for the case study.

The sensitivity of health impacts to the following parameters are tested for gypsum board: ACH, temperature, material age and material thickness using a one-at-a-time (OAT) Morris method (Morris 1991). For the model with  $N$  input parameters, we calculate the global sensitivity using the mean elementary effects  $\mu$  of  $i^{th}$  parameter of value  $j$ ,  $x_i^j$ , for a total of  $r$  values:

$$\mu_i = \frac{1}{r} \sum_{j=1}^r \frac{f(x_1, \dots, x_i^j + \Delta_i, \dots, x_N) - f(x_1, \dots, x_i^j, \dots, x_N)}{\Delta_i} \tag{2-9}$$

Where a large  $\mu$  indicates a large effect of the parameter on the model output.

## 2.9. Applicability and assumptions

The framework is applicable under the following conditions: 1) data is available on test chamber conditions, namely volume, air change rate and temperature, 2) dimensions of the material are known and 3) at least two measurement points for at least three substances are available, since emission data for at least two substances are used for calibration and the model is validated with at least one other substance.

The model is applied under the following assumptions: 1) the concentration of the substance is initially identical for all layers of the same material, 2) the concentration of the substance in each layer is uniform at any point in time, 3) the substances emitted are volatile, 4)  $D_m$  and  $K_{ma}$  are unique to all layers of the same material and 5) emissions occur under stable atmospheric conditions with constant pressure, relative humidity and temperature.

## 3. Results

### 3.1. Deviation of predicted values from measured values

The above framework is applied to the two mono-layered materials with available measurement data: gypsum board and bamboo flooring. Hexanal, toluene and n-hexadecane were selected for the calibration of the gypsum board parameters whereas acetic acid and phenol were used for bamboo flooring calibration based on the criteria described in annex A1.2. Figure 2-3 presents for each of these chemicals-material combinations the deviation between the log of observed and predicted ratios of concentrations  $C_{t_1}/C_{t_2}$ , as calculated by equation (2-3).

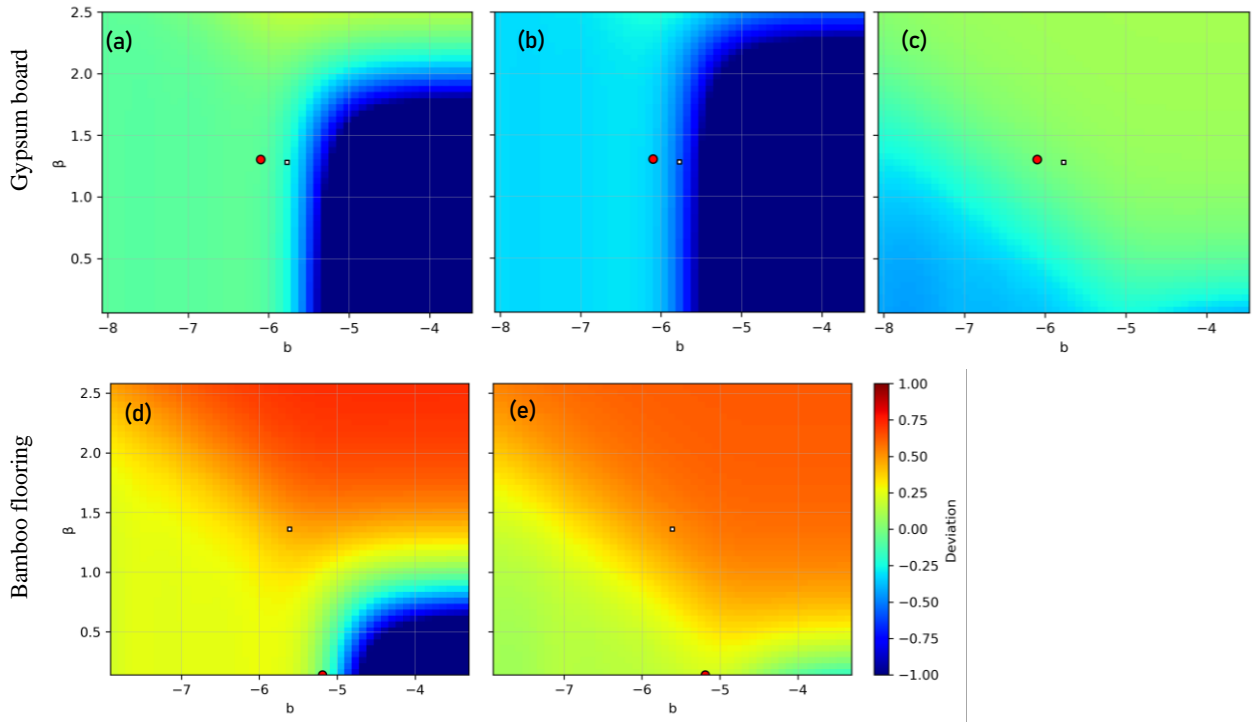


Figure 2-3: Heat maps of  $d_{b-\beta_s}$ , the deviation between the log of observed and predicted ratios of concentrations  $C_{t_1}/C_{t_2}$  for (a) hexanal, (b) toluene, (c) *n*-hexadecane emissions from gypsum board and (d) acetic acid and (e) phenol from bamboo flooring as a function of the material coefficient  $b$  for diffusion and  $\beta$  for material-air partition. The reference default values of  $b$  and  $\beta$  are indicated by a yellow dot at the centre and optimal values by a larger red dot. The scale of deviation, from -1 to 1, is given on the right side of (e).

The optimal parameters  $b$  and  $\beta$  minimize the average deviation across chemicals as determined by equation (2-5) (black-outlined red dots in Figure 2-3) and therefore fall in the light-green or light blue areas that correspond to values that make the model best agree with measurements and yield smallest deviations. For the gypsum board, the deviation primarily depends on the diffusion coefficient  $b$ , with deviation of up to a factor 10 ( $d_{b-\beta_s} = -1$ , dark blue area) between modelled and measured concentrations of hexanal and toluene at low diffusion coefficient. The optimal  $b$  and  $\beta$  combination is obtained for  $b = -6.10$  and  $\beta = 1.3$ , which is close (within a factor 2 for the diffusion coefficient) to the reference values of -5.77 and 1.26.

For the bamboo flooring, emissions are sensitive to both diffusion and material-air partition coefficients when considering acetic acid and phenol. The low-deviation (light green) regions are quite limited. Based on the deviation and penalty, the optimal solutions of  $b$  and  $\beta$  for bamboo flooring are -5.19 and 0.14, thus with a factor 16 lower on the material air partition coefficient than the default reference value of -5.61 and 1.36.



### 3.2. Extrapolated emission and concentration profiles

For each substance, the optimal parameters are used to determine  $MF_0$  based on equation (2-6) are applied to the model, and obtain the air concentration of different emitted substances as a function of time. Figure 2-4 compares the predicted concentration curves to the measurement points and present the mean predicted concentrations over 10, 100 and 10000 days, showing a relatively good concordance between measured and modelled values.

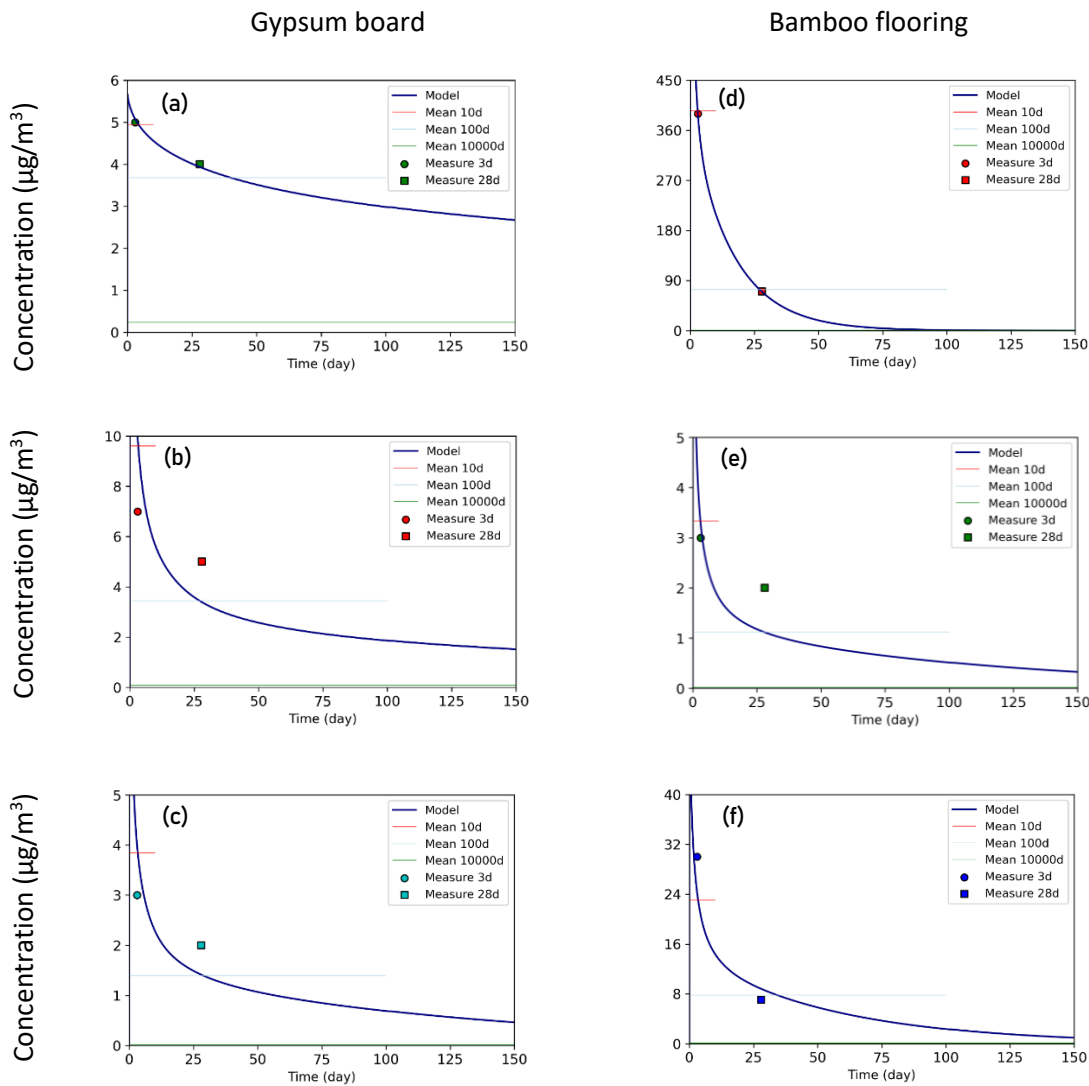


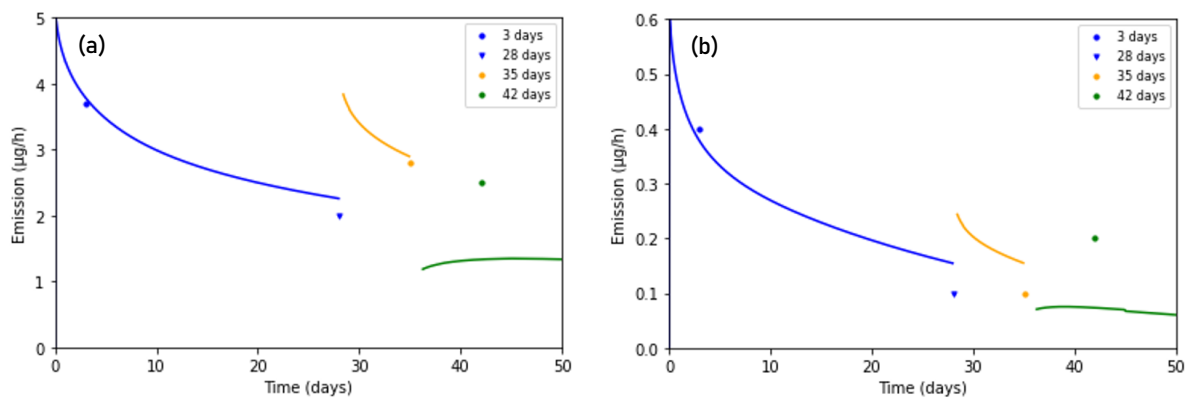
Figure 2-4: Evolution of the predicted concentrations for (a) n-hexadecane, (b) n-pentadecane, (c) nonanal emitted by gypsum board, and (d) acetic acid, (e) 2-ethyl 1-hexanol, (f) phenol emitted by bamboo flooring: The mean predicted concentrations over 10, 100 and 10000 days are presented by red, light blue and green lines respectively.

The volatile nature of the considered substances is reflected in the decrease in mean concentrations with increasing time frame. Since the first measured point is higher, its squared difference to the model tends to be higher as well and the concentration curves tend to be closer to this first

measured value, within a factor 1.4. We also note that it is a non-conservative method (unlike regular LCA methods), since it lies between a high and low mass fraction that give each measurement. For instance, for nonanal, the  $MF_0$  that yields the correct  $C_{3d}$  is lower than that giving  $C_{28d}$ , and the retained value is lower than the maximum possible value at  $C_{28d}$ .

### 3.2.1. Exploratory approach accounting for change in ACH in test chambers

For some test chamber experiments, the air change rate (ACH) is varied during measurements in order to study their implications on emission rates. The model was adapted to take these changes into account in this exploratory step. The material studied is unvarnished plywood board with four emission rates ( $\mu\text{g}/\text{m}^2/\text{h}$ ) available for six VOCs at 3 days, 28 days, 35 days and 42 days. The chamber is ventilated at 0.063 m<sup>3</sup>/h up to day 28, 0.124 m<sup>3</sup>/h for days 29 to 35 and 0.039 m<sup>3</sup>/h for days 36 to 42. Heat maps representing the deviation according to equation (2-3) are presented in annex A1.3. Emission curves for two substances are presented in Figure 2-5 for (a) hexanal and (b) propanal.



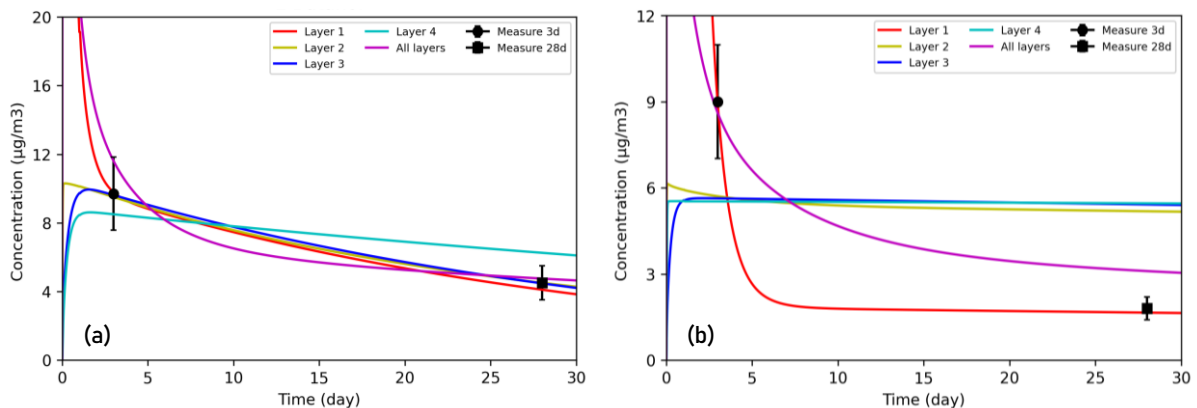
*Figure 2-5: Emission profile for two substances emitted by the plywood board: (a) hexanal and (b) propanal. Curves represent predicted emission rates and markers represent measured rates in blue, yellow and green for three different ACH*

An increase in ACH leads to increased predicted emission rates for both substances due to an increase in convection in the boundary layer (see annex A1.1). This agrees with the trend in measured rates for hexanal, but not for propanal, for which the emission rate increases at 42 days, though the ACH decreases at that point. Consequent differences between prediction and measurements are observed (A1.3). The reasons could be that model assumptions do not match real-life experimental set-ups: the initial concentrations through the material might be non-uniform and changes in ventilation rates could be gradual, while they are modelled as instantaneous. There might also be uncertainties in measurements, which are unknown.

### 3.2.2. Exploratory approach for multilayered materials

The case of PVC floor was selected to test the approach on a multilayered material composed of a PUR surface treatment, a wear layer of PVC, a layer of glass fibre, another layer of PVC for resistance and a polyurethane foam backing. The reference values ( $b$  and  $\beta$  corresponding to -6.77 and 2.26 respectively) treat PVC flooring as a mono-layered material, but they could vary for different products of the same category with alternate compositions. The importance of considering the different layers instead of treating the material as one block is assessed.

The total thickness of the flooring is 3.25 mm and the thickness of the wear layer is 0.8 mm, but the other thicknesses are unknown and set according to average references on the market, presented in annex A1.5. No information is available on the distribution of the substances in the different layers. Due to high levels of uncertainty related to these parameters, this study is exploratory and shows the applicability of the developed approach to multilayered materials, and, in particular, to predict layers in which substances are most likely to be initially present. Figure 2-6 shows the predicted air concentrations assuming initial presence in each or all layers of the flooring for ideal model parameters ( $b$ ,  $\beta$  and  $MF_0$ ) within the uncertainty range that predict concentrations closest to the measurements.



*Figure 2-6: Concentration curves for (a) 1-butanol and (b) phenol assuming initial presence in each layer or all layers of the material and measurement points at 3 and 28 days represented by markers with uncertainty bars*

The least error between predicted and measured concentrations for 1-butanol is obtained by assuming its initial presence in layer 3, while phenol is most likely to be present in layer 1. Coefficients for each layer (unique  $b$  and  $\beta$  for each material and unique  $MF_0$  for each substance) yielding lowest deviations are calculated and resulting concentration curves are presented in annex A1.5.

### 3.3. Uncertainties: gypsum board and bamboo flooring

The model is validated against the test substances calibration, estimating the percentage error, the root mean squared log error (RMSLE) and the mean absolute log error (MALE). Since predicted concentration of different substances can vary over orders of magnitude, selecting the error on the log

of concentrations gives equal importance to relative errors across the entire range of concentration and avoids a bias towards substances with higher concentrations. Figure 2-7 compares the predicted and measured air concentration, with circle markers for the gypsum board and triangles for the bamboo flooring mono-layered material.

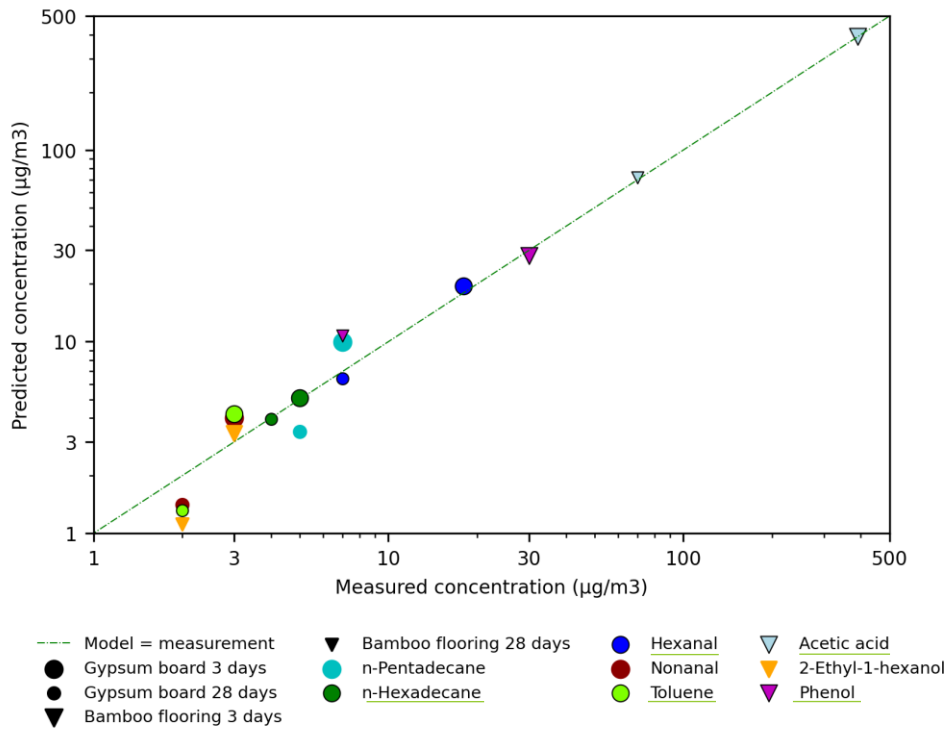


Figure 2-7: Measured v/s predicted concentrations for 3-day (bigger markers) and 28-day (smaller markers) measurements from gypsum board (circle markers) and bamboo flooring (triangle markers) for all substances with both measurements known. The substances with a black-outlined marker have been selected for the calibration process and are underlined in the legend

Figure 2-7 shows good agreement between measured and modelled data for both the calibration data and for the test data with less than a factor two for the test substances pentadecane and nonanal (cyan and brown circles) in the gypsum flooring and for 2-ethyl 1-hexanol in the bamboo flooring (orange triangle).

Table 2-2 summarises the errors (percentage error, RMSLE and MALE) for the substances divided into three categories: all substances, only test substances and only substances selected for the calibration process.

*Table 2-2: Relative error, Root Mean Squared Log Error, and Mean Absolute Log Error (MALE) for all substances, only validation points and only substances selected for the calibration*

<b>Indicator</b>	<b>All substances</b>	<b>Only test substances</b>	<b>Only calibration</b>
<b>% Error</b>	22%	32%	16%
<b>% Error 3d</b>	18%	29%	11%
<b>% Error 28d</b>	26%	35%	20%
<b>RMSLE</b>	0.02	0.05	0.02
<b>RMSLE uncertainty factor</b>	1.04	1.13	1.04
<b>MALE</b>	0.19	0.30	0.13
<b>MALE uncertainty factor</b>	1.56	1.99	1.34

As expected, the error indicators are lower for substances used in the calibration procedure whereas, prediction remains within max a factor 2 of the observed value for the test substances. Relative error, for all substances and all measurements, on the predicted values is of 22%, corresponding to average errors on air concentration measurements, generally around 20%, but which could range from 5% to 25% (Wilke et al. 2009; Jia, Batterman, and Relyea 2012).

For the bamboo flooring, the predicted initial mass fractions of  $7.6 \times 10^{-7}$  for phenol and  $8.4 \times 10^{-8}$  for toluene respectively 2 and 6 orders of magnitude lower than reported ones in Pharos. Lower predictions could be explained by the difference between the composition of the product under study and those referenced in Pharos. Calculated mass fractions correspond to the amount of substance in the material at the beginning of chamber tests. Thus, low-end values could also come from the time elapsed between the manufacture and the chamber tests, and, consequently, partial emission of substances before measurements are taken.

Uncertainties related to unvarnished plywood board and PVC flooring are presented in sections [A1.3](#) and [A1.5](#).

### 3.4. Application to a case study: gypsum board and bamboo flooring

The case study parameters, optimal  $b$  and  $\beta$  and mass fractions calculated previously for both materials are applied to the model for substances studied (those with two measured data). For points with single measured data,  $MF_0$  is back-calculated from the measured concentration using the same optimal  $b$  and  $\beta$  as for the other substances. The intake fractions are calculated from equation (2-8) using concentrations and exposure fractions and the consequent health damages are evaluated using equation (2-7).

### 3.4.1. Mass in different compartments

Volatile substances diffuse between different compartments defined as: 1) the material under study (walls), 2) other materials (rest of building envelope), 3) indoor air, 4) exhaust air of 1 vol/h (from indoor to outdoor, with the same concentration as indoor air), 5) outdoor air and 6) the human body. Figure 2-8 shows the mass of two substances with different emission dynamics in each compartment as a function of time: toluene and n-hexadecane.

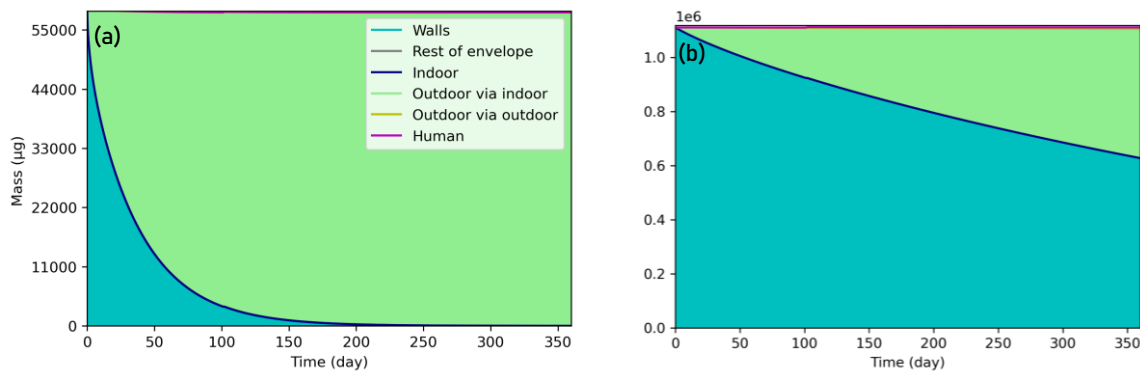


Figure 2-8: Mass balance of two substances: (a) toluene and (b) n-hexadecane with different emission dynamics emitted by gypsum board in six compartments: walls (emitting material), rest of envelope, indoor air, outdoor air via indoor, outdoor air via outdoor and human body

Toluene has a  $D_m$  of about one order of magnitude lower than n-hexadecane and a  $K_{ma}$  two orders of magnitude lower and is emitted from the walls and evacuated almost completely after 200 days to outdoor air. The initial mass of n-hexadecane is about 20 times higher than toluene, contributing to its slower release together with its high partition coefficient. A large fraction of it remains inside the walls after a year. Outdoor contribution via outdoor sources is equal to zero it is considered that there is no VOC emission outdoors. The intake by occupants is negligible compared to the total mass, due to low masses in indoor air (resulting from low room volume compared to outdoors and high ventilation rate) and relatively low breathing rates compared to evacuation rates:  $1.85 \times 10^{-4} \text{ m}^3/\text{s}$  breathing rate versus  $1.03 \times 10^{-2} \text{ m}^3/\text{s}$  ventilation rate. Intake quantities are, however, not necessarily a measure of the dangerousness of the product: for this, health impacts have to be calculated using effect factors.

### 3.4.2. Health impacts

Health impacts are calculated according to equation (2-7) and using data available from USEtox and ToxVal for inhalation and ingestion (due to lack of data for dermal uptake and dermal contact). Figure 2-9 resents the intake (µg) and health impacts (µDALY) of substances emitted by gypsum board and bamboo flooring over short, medium and long-term exposure corresponding to 10, 100 and 10000 days respectively.

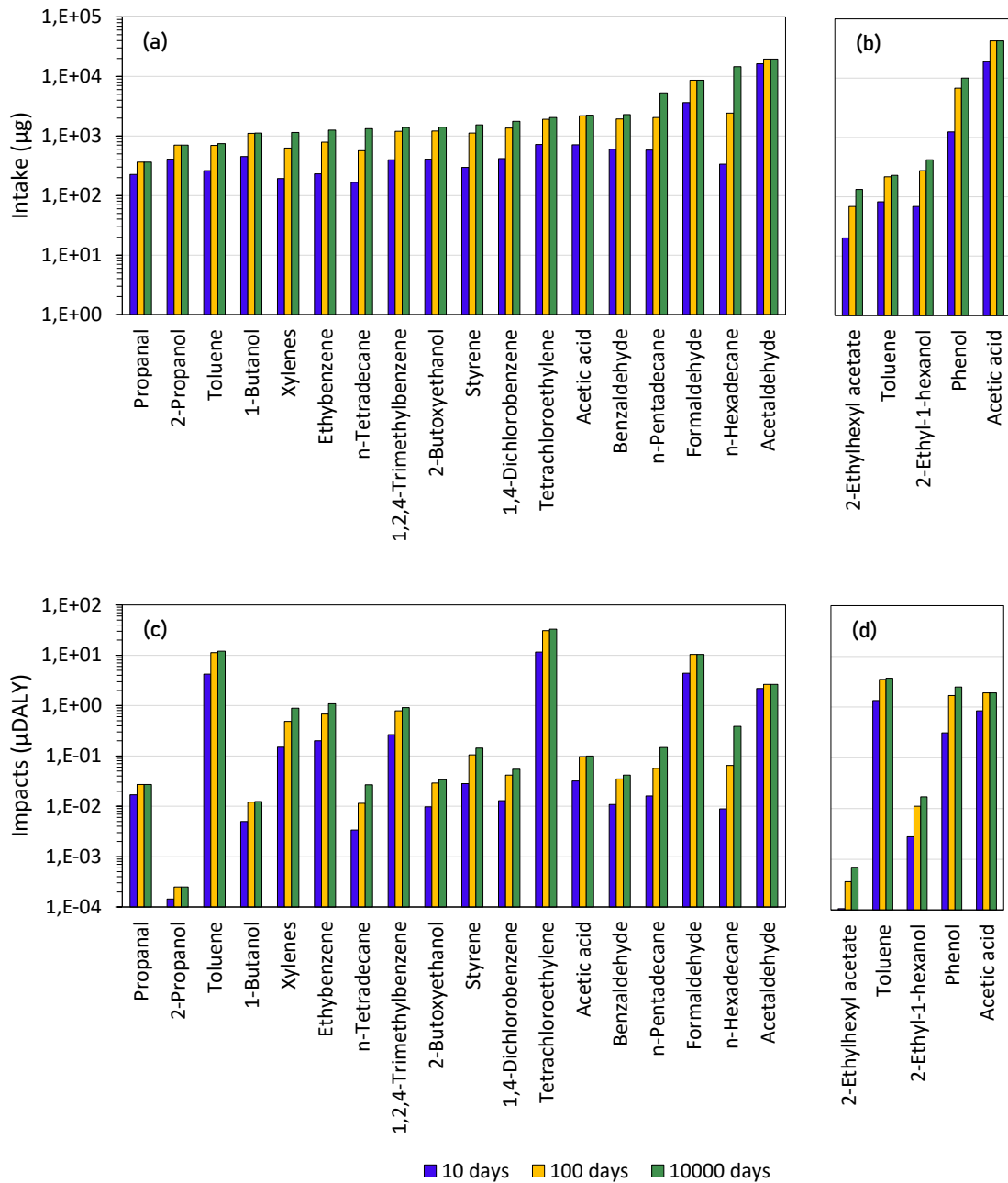


Figure 2-9: Intake quantities (µg) for (a) 17 substances emitted by gypsum board and (b) 6 substances emitted by bamboo flooring and health impacts (µDALY) for different substances emitted by (c) gypsum board and (d) bamboo flooring over time frames of 10, 100 and 10000 days

Figure 2-9 shows that the differences between short, medium and long-term exposures for certain substances, namely acetaldehyde emitted from gypsum board or acetic acid from bamboo flooring, are not significant since these substances are highly volatile and rapidly emitted. We also note that, for some substances, such as n-hexadecane emitted from gypsum board, the short-term impacts can be lower but long-term impacts higher than certain substances such as n-pentadecane, since their emission dynamics are different.

It can also be noted that impacts can be different for similar intake quantities, or that they can be higher for substances with lower intake quantities since there can be substantial variations between effect factors of different substances. For example, the intake quantity of acetic acid from bamboo flooring is of  $4.28 \times 10^4 \mu\text{g}$  while for phenol it is of  $1.00 \times 10^4 \mu\text{g}$ . Their respective health damages are 1.91  $\mu\text{DALY}$  and 2.50  $\mu\text{DALY}$ . Despite having over 10 times the intake quantity of phenol, acetic acid is responsible for less health impacts due to its lower effect factor (given in [Annex Table 1](#)). It can also be noted that the differences between the health damage from different substances are several orders of magnitude high and that the substances of concern in each material can be different. In the case of gypsum board, tetrachloroethylene is responsible for 53% of the total damage from VOC emissions, toluene for 19% and formaldehyde 17%. For bamboo flooring, toluene has the highest contribution, accounting for 46% of health impacts, phenol 31% and acetic acid 23%. [Table 2-3](#) summarises the health damages related to the exposure to VOCs emitted by the three materials for short, medium and long-term exposures. The equivalent damage for 1  $\text{m}^2$  flooring area per year is also given based on the long-term exposure.

*Table 2-3: Health damages related to the intake of substances with known toxicity data for gypsum walls and bamboo floor for the considered office (3 persons in 15  $\text{m}^2$ ) over three exposure periods, and per unit area of flooring per year*

<b>Health impacts</b>	<b>Gypsum walls</b>	<b>Bamboo floor</b>
10 days ( $\mu\text{DALY}$ )	23.18	2.50
100 days ( $\mu\text{DALY}$ )	57.60	7.10
10000 days ( $\mu\text{DALY}$ )	61.97	8.14
10000 days ( $\mu\text{DALY}/\text{m}^2/\text{year}$ )	0.15	0.02
10000 days ( $\mu\text{DALY}/\text{kg}$ )	0.11	0.02

It can be noted that, for the functional unit of 1  $\text{m}^2$  flooring area, gypsum board leads to more health damage on occupants than bamboo floor. Results can also be interpreted per unit mass of material and can be added to the material's LCA: 0.11  $\mu\text{DALY}_{\text{IAQ}}/\text{kg}$  for gypsum board and 0.02  $\mu\text{DALY}_{\text{IAQ}}/\text{kg}$  for bamboo flooring. These results are specific to the case study. For different ACH, temperatures, material age and thicknesses, impacts are expected to change. In the next section, a simplified sensitivity analysis will help identify important parameters determining health impacts related to the emission of VOCs from materials.

### 3.4.3. Sensitivity analysis

The effect of different parameters on the release of VOCs from gypsum board and health impacts generated are going to be studied: 1) ventilation rates, 2) temperatures, 3) material ages and 4)



material thicknesses using the Morris method expressed in equation (2-9). Results correspond to the case study, with concentration curves for one substance (toluene) and health impacts for all substances emitted.

**Ventilation**

The ventilation rate (or air changes per hour, ACH) is the main mode of the removal of pollutants, often large compared to adsorption and degradation in indoor spaces (Hellweg et al. 2009) and is thus expected to affect exposure and impacts. Figure 2-10 shows (a) the concentration curves and (b) health damages related to the intake of VOCs emitted by gypsum board for different ventilation rates ranging from 0.2 ACH to 2 ACH.

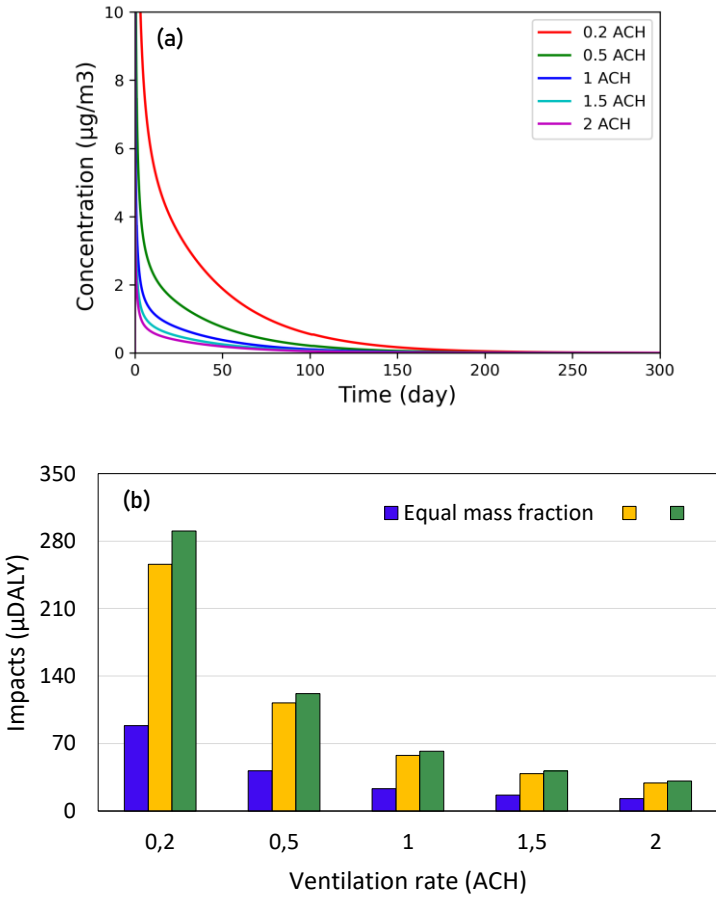


Figure 2-10: (a) Concentration curves for toluene emitted by gypsum board at different air changes per hour from 0.2 to 2 vol/h and (b) health damages related to the intake of VOCs emitted by gypsum board for different ventilation rates in the office

Impacts decrease almost proportionally to increase in ventilation rates for all time scales. Even though emission rates increase with increased ventilation rates, the rate of evacuation is higher than the rate of emission and concentrations thus decrease. The sensitivity  $\omega$  of health impacts ( $\mu\text{DALY}$ ) to

ventilation ( $h^{-1}$ ) for the three time frames is equal to 16.3  $\mu$ DALYh, 47.1  $\mu$ DALYh and 52.6  $\mu$ DALYh, indicating that long-term impacts are affected most.

**Temperature**

Emission dynamics are dependent of temperature. The higher the temperature, the higher the diffusion coefficient and the lower the partition coefficient if other variables remain unchanged (ref: equations (2-1), where  $\tau - 3486$  is negative, and (2-2)) and thus, faster emissions. Indoor temperatures are varied within a reasonable range: from 10°C (room with low insulation and low heating) to 40°C (room with low insulation and low occultation in hot regions) and other parameters remain unchanged. Figure 2-11 (a) and (b) show the concentration curves for toluene emitted by gypsum board and (c) shows health impacts for all substances emitted at the given temperatures.

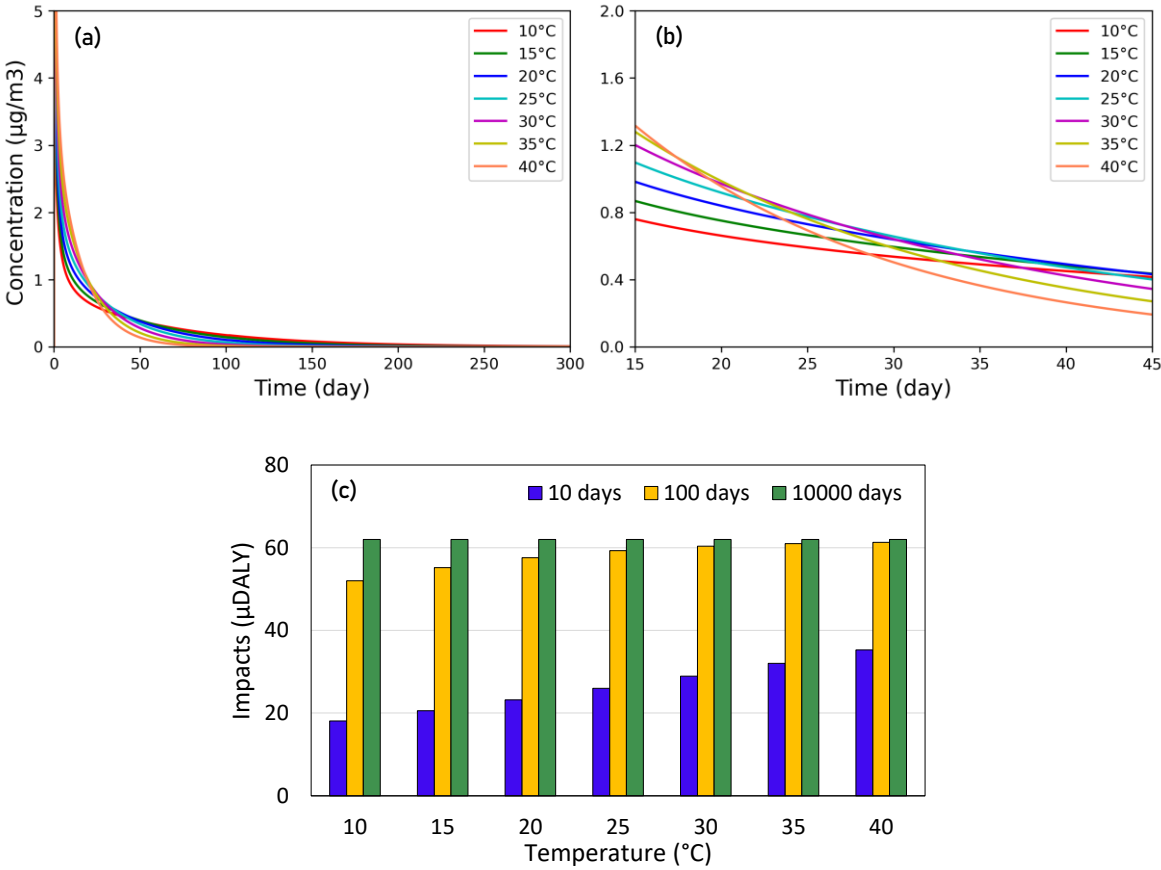


Figure 2-11: (a) Concentration of toluene in air at different temperatures between 10 °C and 40 °C for (a) 0-300 days and (b) 14-30 days and (b) Health impacts related to the intake of substances emitted by gypsum board for three different time scales: 10 days, 100 days and 10000 days and for the same temperatures

From Figure 2-11 (a) and (b), with a change in temperature, we observe a change in the emission dynamic, defined by the rate of change of concentration, affected by the emission rate  $ER(t)$ . Concentrations at high temperatures tend to be higher at the beginning and eventually decrease beyond concentrations at lower temperatures (e.g. at about 28 days the concentration at 40°C becomes lower

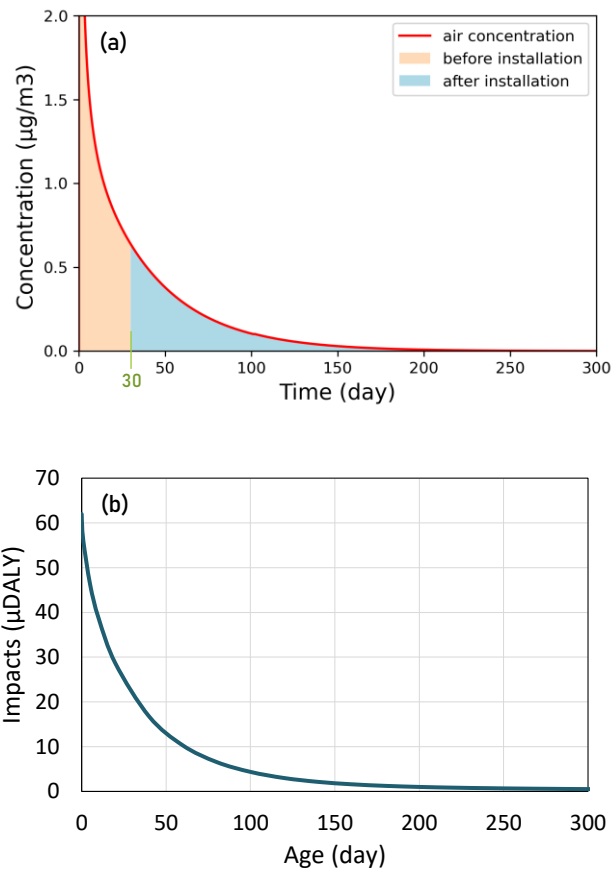
than at 10°C), which can be explained by the following: since the mass of substance in the material  $M_s(t)$  initially decreases faster for higher temperatures due to higher emission rates  $ER(t)$ , the remaining mass inside the material at any point in time ( $t > 0$ ) would be lower for higher temperatures.  $ER(t)$ , dependent of  $M_s(t)$ , also decreases with time. With other parameters constant, air concentrations also decrease with decreasing  $ER(t)$ , until they become lower at higher temperatures. Concentrations at all temperatures tend towards zero when the substances reach equilibrium within and outside the material or when they are entirely emitted.

Figure 2-11 (c) shows that, for lower time frames, the difference in health impacts for varying temperatures are more important than for higher time frames. There is up to 94% increase in impacts between 10 °C and 40 °C for a 10-day time scale while this difference is close to zero over 10000 days. This is reflected in the sensitivity of health impacts to changes in temperature which are higher for shorter time frames:  $1.3 \times 10^{-2} \mu\text{DALY/K}$  for short-term,  $6.2 \times 10^{-3} \mu\text{DALY/K}$  for mid-term and  $4.1 \times 10^{-5} \mu\text{DALY/K}$  for long-term impacts.

### Material age

In the case study, we consider the  $MF_0$  of VOCs in the office materials to be equivalent to those of the sample studied in test chambers. We define the “age” of the material as the time between the beginning of chamber tests (for which we have calculated  $MF_0$ ) until its installation indoors. Since the material’s age determines the amount of VOC that has been emitted and the mass remaining inside, it has an influence on indoor air concentrations and health impacts.

The air concentrations of toluene and health impacts resulting from the intake of VOCs emitted by gypsum board in an office for different material ages are presented in Figure 2-12 (a) and (b).



*Figure 2-12: (a) Air concentrations of toluene emitted by gypsum board in an office and (b) health impacts from all substances emitted for different material ages (<1 to 300 days)*

Figure 2-12 (a) shows that the indoor concentration of toluene released by monolayered gypsum board decreases with time, until reaching very low and near-to-zero concentrations after about 200 days. Consequently, concentrations to which occupants are exposed decrease if the age of the material is higher. The example of a material age of 30 days is shown in Figure 2-12 (a): the pale orange area shows the concentrations to which occupants are not exposed over 30 days and the blue-shaded area shows concentrations to which occupants are exposed ( $\mu\text{g}\cdot\text{s}/\text{m}^3$ , concentration  $\times$  time). From Figure 2-12 (b), we note that, as expected from the concentration curve and the PiF (equation (2-8)), intake and health impacts decrease with increasing material age. The decrease is more important in the beginning and eventually tends towards zero. There is a 13-fold decrease from 62  $\mu\text{DALY}$  at age 0 days to 4.4  $\mu\text{DALY}$  at age 100 days.

The sensitivity over short, medium or high material ages, corresponding to up to 10 days, 10-100 days and 100-1000 days are calculated: 5.62  $\mu\text{DALY}/\text{d}$ , 0.99  $\mu\text{DALY}/\text{d}$  and 0.15  $\mu\text{DALY}/\text{d}$  for short, medium and high material ages.

## Material thickness

Two possibilities are studied for variations in material thickness. In the first, the mass fraction  $MF_0$  ( $\text{kg}_{\text{substance}}/\text{kg}_{\text{material}}$ ) is kept identical for all material thicknesses. The second consists in variations of the material thickness for an equivalent initial substance mass,  $M_{s_0}$  ( $\text{kg}_{\text{substance}}$ ) (e.g. for the application of chemicals relative to the material surface and not volume). The initial mass of the substance inside the material is given by the product of  $MF_0$ , its volume  $V_{\text{material}}$  ( $\text{m}^3$ ) and density  $\rho_{\text{material}}$  ( $\text{kg}/\text{m}^3$ ). Thus, for thicker materials with equal surface area,  $M_{s_0}$  is higher in the first case and  $MF_0$  is smaller in the second case. Figure 2-13 (a) and (b) show concentrations and health impacts for varying material thickness with constant  $MF_0$  and Figure 2-13 (c) and (d) show concentrations and health impacts for different material thicknesses and constant  $M_{s_0}$ .

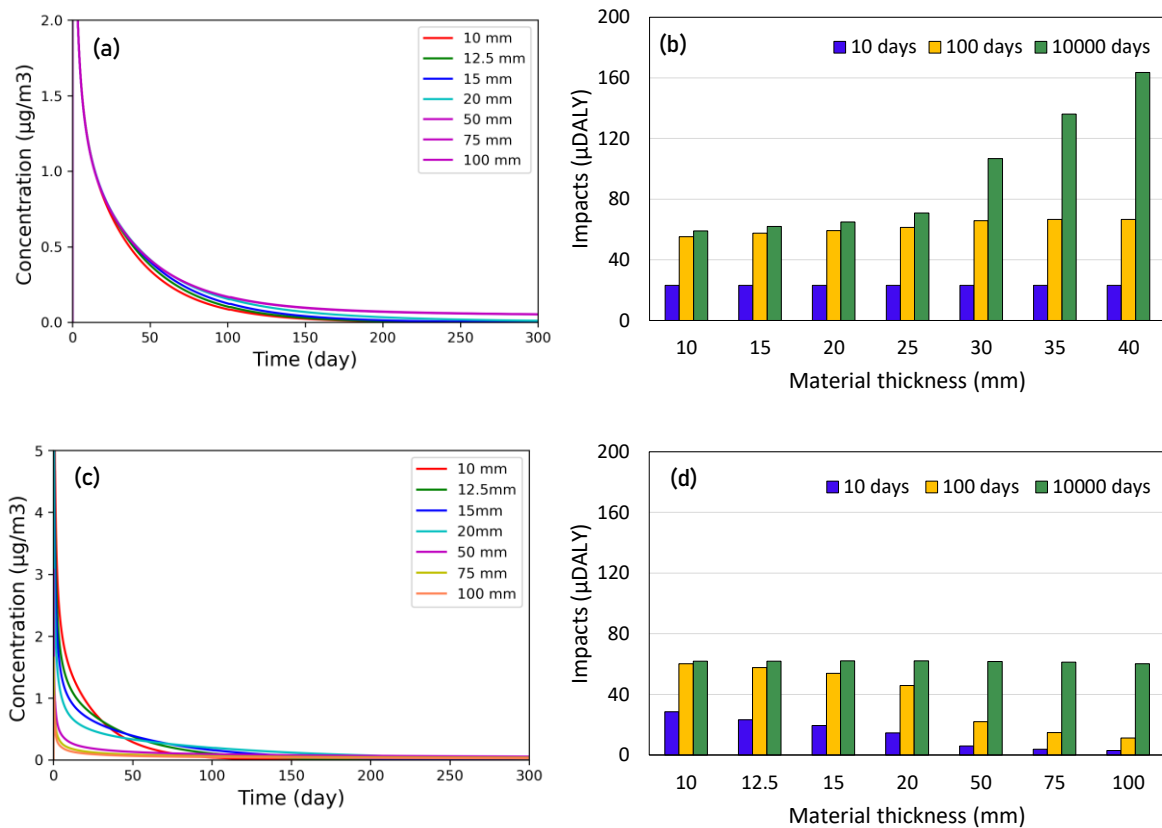


Figure 2-13: Air concentration of toluene emitted by gypsum board in an office for different material thicknesses ranging from 10 mm to 100 mm (a) for equal mass fractions and (c) for equal initial mass of substance in the material and consequent health damages for the different thicknesses and three time frames: 10, 100 and 10000 days (b) for equal mass fractions and (d) for equal initial mass of substance in the material

An increase in material thickness for equal  $MF_0$  results in higher air concentrations, and consequently higher impacts. Figure 2-13 (a) and (b) show that on the short-term (10 days), concentrations and health impacts are similar for all thicknesses and differences start to grow in higher time frames: initially, VOC particles closer to the surface of the material escape faster (in the same way

for the different material thicknesses) while those further away (in higher thicknesses) diffuse through the material before being emitted. The sensitivity of material thickness to health impacts for equal  $MF_0$  for short, medium and long-term exposure are 0.01  $\mu$ DALY/mm, 1.05  $\mu$ DALY/mm and 4.92  $\mu$ DALY/mm. Long-term impacts are most sensitive to changes in thickness.

For constant initial mass  $M_{s_0}$ , only short-term and mid-term impacts are affected with variations in thicknesses, as shown in Figure 2-13 (c) and (d). Since the substance's mass concentration  $MF_0$  is lower within thicker materials, diffusion is longer through the material layers, hence flattening the concentration curve. Long-term health impacts follow a slight decrease (from 61.95  $\mu$ DALY to 60.28  $\mu$ DALY) from 10 mm to 100 mm thicknesses, since the release is slower for thicker materials:  $3.23 \times 10^{-12}$   $\mu$ g of VOCs remain in the material of 10 mm thickness, while  $7.11 \times 10^5$   $\mu$ g remains for 100 mm thickness after 10000 days. The sensitivity of material thickness to health impacts for equal  $M_{s_0}$  for short, medium and long-term exposure are 0.6  $\mu$ DALY/mm, 1.0  $\mu$ DALY/mm and 0.01  $\mu$ DALY/mm.

A summary of the parameters and their sensitivity index  $\omega$  is given in Table 2-4.

*Table 2-4: Summary of the sensitivity index ( $\omega$  based on the Morris method) of four model parameters on health impacts: ventilation rate, temperature, material age and material thickness*

Parameter	$\omega_{\text{short-term}}$	$\omega_{\text{mid-term}}$	$\omega_{\text{long-term}}$
Ventilation rate ( $\mu$ DALY/(vol/h))	$1.63 \times 10^1$	$4.71 \times 10^1$	$5.26 \times 10^1$
Temperature ( $\mu$ DALY/K)	$1.28 \times 10^{-2}$	$6.20 \times 10^{-3}$	$4.10 \times 10^{-5}$
Material age ( $\mu$ DALY/d)	$5.62 \times 10^0$	$9.86 \times 10^{-1}$	$1.51 \times 10^{-1}$
Material thickness, equal $MF_0$ ( $\mu$ DALY/mm)	$1.40 \times 10^{-2}$	$1.05 \times 10^0$	$4.92 \times 10^0$
Material thickness, equal $M_{s_0}$ ( $\mu$ DALY/mm)	$6.12 \times 10^{-1}$	$1.00 \times 10^0$	$1.05 \times 10^{-2}$

The parameters have not been normalised and cannot be compared to each other, but the short, medium or long-term effects can be compared: ventilation rates and material thickness for equal  $MF_0$  have more important effects on the long-term, while other parameters have higher effects on the short-term.

#### 4. Conclusion, limitations and perspectives

This work demonstrated a framework for the calibration of an emission model in order to fix parameters having relatively high uncertainties (factor 10 to 100) and calculate unknown substance initial mass fractions within the material. This method allows to 1) extrapolate measured emission data

for specific materials (only valid at few points in time and for short time spans), 2) scale concentrations to real building contexts and 3) calculate long-term occupant exposure and health impacts.

The framework was tested on two mono-layered materials: gypsum board and bamboo flooring, yielding an uncertainty factor of 1.1 based on the RMSLE (root mean square log of error). Optimal parameters and mass fractions were coupled with building parameters to obtain the full concentration and exposure profiles for all substances emitted and calculate short, medium and long-term health impacts: 26  $\mu$ DALY, 65  $\mu$ DALY and 70  $\mu$ DALY. Through an exploratory approach, it was concluded from a test with unvarnished plywood that the model needs to be adapted further to account for changing ventilation rates and precisely match real-life experimental set-up. Its applicability to multilayered materials was also tested on a PVC flooring. The framework allows to identify layers in which substances are more likely to be initially present, but sufficient data on the material's thickness are required. If the initial positions of substances are known, the framework can be used to fix model parameters for each material layer.

Through sensitivity tests, it was concluded that ventilation rates, mass thickness (for equal mass fraction), temperature and material age could have considerable effects on the emission profile and consequently on health impacts on the short-term (especially for temperature, age and thickness) and long-term (for ventilation rates). These tests indicate that, in order to reduce health impacts of VOCs emitted by building materials, the following measures, for short durations, could be taken to evacuate most VOCs before leaving the factory (assuming that emitted gases are evacuated or degraded adequately to avoid health impacts during treatments):

- Treating materials with high heat (rapid release of substances), but at a given environmental cost linked to energy consumption
- Keeping materials for a certain duration in a highly-ventilated room (increasing their age before installation)
- Coupling the two above measures that involve temperature, age and ventilation rates to achieve low-emission materials
- Reuse of materials (e.g. second-hand furniture) where a major part of VOCs has most certainly been released

Results of this study are valid for given assumptions, in particular VOC emissions from mono-layered materials and uniform conditions (ACH, pressure and temperature) from the chamber tests to building use, but construction materials do not have the same fate as tested materials. They can go through factory, warehouse, hardware store and finally construction site and experience temperature, pressure and ACH variations. If possible, these should be included in the study to obtain real occupant exposure. For multi-layered materials, emission dynamics are more complex and depend on the properties of all materials: substances can sorb from one material to another and into air, changing the

emission dynamics and peaks can occur after a few days (Piasecki, Kostyrko, and Goljan 2021). In general, materials such as gypsum board are coated with surface finishes (paint or varnishes) which should also be included in health impact assessments. To define ideal temperatures for the material's treatment and ideal material age at installation, a prior study on the emission profile of the VOCs emitted by the specific material and environmental conditions is also required.

This method is dependent on emission data, but these are often confidential or incomplete. Due to regulations and rising concern around material VOCs, these data are expected to become more readily available in the future. Furthermore, health impacts are possibly underestimated since toxicity data are lacking for a considerable number of substances and should be updated when available. Another uncertainty lies in the possible chemical reactions occurring between VOCs and precursors such as O<sub>3</sub> and NO<sub>2</sub>, leading to the creation or consumption of substances (C. J. Weschler 2011). These results can thus be used for ecodesign in building construction or material manufacture to choose between materials or paints/varnishes having lower impacts on health, considering both their IAQ impacts and those related to their extraction or transformation. The framework can also help dimension adequate ventilation to evacuate substances.





# 3. Health impacts of VOCs from occupants and indoor activities

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## Résumé en français

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Dans ce chapitre, les impacts liés aux composés organiques volatils (COV) émis par des occupants et des activités en intérieur sont étudiés. Une méthode est proposée pour suivre le parcours des polluants depuis l'émission jusqu'aux effets sur la santé : des taux d'émission de COV sont intégrés au modèle INCA-Indoor pour calculer leurs concentrations dans l'air, puis l'assimilation et les impacts sur la santé en DALY (*disability-adjusted life years*) sont évalués avec le modèle USEtox. Les effets de réactions chimiques dans l'air intérieur sont aussi étudiés.

La méthode est appliquée à un bureau occupé par trois personnes, avec l'utilisation de produits ménagers émettant des COV dans l'air. Les résultats indiquent que les impacts liés au nettoyage sont plus importants en été qu'en hiver : 0.16  $\mu$ DALY/jour contre 0.15  $\mu$ DALY/jour (5 secondes/jour), étant donné que les réactions sont plus importantes (températures plus élevées et plus d'ozone). Sans réactions chimiques, les impacts s'élèvent à 0.11  $\mu$ DALY/jour (soit 31% en moins en été et 25% en moins en hiver).

La contribution des occupants est négligeable pour les COV mais importante pour le CO<sub>2</sub> : les concentrations sont supérieures aux limites recommandées durant toute l'occupation du bureau, mais il n'y a pas de facteur d'effet associé. Finalement, on constate que les formations de particules fines par coagulation et nucléation augmentent les concentrations massiques d'un facteur 1.2 et les concentrations en nombre d'un facteur allant jusqu'à 25000. Nous en concluons que les réactions chimiques dans l'air intérieur ont une influence non-négligeable sur le calcul d'impacts de la qualité de l'air intérieur et qu'une étude plus profonde de leurs effets sur la santé est recommandée.

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## Abstract

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Occupants and indoor activities are sources of volatile organic compounds (VOCs). We propose a framework to simulate the pollutant pathway using VOC emission rates, the INCA-Indoor model to derive dynamic concentrations, and the USEtox model to evaluate health impacts in DALYs (Disability-Adjusted Life Years). We test the applicability of the framework on a case study, and assess the effect of indoor chemical reactions on health impacts. Cleaning activities led to the highest impacts, while occupants' contribution to VOC concentrations were negligible since emission rates are much lower than for activities. However, their contribution to CO<sub>2</sub> concentrations exceeded recommended limits. Health impacts were affected by chemical reactions, different according to seasons: 0.16  $\mu$ DALY/day (5 seconds/day) in summer when ozone-dependant chemical reactions are more important against 0.15  $\mu$ DALY/day in winter. D-limonene, emitted by cleaning products, was the highest contributor to health impacts, followed by formaldehyde, which was a product of indoor chemical reactions. Without these reactions, health impacts amount to 0.11  $\mu$ DALY/day for both seasons (31% lower in summer and 25% lower in winter). Secondary organic aerosol (SOA) formation affected indoor particulate matter mass concentrations by up to a factor 1.2 and their number concentrations by up to a factor 25000 in the presence of VOC emissions. Our results show that chemical reactions and SOA formation are important factors to consider in indoor air quality impact assessment.

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## 1. Introduction

Occupants and indoor activities were identified as sources of pollutants, contributing to the presence of CO<sub>2</sub>, volatile organic compounds (VOCs) and particulate matter indoors (William W. Nazaroff and Weschler 2004; Mochalski et al. 2014; Tang et al. 2016; Licina et al. 2016; Persily and de Jonge 2017; Kruza and Carslaw 2019b). Direct contributions to indoor VOCs include occupant skin (Mochalski et al. 2014) and breath (Sun, He, and Yang 2017) which are dependent on diet, metabolism, smoking habits, use of perfumes and cosmetics and cutaneous reactions. Activities contributing to these emissions include the use of air fresheners and cleaning products in which VOCs are used to prevent bacteria formation and mask or add odours (William W. Nazaroff and Weschler 2004).

Emissions from occupants and indoor activities occur in relatively small volumes with low air renewal rates and high surface-to-volume ratios. Thus, resulting VOC concentrations can be higher than acute toxicological value thresholds, especially for cleaning (Karr et al. 2021). Furthermore, indoor environments are particularly favourable to chemical reactions between gases in air, such as ozone and VOCs (in particular terpenoids), producing secondary VOCs or organic aerosols (SOA). Studies suggest that their effects on health could be non-negligible, but they are complex due to the large variety of substances present in air and high number of possible reactions (William W. Nazaroff and Weschler 2004; C. J. Weschler 2011; W. W. Nazaroff and Goldstein 2015; Charles J. Weschler and Carslaw 2018). The INCA-Indoor model considers chemical reactions between different species in air concentration calculations (Mendez et al. 2015; Berger et al. 2023).

The main mode of evacuation is ventilation: ventilating after activity can help reduce concentrations by a factor of 10 to 1000 (Karr et al. 2021). However, high ventilation rates could lead to higher energy consumption for ventilation fans or for heating in winter and thus increase environmental impacts. Ideal ventilation strategies could thus be elaborated by including occupant and activity related impacts in life cycle assessment (LCA) to avoid the transfer of impacts. Environmental impacts of different cleaning products have been calculated using LCA, but no link was made with the exposure to VOCs released in the *operation* stage (Van Lieshout et al. 2015). Indoor air concentrations and emission rates of chemicals have been measured for breath and skin (Fenske and Paulson 1999; Sun, He, and Yang 2017; Kruza and Carslaw 2019b; Zou, He, and Yang 2020) and different activities (William W. Nazaroff and Weschler 2004; Singer et al. 2006; Y. Huang, Ho, Ho, Lee, Gao, et al. 2011), but their impacts on health have not been evaluated. Some studies aimed at assessing exposure by inhalation from measurements or simulations (Zhu, Cao, and Beauchamp 2001; Dimitroulopoulou et al. 2015) and calculating inhalation intake fractions based on building and occupant factors (William W. Nazaroff 2008). A health risk assessment has been conducted for several cleaning products (Karr et al. 2021), but not for occupant VOC emissions and with no link to LCA. In order to account for occupant

and activities VOC emissions in LCA, the entire pollutant pathway has to be studied and impacts have to be calculated using units common to LCA indicators.

The main objective of this chapter is to present a framework linking emission rates from occupants and activities to exposure concentrations and health damage. The framework is applied on a case study with the following steps:

1. Collect breath, skin, clothes and activity VOC emission rates for a given activity scenario.
2. Calculate indoor VOC concentrations for given activity scenario, emission rates and building parameters using INCA-Indoor.
3. Evaluate health damages by coupling concentrations to occupant exposure via different intake pathways and effect factors.

We also explore the effects of indoor air chemistry through the INCA-Indoor model simulations in order to assess the importance of these reactions on occupant health.

## 2. Materials and methods

### 2.1. General framework

The proposed framework for the calculation of health damages resulting from exposure to indoor VOCs from occupants and indoor activities is presented in [Figure 3-1](#).

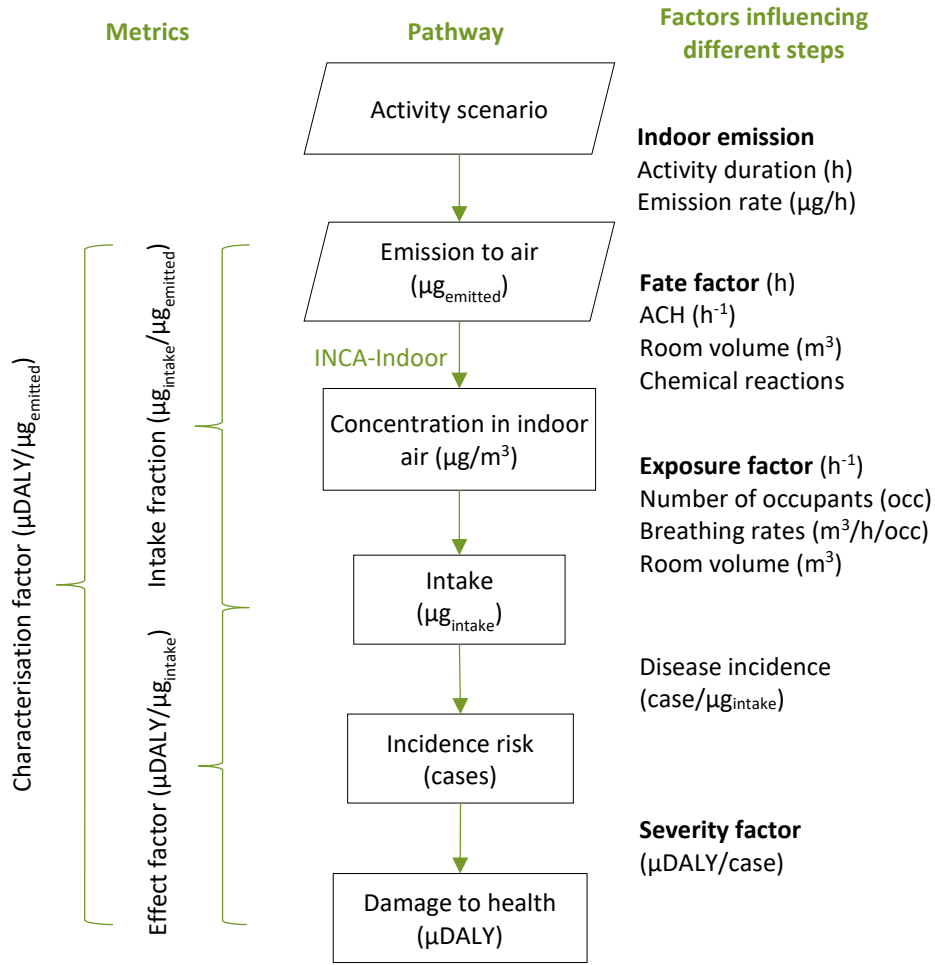


Figure 3-1: Framework for evaluating health damage from occupants and activities: linking emission rates to exposure and health damage

An activity scenario is defined using identified activities with available emission rates ( $\mu\text{g}/\text{h}$ ) from literature. Based on the scenario and building characteristics (air flow rates and room volume) modelled on Pleiades (IZUBA ÉNERGIES 2001a), meteorological conditions (wind speed, temperature) and outdoor pollution levels, indoor air concentrations are calculated by INCA-Indoor, coupled with CONTAM for the airflow modelling. Dynamic concentrations are coupled with exposure factors (which depend on occupancy scenario) to calculate the intake  $M_{\text{in,path}}$  ( $\mu\text{g}_{\text{intake}}$ ) for four different pathways: inhalation, ingestion, direct dermal contact and gaseous dermal uptake (Micolier 2019; Rosenbaum, Margni, and Jolliet 2007; L. Huang, Ernstoff, et al. 2017). Health impacts for each substance are evaluated using  $M_{\text{in,path}}$  and the effect factor  $EF$  ( $\mu\text{DALY}/\mu\text{g}_{\text{intake}}$ ) for each path. The total intake  $M_{\text{in,path}}$  of a substance is calculated by equation (3-1) below for a given intake pathway:

$$M_{\text{in,path}} = V_{\text{room}} \int_{t=0}^T XF_{\text{path}} \times C_{\text{in}} dt \quad (3-1)$$

Where  $V_{\text{room}}$  ( $\text{m}^3$ ) is the room volume,  $C_{\text{in}}$  ( $\mu\text{g}/\text{m}^3$ ) is the indoor air concentration of the substance,  $T$  (h) is the duration of exposure and  $XF_{\text{path}}$  ( $\text{h}^{-1}$ ) is the exposure factor of all occupants for a given pathway, described by equations (1-2), (1-3), (1-4) and (1-5). Health impacts are calculated using the USEtox model (Fantke, Bijster, et al. 2017). The total health impacts  $HI$  (DALY) is given by the product of the effect factor  $EF$  (DALY/ $\text{kg}_{\text{intake}}$ ) and the total mass intake  $M_{\text{in}}$  ( $\text{kg}_{\text{intake}}$ ) of the substance for each pathway.

$$HI = \sum_{\text{path}} M_{\text{in,path}} \times EF_{\text{path}} \quad (3-2)$$

The effect factor of a substance can be obtained from its  $ED_{10}$  or  $ED_{50}$  (the lifetime doses per person that cause a disease probability of 10% and 50% after intake), as described in A1.3.

## 2.2. Case study description

A  $37 \text{ m}^3$  office occupied by 3 persons during working days from 8:30-12:30 and 13:30-18:00 is used as case study. Common activities, identified as important contributors to indoor VOCs, are considered: cleaning and using air fresheners (William W. Nazaroff and Weschler 2004; Singer et al. 2006). The room is cleaned once per week in the morning and occupants use air fresheners at 14:00. A fixed air renewal rate of  $1 \text{ h}^{-1}$  is considered, with a standard infiltration rate of  $1.7 \text{ m}^3/\text{m}^2\cdot\text{h}$  under 4 Pa, corresponding to the low-end of average tertiary buildings (Carrié et al. 2006).

## 2.3. Emission data

Data obtained from various studies measuring VOC emission rates from occupants and activities and the durations are summarised in Table 3- 1. The list of emitted substances and their emission rates are presented in A2.1.



Table 3- 1: Activities considered in the office and their durations

Emission Source	Reference	Duration (min)
Skin	Mochalski et al. 2014	During occupant presence
Exhaled air	Riess et al. 2010; Persily and de Jonge 2017	During occupant presence
Shoes	Kurosawa et al. 2008	During occupant presence
Air freshener	Singer et al. 2006	1 <sup>a</sup>
Counter cleaner	Singer et al. 2006	10
Floor cleaner	Singer et al. 2006	15

- a. INCA-Indoor has a minimum time step of one minute, hence the use of a few seconds of deodorant cannot be simulated

The contribution of occupants to indoor CO<sub>2</sub> emission is also studied. The emission rate  $V_{CO_2}$  (L/s) can be determined by the following relationship established by Persily and de Jonge (2017) as a function of basal metabolic rate  $BMR$  (MJ/day) and metabolic rate  $M$  (met):

$$V_{CO_2} = 1.79 \times 10^{-4} \times BMR \times M \times (T/P) \quad (3-3)$$

where  $T$  (K) is the air temperature and  $P$  (kPa) the pressure. The mass of CO<sub>2</sub> generated can be calculated from its density of 1.84 g/L. The  $BMR$  according to age and sex can be obtained from the same study and  $M$  can be determined according to the activity intensity (Harvard School of Public Health 2012). Since office activities are generally of low intensity, we consider no significant variation in the amount of CO<sub>2</sub> released per person. The average of female and male CO<sub>2</sub> emissions for the 21-30 age group are calculated for 1.2 met activity intensity, corresponding to office work, resulting in 28 g/h/occ.

#### 2.4. Simulation using INCA-Indoor

The INCA-Indoor multizone model uses building parameters (e.g. room volume, infiltration rates), occupancy scenarios, activity emission rates, indoor gas phase chemistry, air renewal rates and sorption to material surfaces to calculate indoor concentrations of gases and particulate matter. Air flows are simulated using CONTAM (Dols and Polidoro 2020) considering infiltration, exfiltration and flows between rooms based on indoor and outdoor pressures and temperatures (Mendez et al. 2015). Indoor temperatures are considered to be constant and equal to 20 °C. Air is treated as an ideal gas, obeying to the ideal gas law. Gas-Surface and Particle-Surface interactions are treated by describing a boundary layer scheme, using SAPRC-07 detailed chemical mechanism involving about 800 VOCs and their reactivity with the main oxidants (OH·, O<sub>3</sub>, NO<sub>3</sub>·, etc) described in Mendez et al. (2015).

It is important to note that not all substances emitted by the activities considered are treated by INCA-Indoor: out of the 79 different substances emitted by occupants and activities altogether, 51 of them are treated by INCA-Indoor and are considered in the results. In particular, there are 37 out of 49 substances treated for skin emissions, 12 out of 15 for breath, 2 out of 3 for shoes, 3 out of 9 for air freshener and 6 out of 16 for the counter and floor cleaners. It is thus expected that exposure will be underestimated.

### 2.3.1. Scenarios

INCA-Indoor can simulate concentrations without indoor emissions or be adapted to account for emissions from activities or from occupants. Simulations can include precursors to indoor chemical reactions: ozone (O<sub>3</sub>) or nitrogen oxides (NO and NO<sub>2</sub>). They react with VOCs (Mendez et al. 2015), and their concentrations, affected by seasons, can also be set to zero throughout the simulation in order to evaluate their influence. INCA-Indoor can include secondary organic aerosol (SOA) formation through condensation and nucleation (Berger et al. 2023). Other physical processes occurring to particulate matter such as coagulation and deposition have also been validated by Berger et al. (2023). Results are presented for different combinations, summarised in Table 3-2.

*Table 3-2: Simulated scenarios according to season and consideration of indoor air chemistry or secondary organic aerosol formation*

Scenario name	Definition
act	Emissions from activities
occ	Occupant direct emissions from skin, breath and clothes
inor	Presence of inorganic precursors: O <sub>3</sub> and NO <sub>x</sub>
noinor	No O <sub>3</sub> and NO <sub>x</sub>
summer	Calculation in summer season
winter	Calculation in winter season
emi	With indoor emissions (activities and occupants)
noemi	Without indoor emissions
SOA	With secondary organic aerosol formation

## 3. Results

### 3.1. Primary VOC concentration curves

INCA-Indoor can evaluate concentrations for a desired period, usually set at one week. A duration of one day is selected for this study to lower calculation times, since 24 hours are sufficient to

evacuate all emitted/produced pollutants and observe concentration profiles; Results are thus representative of one working day. Due to the large number of substances (64) and for ease of comprehension, only those with the highest concentrations and/or leading to significant impacts are presented.

Indoor activities (cleaning and the use of air freshener) contribute to the increase of mainly two substances: d-limonene and terpinolene. The different scenarios presented in Figure 3-2 are:

- 1) concentrations with chemical reactions with inorganic gases  $O_3$  and  $NO_x$  (*inor*)
- 2) concentrations without these reactions (*no inor*)
- 3) concentrations related to occupant contributions only (*occ*)
- 4) concentrations without indoor emissions (*no emi*)

Results are presented for a winter day. Figure 3-2 shows the indoor air concentrations of (a) terpinolene and (b) d-limonene which increase rapidly during cleaning by about  $570 \mu\text{g}/\text{m}^3$  for both substances, then follow an exponential decay due to a evacuation by air renewal.

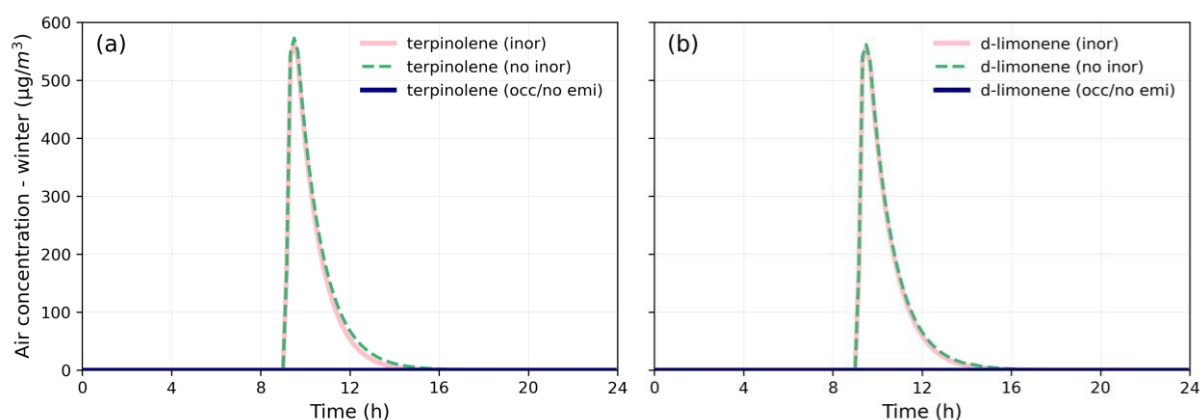


Figure 3-2: Air concentrations of two main VOCs (a) terpinolene and (b) d-limonene emitted by occupants and activities with the presence of  $O_3$  and  $NO_x$  (*inor*), without  $O_3$  and  $NO_x$  (*no inor*), and two equal scenarios: by occupants only or without indoor emissions (*occ/ no emi*)

The peak of terpinolene concentration decreases from  $573 \mu\text{g}/\text{m}^3$  without reactions to  $563 \mu\text{g}/\text{m}^3$  with reactions and d-limonene from  $561.5 \mu\text{g}/\text{m}^3$  to  $557.5 \mu\text{g}/\text{m}^3$ , corresponding to 1.7% and 0.7% change respectively. For health impact assessment in LCA, the area under the curve, representing the exposure to a certain concentration over a given duration, is of particular interest (see equation (3-1)). The maximum possible exposure (total area) increases with chemical reactions, from  $2.90 \times 10^6 \mu\text{g}\cdot\text{s}/\text{m}^3$  to  $3.15 \times 10^6 \mu\text{g}\cdot\text{s}/\text{m}^3$  for toluene (9% increase) and from  $2.96 \times 10^6 \mu\text{g}\cdot\text{s}/\text{m}^3$  to  $3.08 \times 10^6 \mu\text{g}\cdot\text{s}/\text{m}^3$  for d-limonene (4% increase), indicating that these substances are produced during chemical reactions between precursors such as ozone and VOCs. The actual exposure depends on the presence of occupants.

It can also be noted that the presence of these substances is entirely due to activities: occupant emissions of these substances are negligible and comparable to a “no emission” scenario. However, occupants contribute to the presence of CO<sub>2</sub> through breathing, as seen in Figure 3-3. Results are presented in ppm (parts per million, where 1 ppm CO<sub>2</sub> = 1800 µg/m<sup>3</sup>) since it corresponds to the concentration unit generally used for CO<sub>2</sub> and can hence be compared to recommendations.

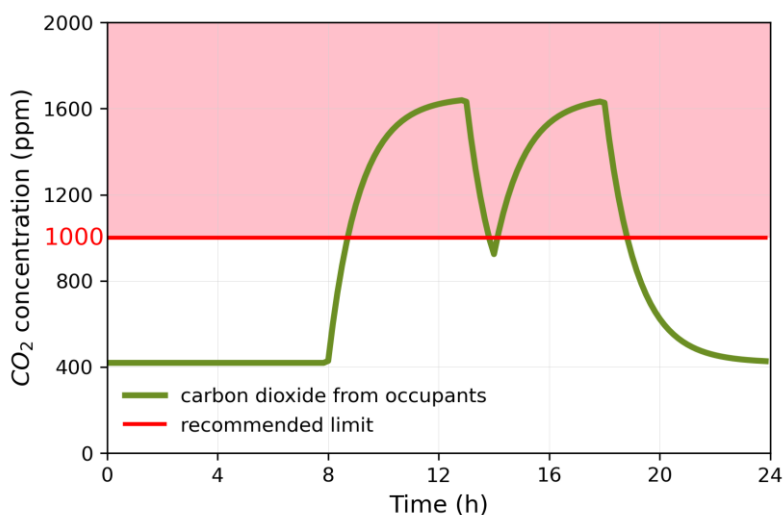


Figure 3-3: CO<sub>2</sub> concentration in air (green curve) and the recommended limit (red line and shaded area)

The CO<sub>2</sub> concentration increases while the occupants are present and decreases during their absence. It can also be noted that, for a large part of the day and during almost the whole occupancy period of the office, CO<sub>2</sub> concentrations are above the recommended limit of 1000 ppm set by different national guidelines (Canada Health 2021; ANSES 2013b). The concentrations remain within an acceptable range where occupants could experience increased drowsiness and lack of concentration, as compared to stronger symptoms such as increased heart rates, headaches or even suffocation occurring at much higher concentrations of over 40000 ppm according to the USDA FSIS Safety (Food Safety and Inspection Service).

### 3.2. Secondary species concentration curves

We explore the importance of indoor air chemistry on calculated concentrations of VOCs and particulate matter. We test for seasonal variations (*summer/winter*), with or without indoor air chemistry formed by inorganic gases: O<sub>3</sub> and NO<sub>x</sub> (*inor/no inor*), with or without secondary organic aerosol (SOA) formation and with or without indoor VOC emissions (*emi/no emi*).

VOCs are precursors to many indoor chemical reactions: they react with O<sub>3</sub> and other oxidants, and hence reactions are dependent on the presence of these inorganic species. The variations in concentrations are shown during winter and summer in Figure 3-4. Figure 3-4 (a) and (b) show the difference in concentrations with and without chemical reactions for three VOCs, O<sub>3</sub>, NO<sub>x</sub> and CO<sub>2</sub>.

Figure 3-4 (a\*) and (b\*) illustrate a focus on concentrations below  $5 \mu\text{g}/\text{m}^3$  due to formaldehyde, a highly toxic substance which lies within this limit, but has considerable health impacts. Negative values mean that concentrations decrease with chemical reactions.

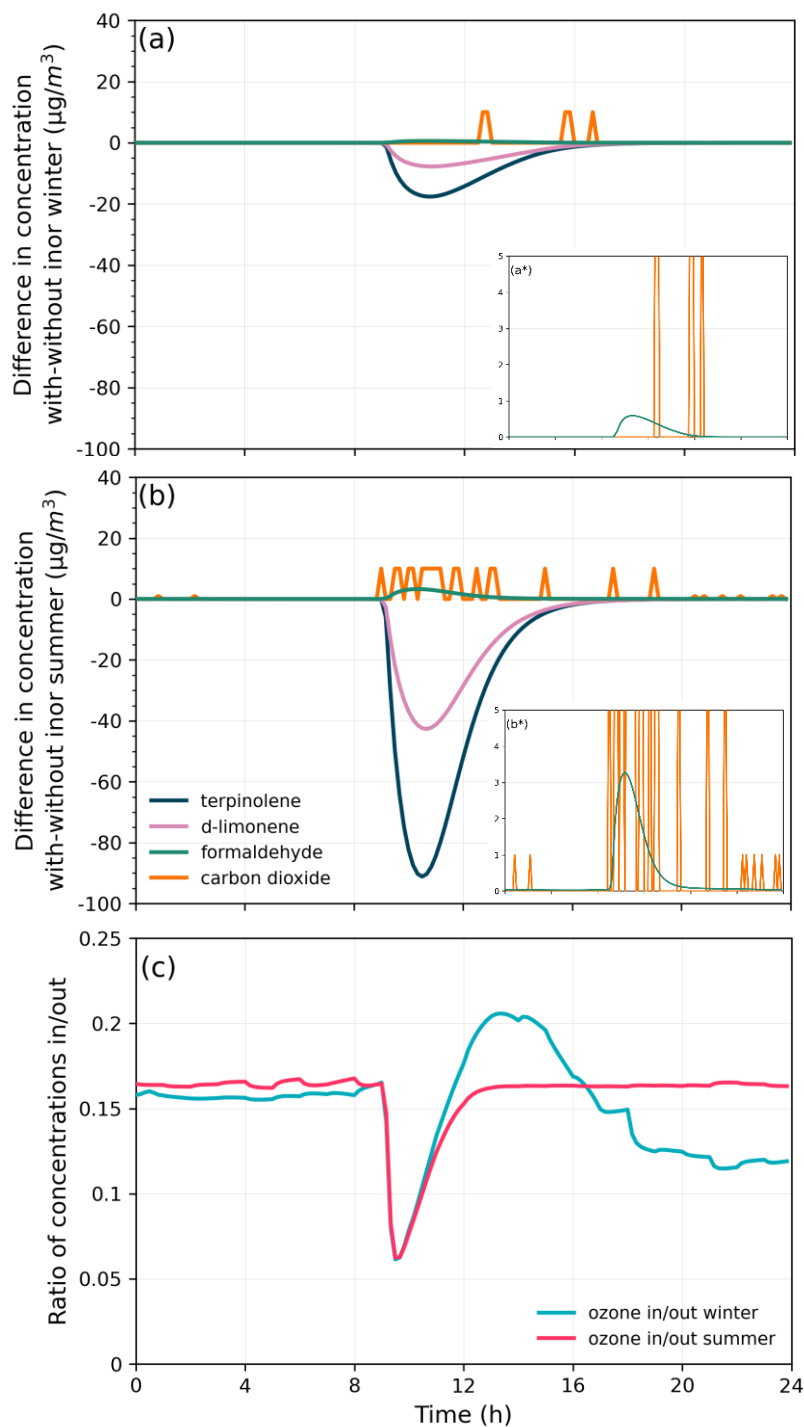


Figure 3-4: Difference in concentrations with and without reactions in (a) winter and (b) summer for three VOCs, carbon dioxide, ozone and nitrogen oxides and (c) indoor concentrations of ozone in winter and summer

Some substances are produced (positive concentration difference), while others are consumed (negative concentration difference) during the reactions. During summer,  $90 \mu\text{g}/\text{m}^3$  (16%) of terpinolene (emitted by cleaning activities) is consumed by indoor chemical reactions, against  $18 \mu\text{g}/\text{m}^3$  (3%) in winter. Without chemical reactions, concentration peaks are equal to  $573 \mu\text{g}/\text{m}^3$  in both seasons. We note the increase in concentration of formaldehyde in Figure 3-4 (a\*) and (b\*), which is only a product

of chemical reactions and not emitted by the indoor activities considered. Formaldehyde is produced by reactions between organic compounds (e.g. isoprene, monoterpenes and oxygenated compounds) and hydroxyl radicals, nitrate radicals or ozone (Salthammer, Mentese, and Marutzky 2010). In particular, d-limonene, beta-pinene and terpinolene, emitted by the activities considered in this study, were found to react with ozone to produce formaldehyde (Atkinson and Carter 1984; Grosjean et al. 1993). Its production is higher in summer (with a peak of  $3.56 \mu\text{g}/\text{m}^3$ ) than in winter (with a peak of  $0.59 \mu\text{g}/\text{m}^3$ ).

Higher effects of indoor air chemistry in summer can be explained by outdoor ozone concentrations which are higher in summer than in winter, with an average of  $98 \mu\text{g}/\text{m}^3$  in summer and  $12 \mu\text{g}/\text{m}^3$  in winter due to higher outdoor temperatures and lower wind speeds favouring its formation in the atmosphere (see [Annex Figure 6](#)). From [Figure 3-4 \(c\)](#), we also note that the ratio between indoor and outdoor ozone are similar in summer and winter, with a sudden drop corresponding to the beginning of activities and VOC emissions. It illustrates that  $\text{O}_3$  is consumed in these chemical reactions.

Formation of secondary organic aerosols (SOA) by nucleation, coagulation or condensation can affect the concentration of particulate matter. In [Figure 3-5](#), the effect of these reactions on air concentrations of PM are shown for (a) winter and summer and (b) with and without indoor VOC emissions.

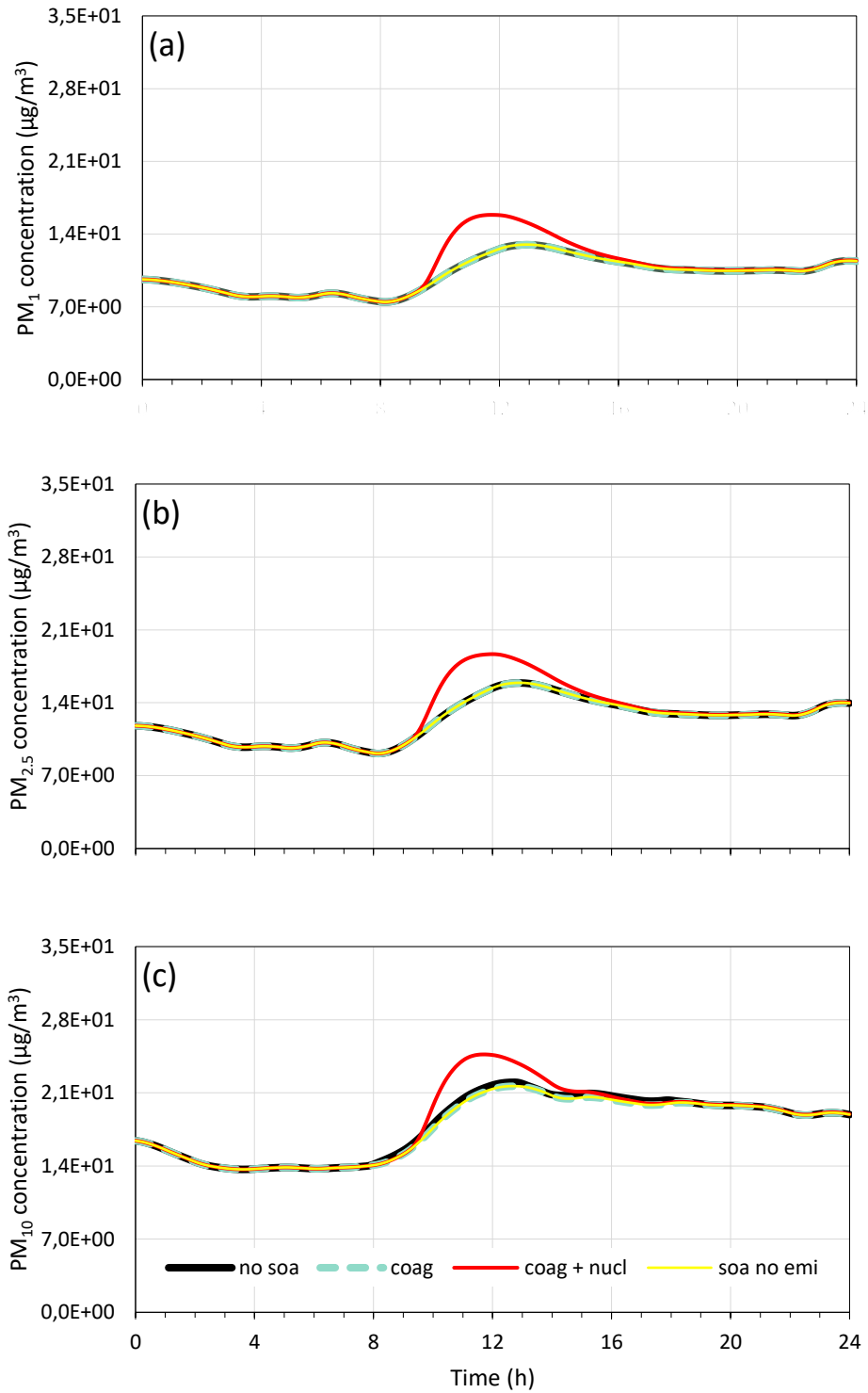


Figure 3-5: Difference in PM concentrations with or without SOA formation by coagulation (coag) and nucleation (nucl) in summer with or without emissions (no emi) for: a)  $\text{PM}_{10}$ , b)  $\text{PM}_{2.5}$  and c) for  $\text{PM}_1$

We note that  $\text{PM}_1$ ,  $\text{PM}_{2.5}$  and  $\text{PM}_{10}$  concentrations increase by the formation of SOA by up to  $2.9 \mu\text{g}/\text{m}^3$ ,  $2.8 \mu\text{g}/\text{m}^3$  and  $2.7 \mu\text{g}/\text{m}^3$  respectively. This change in mass is almost entirely due to the smallest particles of diameter under  $0.12 \mu\text{m}$ , as illustrated in Annex Figure 7. These small, and



consequently light, particles are formed by nucleation. Their formation leads to a significant increase in particle number (by a factor 25000) but a lesser increase in mass (by up to a factor 1.2). These formations are mainly due to nucleation, which coincide with the release of VOCs from activities at 9 a.m. We also note that there is no change in PM concentration in the absence of emissions. The main organic species responsible for SOA formation (as modelled in INCA-Indoor) include d-limonene,  $\alpha$ -pinene,  $\beta$ -pinene, camphene,  $\gamma$ -terpinene and terpinolene, emitted by the cleaning detergent and air freshener. Those SOA precursors are mainly oxidised by OH radicals and O<sub>3</sub>, their oxidation products, having low vapor pressure, tends to condense on existing particles or can even nucleate to form new particles. Gas phase reactivity leading to SOA formation has been adapted from (Couvidat et al. 2012) and adapted to INCA-Indoor by (Berger et al. 2023). Other emitted substances, namely monoterpenes that are found in essential oils and used to enhance odours, are not treated by INCA-Indoor but could influence SOA formation.

### 3.3. Intake quantities and health damages

Intake quantities  $M_{in,path}$  ( $\mu\text{g}_{intake}/\text{day}$ ) are calculated and coupled with available effect factors ( $\mu\text{DALY}/\mu\text{g}_{intake}$ ) for each pathway and each substance to obtain the total health impact per day ( $\mu\text{DALY}/\text{day}$ ) across all substances. Total health impacts are the sum of impacts of substances for which toxicity data are available: 28 out of 57 VOCs studied (51 emitted indoors and 6 from chemical reactions). [Figure 3-6](#) shows the VOC intake quantities and health impacts for seven scenarios.

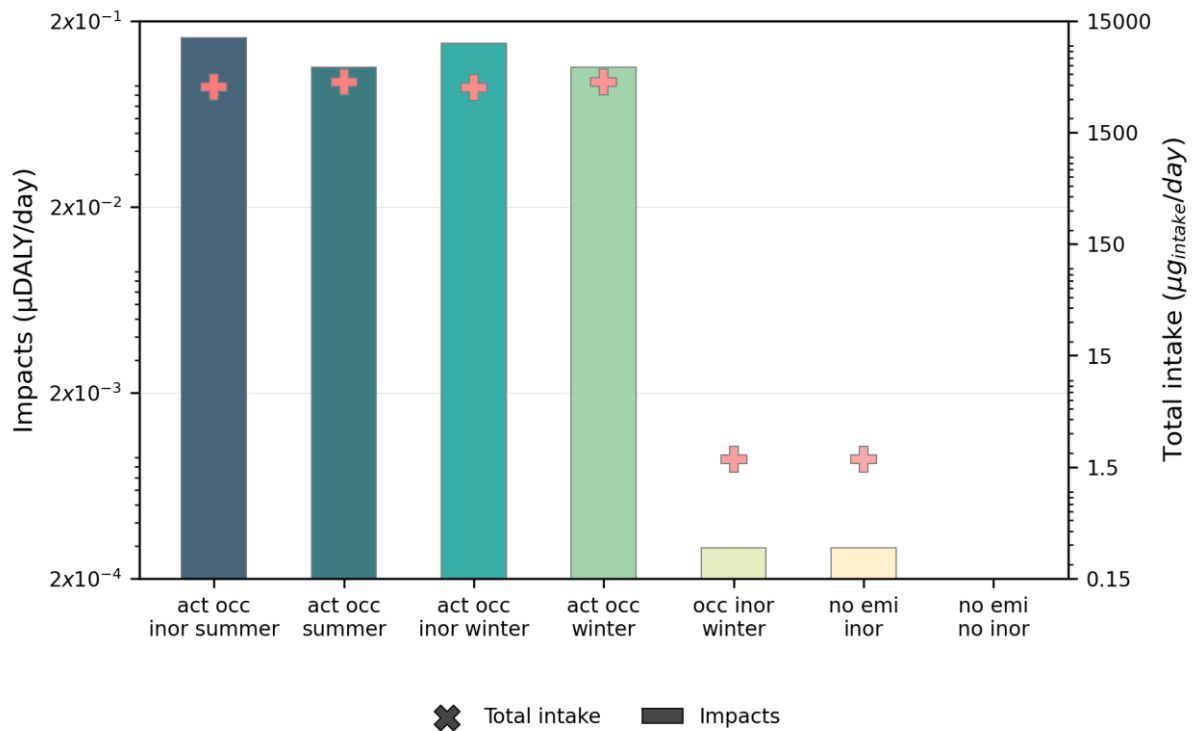


Figure 3-6: Intake quantities ( $\mu\text{g}/\text{day}$ ) and impacts ( $\mu\text{DALY}/\text{day}$ ) for seven scenarios: activities and occupant sources in summer, considering indoor chemical reactions (act occ inor summer) and without chemical reactions (act occ summer), activities and occupant sources in winter, with reactions (act occ inor winter) and without reactions (act occ winter), occupants in winter with reactions (occ inor winter), no indoor emissions in winter with reactions (no emission inor) and without reactions (no emission no inor)

From Figure 3-6 we note that occupants' contribution to indoor air concentrations from skin and breathing emissions are negligible as compared to activities, since their VOC emission rates are lower than those of activities by a ratio  $10^4$  to  $10^{10}$ . We also observe that, in the absence of chemical reactions, impacts in summer and in winter are the same. However, these reactions lead to higher impacts in summer:  $0.16 \mu\text{DALY}/\text{day}$  in summer (+31%) and  $0.15 \mu\text{DALY}/\text{day}$  in winter (+25%). Though there is a more important decrease in terpinolene and d-limonene exposure (decrease of  $2.49 \times 10^5 \mu\text{g}\cdot\text{s}/\text{m}^3$  in summer and  $1.23 \times 10^5 \mu\text{g}\cdot\text{s}/\text{m}^3$  in winter), the increase in formaldehyde exposure is more important (increase of  $8.90 \times 10^4 \mu\text{g}\cdot\text{s}/\text{m}^3$  in summer v/s  $8.38 \times 10^3 \mu\text{g}\cdot\text{s}/\text{m}^3$  in winter).

Intake quantities are slightly different with or without chemical reactions for scenarios with activity and occupant emissions, despite constant ACH and equal emission rates. This is due to variations in indoor concentrations with indoor chemical reactions. The intake is lower in summer in the presence of chemical reactions, since some substances such as d-limonene and terpinolene are consumed, leading to a decrease in air concentration by  $40 \mu\text{g}/\text{m}^3$  to over  $80 \mu\text{g}/\text{m}^3$  respectively (see Figure 3-4), but the impacts are higher due to formaldehyde formation. These reflect the difference in

toxicity of different substances, especially for formaldehyde which is highly toxic and one of the main contributors to health impacts. Figure 3-7 illustrates the intake quantities and health impacts of the substances having the highest impacts.

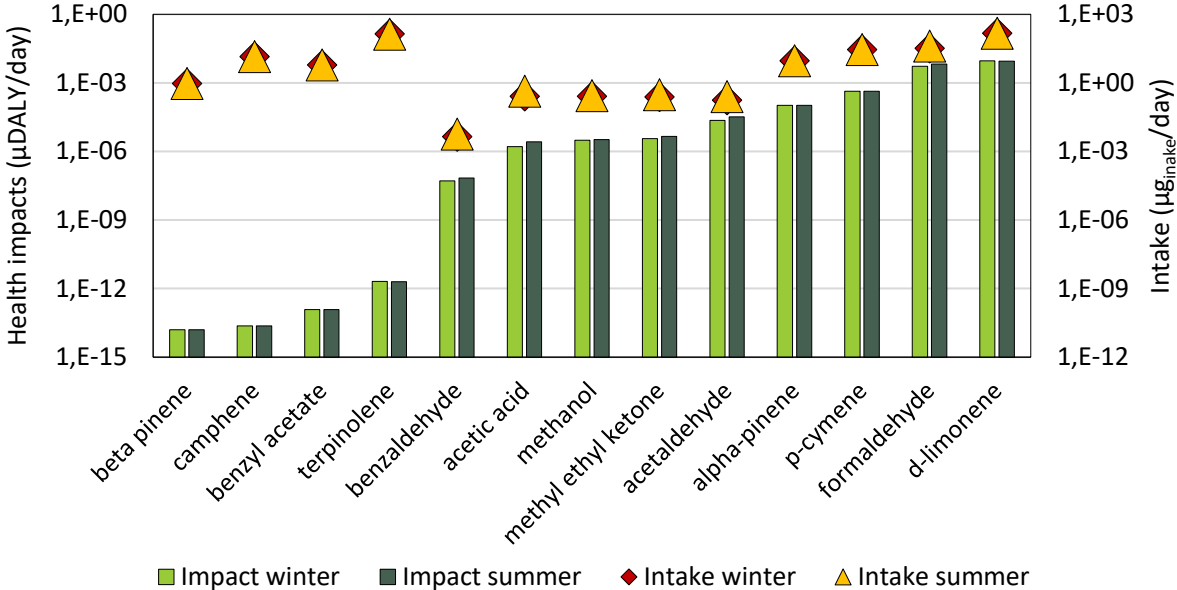


Figure 3-7: Health impacts (left axis) and intake quantities (right axis) due to indoor activity and occupant emissions for VOCs treated by INCA-Indoor and with known toxicity data in winter and in summer, with chemical reactions

It is also interesting to note that impacts occur even in the absence of indoor emissions, mainly due to the formation of formaldehyde: to  $3 \times 10^{-4}$  μDALY/day with the occurrence of these reactions. This indicates that, though efforts can be made to reduce indoor VOC emissions, VOCs formed by chemical reactions should not be neglected.

We note that the increment in PM concentration due to SOA formation leads to an increment in inhalation of 9 μg<sub>PM2.5</sub> (5%). Resulting increase in health impacts are expected to be of the same order, but are outside of the scope of this chapter which focuses on VOC impacts of indoor activities.

#### 4. Conclusions and perspectives

In this chapter, a framework linking VOC emission data to health impacts was presented. Applied to a case study, the health damage due to VOCs emitted by occupants (skin, breath and shoes) and office activities (floor and counter cleaning, use of air freshener) has been calculated. Damages range from 0.16 μDALY/day in summer and 0.15 μDALY/day in winter, leading to about 17 μDALY/year for all occupants (about 9 minutes<sub>lost</sub>/year). Impacts are most certainly underestimated due to lack of toxicity data, but also because INCA-Indoor does not treat all substances: 32 emission rates out of 92 (35%) were not considered. We also note that variations in seasons can affect indoor VOC concentrations, since chemical reactions are more important at higher temperatures. Indoor

chemistry is thus important to consider as it can affect the health of occupants: health damages in summer with reactions were of 0.16  $\mu$ DALY/day v/s 0.11  $\mu$ DALY/day without (+45%). The formation of secondary organic aerosols affects PM concentrations in this case study by up to 2.9  $\mu$ g/m<sup>3</sup> (up to 21%) due to coagulation and nucleation.

A perspective of this study is to include more toxicity data when they become available. Though it is difficult to assess impacts of all known toxic substances since they rely on laboratory and epidemiology studies, machine learning models are being developed and their results can contribute to the robustness of this study (Hou et al. 2020). The health effects of PM, O<sub>3</sub> and CO<sub>2</sub> should also be considered due to their strong indoor presence and the difference in PM concentrations due to SOA (especially for small particles of diameter <1  $\mu$ m which penetrate more easily into the respiratory tract). We saw, in the case study, that CO<sub>2</sub> concentrations were higher than recommended values throughout the day. Besides climatic effects, CO<sub>2</sub> has up to now been mainly linked to short-term effects such as drowsiness or headaches, but recent studies indicate that it could have long-term health effects such as bone demineralisation, kidney calcification or induced high-level cognitive disabilities (Jacobson et al. 2019). Effort should be made to further characterise its toxicity and derive effect factors for LCA. Special attention should be paid to low-ventilated rooms which can favour the production of toxic substances such as formaldehyde (Salthammer 2015), and SOA formation. A study on secondary substances formation as a function of ventilation rates could help quantify the effect of ventilation rates on secondary species formation.

This framework can be applied to different activities with known emission rates to characterise their effects on health and integrate them into LCA. However, these rates are daily averages, while emissions can occur while the product is drying (e.g. in the case of cleaning). Further study on the duration of evaporation of cleaning products as a function of amount applied, surface area, temperature and ventilation rates is encouraged. The proposed framework can help devise intelligent ventilation systems in order to avoid the introduction of precursors such as O<sub>3</sub> and NO<sub>x</sub> when buildings are not occupied, and ventilate for a given period before occupants enter the building to evacuate pollutants emitted by (and accumulated due to) other sources such as materials.



# 4. Health impacts of fine particulate matter from indoor activities

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Les particules fines (PM<sub>2.5</sub>) ambiantes et intérieures sont un des enjeux sanitaires prioritaires dans le monde, étant responsables de 120 million et 92 million d'années de vie perdues (DALYs) respectivement en 2019. Bien que les méthodes actuelles d'ACV traitent les sources extérieures de PM<sub>2.5</sub>, les impacts des PM<sub>2.5</sub> intérieures (émises à l'intérieur ou pénétrant de l'extérieur) ne sont pas considérés. Pour combler cette lacune, nous développons une méthode liant les émissions de particules fines en intérieur à l'exposition des occupants du bâtiment afin d'évaluer les impacts sur la santé en DALYs.

Le modèle dynamique INCA-Indoor permet de calculer les concentrations de PM<sub>2.5</sub> dans l'air intérieur tenant compte des variations temporelles. Différents scénarios d'ouverture de fenêtres sont considérés (toujours ouvertes/toujours fermées/ouvertes avant/ouvertes pendant/ouvertes après l'activité) et, par conséquent, différents taux de renouvellement d'air (TRA). Les résultats du modèle dynamique sont utilisés comme référence afin d'évaluer les biais d'un modèle paramétrique. Pour tous les scénarios, des facteurs d'effet, facteurs de caractérisation et impacts sur la santé sont calculés pour 19 activités intérieures sur une durée de référence d'une heure, avec le modèle non-linéaire intégré d'exposition-réponse du *Global Burden of Disease* (charge mondiale de la morbidité).

Les concentrations intérieures dépassent la limite recommandée par l'OMS pour toutes les activités avec des TRA très faibles. L'ouverture des fenêtres durant l'activité permet de réduire considérablement les impacts. Par exemple, cuisiner pendant une heure sur une plaque électrique représente 17  $\mu$ DALY/personne (9 minutes perdues) dans une cuisine fermée avec un TRA de 0.2 ACH (taux de renouvellement d'air en h<sup>-1</sup>), alors qu'en ouvrant des fenêtres (considérant une pièce traversante) pendant l'activité, les impacts diminuent à  $4 \times 10^{-3}$   $\mu$ DALY/personne. On associe des impacts très élevés (0.2 à 480  $\mu$ DALY/personne, soit 240 minutes de vie perdues pour des taux de renouvellement d'air très faibles) à la cuisson au feu de bois en intérieur, pratique encore courante en Inde et en Chine rurale. Avec des taux plus réalisés de 3 à 14 ACH, les impacts étaient de 98 à 119  $\mu$ DALY/h<sub>activité</sub> (52 à 63 minutes<sub>perdues</sub>/h<sub>activité</sub>). Pour l'utilisation de bougies, à des taux de renouvellement d'air faibles (0.2 à 0.6 ACH, scénario réaliste), les impacts sont de 7 à 11  $\mu$ DALY/h<sub>activité</sub> (4 to 11 minutes<sub>perdues</sub>/h<sub>activité</sub>).

Les facteurs de caractérisation et impacts calculés constituent un point de départ pour l'introduction des PM<sub>2.5</sub> intérieures en ACV : ils peuvent être adaptés à des durées réelles d'activité pour un scénario donné. Pour des cas d'étude plus spécifiques, la méthode développée peut être utilisée afin de calculer les impacts des PM<sub>2.5</sub> intérieures sur la santé des occupants du bâtiment et, couplé avec des méthodes d'ACV actuelles, permettre d'identifier un taux de ventilation optimal permettant de réduire au mieux le total des impacts sur la santé.

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## Abstract

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Exposure to fine particulate matter (PM<sub>2.5</sub>) is an important contributor to global human disease burden, particularly indoors where people spend the majority of their time and exposure is highest. We propose a framework linking indoor PM<sub>2.5</sub> emissions from human activities to exposure and health impacts. Derived dynamic indoor PM<sub>2.5</sub> concentrations—capturing temporal variations through different window opening scenarios and air renewal rates—are used to estimate uncertainty for a parametric model. Intake fractions ( $\mu\text{g}_{\text{intake}}/\mu\text{g}_{\text{emitted}}$ ), effect factors ( $\mu\text{DALY}/\mu\text{g}_{\text{intake}}$ ), and related impact characterisation factors ( $\mu\text{DALY}/\mu\text{g}_{\text{emitted}}$ ) and impact scores ( $\mu\text{DALY}/\text{h}_{\text{activity}}$ ) are provided for 19 one-hour indoor activities, and can be flexibly scaled to real activity durations. Indoor concentrations exceeded recommended World Health Organization (WHO) limits for all activities at low ventilation rates. Per person, 98 to 119  $\mu\text{DALY}/\text{h}_{\text{activity}}$  (52 to 63  $\text{minutes}_{\text{lost}}/\text{h}_{\text{activity}}$ ) were associated with traditional fuel cook stoves, with high air renewal rates (3 and 14 ACH: air changes per hour in  $\text{h}^{-1}$ ). The burning of candles, at low air renewal rates of 0.2 to 0.6 ACH, results in 7 to 11  $\mu\text{DALY}/\text{h}_{\text{activity}}$  (4 to 11  $\text{minutes}_{\text{lost}}/\text{h}_{\text{activity}}$ ). Derived impact scores and characterisation factors serve as a starting point for integrating indoor PM<sub>2.5</sub> emissions and exposure into life cycle impact and public health assessments.

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## 1. Introduction

Ambient and household fine particulate matter (PM<sub>2.5</sub>) pollution is one of the major global health risk factors, representing 120 million and 92 million Disability-Adjusted Life Years (DALYs) each in 2019 (4.7% and 3.6% of total DALYs) (Murray et al. 2020, 87). Serious health outcomes are associated with PM<sub>2.5</sub> exposure, including chronic obstructive diseases (COPD), ischemic heart disease (IHD), stroke and lung cancer (LC) for adults, and acute lower respiratory diseases (ALRI) for children who are still in the developmental stage (< 5 years) (Burnett et al. 2014; Fantke et al. 2015). Indoor environments, where people spend a high fraction of their time, are particularly important to study: 83% to 90% of exposure occurs indoors (Fantke, Jolliet, et al. 2017). Since buildings have relatively small, enclosed volumes with small air change rates, indoor air concentrations can be particularly high as compared to ambient levels. For instance, PM<sub>2.5</sub> air concentrations were higher than the World Health Organization's (WHO) annual guideline values in schools and kindergartens by a factor 4 to 15 (OQAI 2018; Mandin 2020; Mainka and Fantke 2022). According to these guidelines, annual average concentrations should remain below 5 µg/m<sup>3</sup>, and 24-hour exposures should not exceed 15 µg/m<sup>3</sup> for more than 3 to 4 days per year (WHO 2021). Inhabitants of rural areas, especially in developing countries, are particularly at risk due to wide usage of solid fuel combustion indoors for cooking or heating (Smith 2000; Smith et al. 2011; Rohra et al. 2018).

Indoor PM<sub>2.5</sub> concentrations depend on outdoor pollution levels, penetrating through unfiltered ventilation, indoor primary emissions from activities, and chemical reactions between substances such as the oxidation of volatile organic compounds (VOCs) with ozone, nitrate and hydroxyl radicals which can form secondary PM (Srivastava et al. 2022). Various studies have measured primary PM<sub>2.5</sub> emission rates, indoor concentrations and particle size distributions for different activities (Pagels et al. 2009; Aquilina and Camilleri 2022; Long, Suh, and Koutrakis 2000; He et al. 2004b; Tissari et al. 2008; Shen et al. 2020; Demanega et al. 2021; W. Du et al. 2021; C. Li et al. 2022). While fuel stoves are recognised as strong indoor PM<sub>2.5</sub> sources and linked to premature mortality in developing countries, cooking (especially frying and grilling) and candle burning were also identified as important sources. Occupant contributions to indoor PM<sub>2.5</sub> from the shedding of skin and cloth fibres or the resuspension of particles during activities have also been measured, and are highly dependent on dust coverage and, ultimately, occupant behaviour (e.g. frequency of cleaning or presence in dusty environments) (Ferro, Kopperud, and Hildemann 2004; Corsi, Siegel, and Chiang 2008; Bhangar et al. 2016; Licina, Tian, and Nazaroff 2017; Al Assaad et al. 2020). Indoor concentrations are affected by air change rates (ACH), the presence of filters for mechanical ventilation, and the size and orientation of windows for natural ventilation.

The purpose of this chapter is to propose a framework for integrating indoor human health impacts from PM<sub>2.5</sub> intake into building life cycle assessment. Current LCA methods only treat outdoor PM<sub>2.5</sub> emissions. The UNEP/SETAC Life Cycle Initiative identified the PM<sub>2.5</sub> impact category as one

of the categories requiring refinements, stressing on the need for representative indoor archetypes treating indoor sources of PM<sub>2.5</sub> and consequent occupant exposures (Fantke et al. 2015). Archetypes have been defined by Fantke et al. (2017) according to parameters identified by Hodas et al. (2016), including air renewal rates and occupancy. Parametric models coupled with these indoor archetypes can provide average PM<sub>2.5</sub> concentrations and intake fractions (mass taken in per unit mass emitted) (Fantke, Jolliet, et al. 2017), but usually do not capture temporal variations in emission or ventilation rates. Dynamic simulations, using airflow simulation tools such as CONTAM (used by INCA-Indoor (Mendez et al. 2015)) or COMIS, can provide full concentration/exposure profiles that can be coupled with effect factors to calculate health impacts.

There is still, however, a need for the definition of emission scenarios linked to indoor sources and their emission rates. No factors linking emissions, exposure and health effects to different indoor activities in LCA were calculated in literature. To address this gap, the goal of this chapter is to propose a framework for characterising human exposure to indoor PM<sub>2.5</sub> and related health impacts associated with common indoor activities for their integration to building LCA. To achieve this goal, we defined three specific objectives:

1. To propose a framework for linking human indoor activities to primary PM<sub>2.5</sub> emissions, exposure, effects, and health damage based on a dynamic modelling approach,
2. To calculate dynamic indoor PM<sub>2.5</sub> concentrations and derive a parameterised exposure and effect model for integration in life cycle impact assessment, and
3. To provide a set of impact characterisation factors for different reference indoor activities under different natural ventilation scenarios.

## 2. Materials and methods

### 2.1. Overall followed source-to-damage approach

The pollutant pathway from emission to impact is modelled using the framework recommended in global consensus-building efforts for PM<sub>2.5</sub> exposure and effects (Fantke et al. 2015; Humbert et al. 2011), adapted to indoor contexts, and is shown in [Figure 4-1](#). The functional unit is defined as one hour of activity.

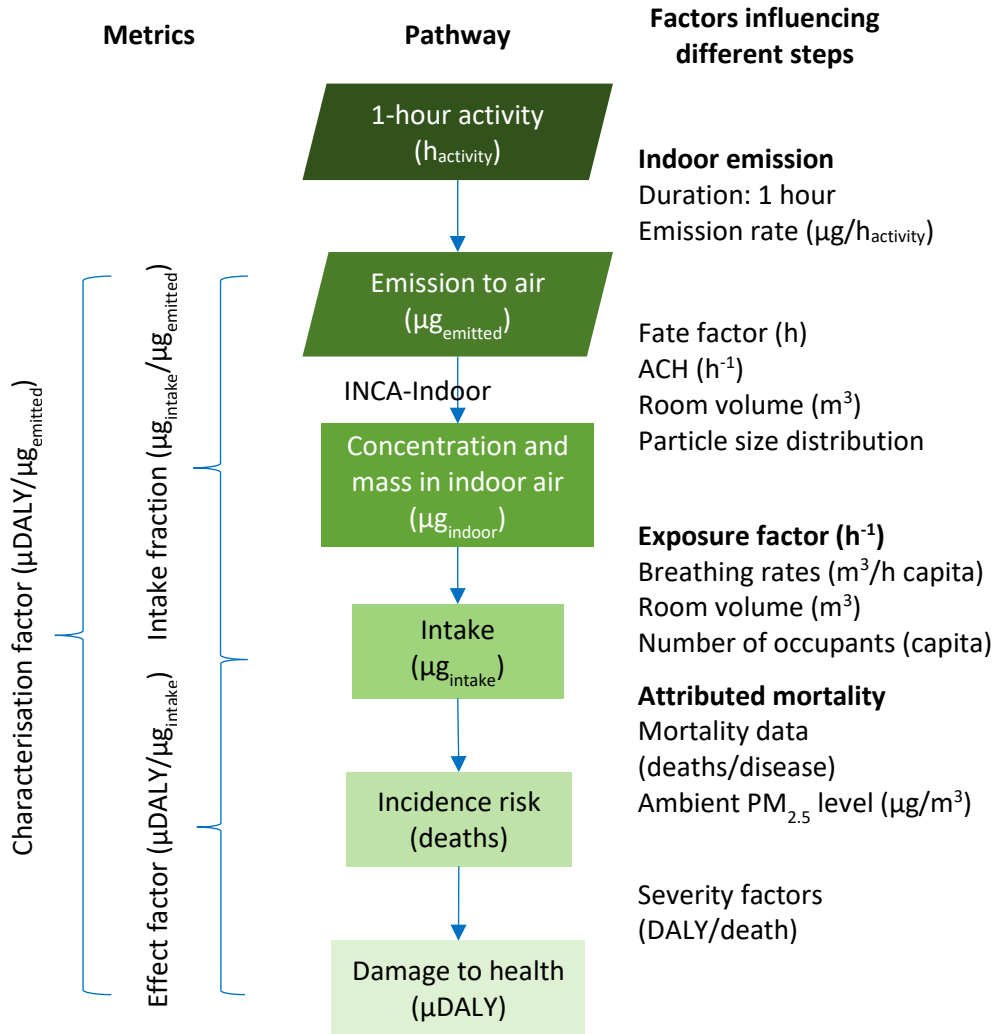


Figure 4-1: Framework for the calculation of activity impact scores from  $\text{PM}_{2.5}$  emission rates

Indoor  $\text{PM}_{2.5}$  concentrations  $C_{\text{in}}$  ( $\mu\text{g}/\text{m}^3$ ) are simulated with INCA-Indoor, considering 1) indoor emission rates of different activities  $\dot{m}_{\text{emitted in,avg}}$  ( $\mu\text{g}_{\text{emitted}}/h_{\text{activity}}$ ) obtained from studies, 2) the penetration from outdoors with natural ventilation  $\dot{m}_{\text{penetration,avg}}$  ( $\mu\text{g}/h$ ) and 3) air renewal rates ( $h^{-1}$ ). These are compared to a parametric model's results, using one hour as reference activity duration under multiple given air change rates  $ACH$  ( $h^{-1}$ ). Using a fate factor  $FF$  (h) or the dynamic fate model INCA-indoor, we determine the evolution of the indoor  $\text{PM}_{2.5}$  concentrations and the resulting time integrated mass inside the room air. These masses are then multiplied by the exposure rates ( $h^{-1}$ ) to yield indoor  $\text{PM}_{2.5}$  intake fractions  $iF$  ( $\mu\text{g}_{\text{intake}}/\mu\text{g}_{\text{emitted}}$ ) (see equation (4-2)). These  $iF$ s are multiplied by effect factors  $EF$  ( $\mu\text{DALY}/\mu\text{g}_{\text{intake}}$ ) to obtain the characterisation factors  $CF$  ( $\mu\text{DALY}/\mu\text{g}_{\text{emitted}}$ ), i.e. the impact per unit of  $\text{PM}_{2.5}$  emitted. Impact scores  $IS$  ( $\mu\text{DALY}/h_{\text{activity}}$ ) for one person are then calculated as the product of the cumulative indoor emission  $m_{\text{emitted in}}$  ( $\mu\text{g}_{\text{emitted}}$ ) and  $CF$ s. This can therefore be expressed as:

$$IS_{\text{activity}} = EF \times iF \times m_{\text{emitted in}} = CF \times m_{\text{emitted in}} \quad (4-1)$$

With

$$iF = \frac{\int_{t=0}^{\infty} BR \times POP \times C_{in,inc} dt}{m_{emitted\ in}} \quad (4-2)$$

Where  $POP$  (cap) is the number of occupants,  $BR$  is the breathing rate of an occupant ( $m^3/cap/h$ ).  $C_{in,inc}(t)$  is the increase in indoor  $PM_{2.5}$  concentration due to the activity related emissions ( $\mu g/m^3$ ) integrated up to infinity (in practice up to the time required to entirely evacuate the particles emitted by the activity). It is given by the difference between indoor concentration with activity ( $C_{in}$ , in  $\mu g/m^3$ ) and without ( $C_{base}$ , in  $\mu g/m^3$ ).

The effect factor depends on the average effective indoor concentration  $\bar{C}_{in}$  and the annual average ambient concentration of the region  $\bar{C}_{out,r}$  (Cohen et al. 2017, Fantke et al. 2019).

$$\begin{aligned} EF(\bar{C}_{in}) &= \frac{dM_{PM_{2.5}}}{dI_{in}} \times SF_{i,r} \quad (4-3) \\ &= \frac{(RR_i(\bar{C}_{in} + \Delta C_{in}) - RR_i(\bar{C}_{in})) \times \frac{M_{i,r}}{RR_i(\bar{C}_{out,r}) \times POP_r} \times SF_{i,r}}{\Delta \bar{C}_{in} \times BR_{yr}} iF \\ &= \int_{t=0}^{t_{day}} \frac{BR \times POP(t)}{V_{room} \times (ACH(t) + DR(t)) + BR \times POP(t)} \end{aligned}$$

Where  $RR_i(-)$  is the relative risk of developing disease  $i$  from exposure to  $\bar{C}_{in}$ ,  $\Delta \bar{C}_{in}$  ( $\mu g/m^3$ ) is the increment on the exposure-response curve,  $M_{i,r}$  (deaths/year) the annual mortality in region  $r$  due to disease  $i$ ,  $I_{in}$  (kg/year) is the intake,  $POP_r$  (persons) is the population of the region,  $SF_{i,r}$  (DALY/death) is the severity factor specific to the region and disease and  $BR_{yr}$  ( $m^3/year$ ) is the breathing rate. The effect factor depends on exposure concentrations and can hence be different for activities with different emission rates. Underlying equations for the relative risk obtained from the integrated exposure-response model (Burnett et al. 2014) are provided in annex A3.1.

The  $EF$  depends on the average effective indoor concentration  $\bar{C}_{in}$ . This can be either calculated by INCA-Indoor, or as a comparison calculated using Fantke et al. (Fantke, Jolliet, et al. 2017), adapted to consider intake and deposition for outdoor  $PM_{2.5}$ :

$$\bar{C}_{in} = \frac{(\dot{m}_{emitted\ in,avg} + \dot{m}_{penetration,avg})}{V_{room} \times \left( ACH_{avg} + DR_{avg} + \frac{BR_{avg} \times POP_{avg}}{V_{room}} \right)} \quad (4-4)$$

$\dot{m}_{penetration,avg}$  ( $\mu g/h$ ) is the average penetration rate of  $PM_{2.5}$  from outdoors, defined by:

$$\dot{m}_{\text{penetration,avg}} = \bar{C}_{\text{out}} \times V_{\text{room}} \times ACH_{\text{avg}} \quad (4-5)$$

## 2.2. Individual Lifetime Risk

The total individual lifetime risk  $\overline{ILR}$  (DALY/person/lifetime) represents the number of life years lost from exposure to PM<sub>2.5</sub> over a lifetime for each one-hour activity. It is calculated using equation (4-6).

$$ER_{\text{out}} = \bar{C}_{\text{out}} \times V_{\text{room}} \times ACH_{\text{d}} \quad (4-6)$$

Where  $N_{\text{activity}}$  is the number of one-hour activities occurring during a lifetime.

## 2.3. Input data: concentrations

To calculate concentrations, room parameters, including ventilation rates, volume, and deposition rates, as well as activity emission rates and durations are needed. Two rooms (small and large) are treated in this chapter, different from previous case studies in Chapter 2 and 3. Resuspension rates are presented in annex A3.2, but are not treated in this chapter since values presented in literature are different for the same activities (Ferro, Kopperud, and Hildemann 2004; Corsi, Siegel, and Chiang 2008; Lewis et al. 2018; Rohadi et al. 2020; B. Wang et al. 2021; Aquilina and Camilleri 2022), and further study on the topic is required to derive impact scores for different scenarios. This indicates that they depend on surface dust loading, which is itself highly dependent on occupant habits such as cleaning frequency or wearing shoes indoors, and the proximity of the building to sources of dust (e.g. construction sites, dry earth or sand). However, they are important sources (resuspension rates of the same order as emission rates) and will be considered in the final case study in Chapter 5 to demonstrate the applicability of our framework.

### 2.3.1. Model parameters

Table 4-1 summarises the main parameters used for the calculation of indoor PM<sub>2.5</sub> concentrations.

*Table 4-1: Concentration model input parameters*

<b>Parameter</b>	<b>Value</b>
Deposition rate $DR$	5 1/d <sup>(a)</sup>
Average air change rate $ACH$	variable <sup>(b)</sup>
Room volume $V_{\text{room}}$	30 m <sup>3</sup> / 67 m <sup>3</sup>
Number of occupants $POP$	1
Reference average outdoor PM <sub>2.5</sub> concentration $C_{\text{out}}$	16 µg/m <sup>3</sup> <sup>(c)</sup>

<sup>(a)</sup> Deposition rate applied to the parametric model, matching the simulation DR which depends on indoor air velocity and surface area.

<sup>(b)</sup> ACH varies according to scenarios presented in [Table 4-4](#).

<sup>(c)</sup> Dynamic outdoor concentrations (with one-hour time step) are used for the simulation and a reference 24-h average is used in the parametric model, corresponding to an average European city (WHO 2016).

### 2.3.2. Activity emission rates

Primary PM<sub>2.5</sub> emission rates are collected from various studies for 19 activities and are presented in [Table 4-2](#).

Table 4-2: Primary PM<sub>2.5</sub> emission rates for 19 activities

Activity	PM <sub>2.5</sub> emission rate (mg/min)	Reference
Candle burning (low)	0.04	Pagels et al. (2009)
Toasting	0.11	He et al. (2004)
Cooking with electric stove (low)	0.11	He et al. (2004)
Candle burning (medium)	0.15	Pagels et al. (2009)
Incense - aromatic (low)	0.16	Lee and Wang (2004)
Gas stove	0.24	He et al. (2004)
Printer	0.28	He et al. (2010)
Frying (low)	0.43	Aquilina and Camilleri (2022)
Grilling (low)	0.62	Aquilina and Camilleri (2022)
Candle with eucalyptus oil diffusion (high)	0.91	He et al. (2004)
Smoking	0.99	He et al. (2004)
Cook stove (low) <sup>a</sup>	1.2	Du et al. (2021)
Cooking (high, with burning)	1.33	Aquilina and Camilleri (2022)
Frying (high)	2.68	He et al. (2004)
Grilling (high)	2.78	He et al. (2004)
Heating stove <sup>d</sup>	3.56	Li et al. (2022)
Incense - traditional (high)	6.21	Lee and Wang (2004)
Cook stove (medium) <sup>b</sup>	7.9	Shen et al. (2020)
Cook stove (high) <sup>c</sup>	120	Du et al. (2021)

a. Fugitive emissions (leakage) from cooking with the burning of coal in an iron stove.

b. Fugitive emissions from cooking with the burning of wood in a brick stove.

c. Fugitive emissions from cooking with the burning of maize straw in a brick stove.

d. Emission rate (mg/min) calculated from the emission factor (g/kg).

For some activities, different values were obtained for different variations of the same activity. For instance, cooking can occur with a low value (on an electric stove) and higher value if burning

occurs. Values for frying and grilling were obtained from two different sources, indicating a possible range of emission rates. For candles and incense sticks, different emission rates are linked to different product references (see Pagels et al. (2009) and Lee and Wang (2004)). Cook stove emission rates are highest (up to 120 mg/min) and correspond to common practices in certain rural homes, e.g. in India, Guatemala or China: (Smith 2000; Smith et al. 2010; W. Du et al. 2021).

### Heating stove emission rates

The mass of wood  $\dot{M}_{\text{coal}}$  (kg/s) required per second is calculated by the following expression:

$$\dot{M}_{\text{coal}} = \frac{P_{\text{heat}}}{HV_{\text{wood}}} \quad (4-7)$$

$P_{\text{heat}}$  (J/s or W) is the average heating power required for a 30 m<sup>3</sup> room ventilated at 0.6 ACH over heating periods. It is calculated from the heating needs which are simulated using Pleiades (IZUBA ÉNERGIES).  $HV_{\text{coal}} = 30$  MJ/kg is the low heating value (LHV) of coal. We obtain the emission rate  $ER$  (mg/min) from equation (4-8):

$$ER = F_{\text{emission}} \times \dot{M}_{\text{coal}} \times 1000 \times 60 \quad (4-8)$$

$F_{\text{emission}} = 4$  gPM<sub>2.5</sub> emitted/kg<sub>coal</sub> burnt is the emission factor (Li et al. 2022).

### 2.3.3. Occupant emission rates

The emission rates of particulate matter by occupants according to different levels of activity (Licina et al. 2017) were used to derive emission rates for PM<sub>2.5</sub> for an average office working day, summarised in Table 4-3.



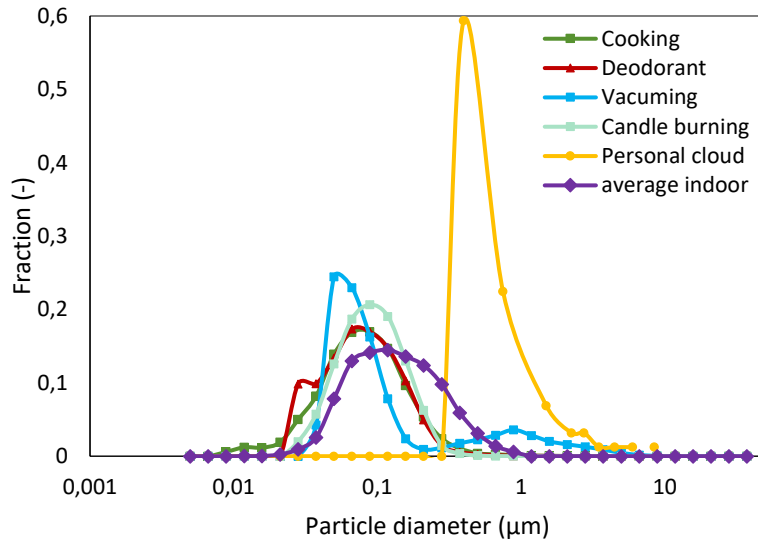
*Table 4-3: Direct occupant primary PM<sub>2.5</sub> emission rates for moderate and intensive activities while seated and for walking at 80 steps/min*

<b>Activity</b>	<b>PM<sub>2.5</sub> emission rate (mg/min)</b>	<b>Reference</b>
Seated, moderate activity	3.61 x 10 <sup>-4</sup>	
Seated, intensive activity	5.46 x 10 <sup>-4</sup>	Derived from Licina et al. (2017)
Walking 80 steps/min	8.11 x 10 <sup>-4</sup>	
Average*	4.27 x 10 <sup>-4</sup>	

\* The average is calculated considering 75% seated moderate activity, 20% seated intensive activity and 5% walking during occupancy, considering an 8-hour working day with occupants sitting for 7.6 hours sitting, and the rest of the time walking.

#### 2.3.4. Particle number-size distribution

In order to model the transport of particles, the size distribution has to be indicated. 27 particle diameter ranges from 0.004 µm to 10 µm are defined in INCA-Indoor. Since particle size distributions are only available for specific activities, we select a more general indoor distribution, irrespective of the emission source (Abt et al. 2000). In order to reduce uncertainty on the concentrations linked to uncertainty on distributions, only PM<sub>2.5</sub> emission data are used. Figure 4-2 shows particle-size distributions for emissions from different activities: cooking (Wallace, Emmerich, and Howard-Reed 2004), using deodorants, vacuum cleaning and candle burning (Demanega et al. 2021), from the personal cloud (Licina, Tian, and Nazaroff 2017) and from average indoor sources (Abt et al. 2000).



*Figure 4-2: Particle size distribution for different activities (cooking, deodorant spray, vacuuming, candle burning), for occupant direct emissions (personal cloud) and for indoors on average*

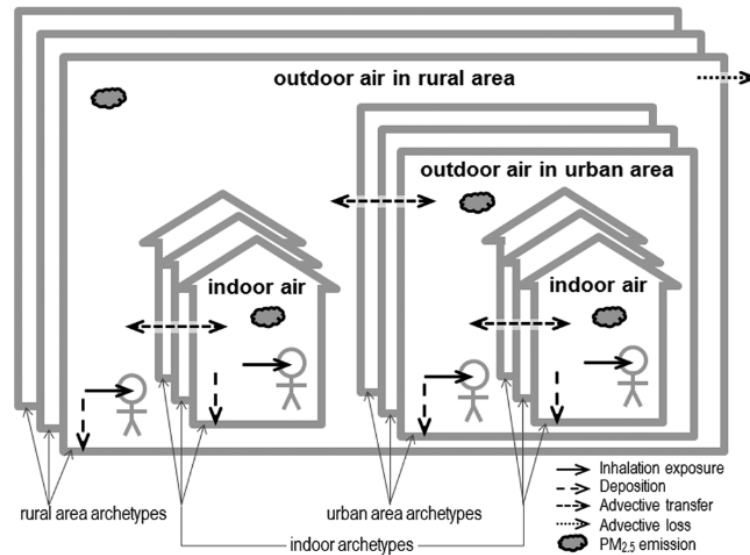
The distribution of particles from the personal cloud is skewed compared to the others which remain within a closer range to the average distribution.

## 2.4. Concentrations

Concentrations are calculated for each activity using both the parametric model and the dynamic model. In both cases, the room under study has a volume of 30 m<sup>3</sup>.

### 2.4.1. Four-compartment model description

The model developed by Fantke et al. (2017), modified to integrate deposition of particles penetrating from outdoors (see equation (4-4)), allows to calculate PM<sub>2,5</sub> concentrations and intake fractions in different compartments presented in Figure 4-3.



*Figure 4-3: Coupled indoor and outdoor air compartments used to assess the exposure to PM<sub>2.5</sub> in urban and rural environments (Fantke, Jolliet, et al. 2017)*

Air penetrates indoors from outdoors and is evacuated from indoors to outdoors. The compartments of interest in this study are indoor air and outdoor air, without differentiating rural and urban areas.

#### 2.4.2. Dynamic model description

Concentrations are calculated using the INCA-Indoor dynamic multizone simulation model (Mendez et al. 2015). The following inputs are necessary for the simulation: 1) building characteristics, including room dimensions, mechanical ventilation rates if any, window sizes and layout (modelled with the Pleiades software (IZUBA ÉNERGIES 2001a), 2) dynamic outdoor PM<sub>2.5</sub> concentrations, 3) meteorological data (temperature, wind speed and direction), 4) indoor PM<sub>2.5</sub> emission rates, 5) time and duration of emissions and 6) particle size distribution. Air flows are simulated with CONTAM based on the opening of windows, infiltration rates under 4 Pa and meteorological data, considering a constant indoor temperature of 20 °C. Concentrations are calculated with a time step of 10 minutes as a function of air flow rates, emission rates, outdoor PM<sub>2.5</sub> concentrations and deposition rates.

Because the dynamic model allows to capture variations in air change rates (ACH), scenarios have been defined to evaluate the effect of window opening on concentrations and health impacts for each activity, hence estimating the uncertainties of the parametric model. Air change exchange rates (ACH) between indoor and outdoor air are defined according to Fantke et al. (Fantke, Jolliet, et al. 2017). The average ACH for OECD countries is 0.64 ACH (Rosenbaum et al. 2015), while low-end values are around 0.2 ACH for airtight buildings (Persily, Musser, and Emmerich 2010). High air change rates are around 3 ACH, and in non-OECD countries, they can reach 14 ACH.

Four standard scenarios are defined: windows always **closed** with infiltration rates of 0.2 ACH and 0.6 ACH, and windows always **open** with high and very high ventilation rates of 3 ACH and

14 ACH respectively. The air change rates indicated are 24-hour averages, but air flows vary during the day for natural ventilation due to changes in wind speed and direction, temperature, and pressure. In the remaining six scenarios, windows are either open **before**, **during** or **after** activity, with high and very high average air renewal rates of 3 ACH and 14 ACH. Since windows are open for a limited duration in the last six scenarios (a minimum of one hour allowed by the model), the ACH can be much higher when open in order to reach the target 24-hour average, but do not exceed a reference comfort speed of 1 m/s (Aynsley 2006). Dynamic ACH are illustrated in Annex Figure 8. Very high ventilation rates typically correspond to hot or tropical regions, where cross-ventilation is common, or recommended values for specific uses, such as kitchens or factories (The Engineering Toolbox 2005). 10 different ventilation scenarios are selected, summarised in Table 4-4, where air change rates are 24-h averages.

*Table 4-4: Window-opening scenarios and average daily air change rates*

Windows	ACH (1/h)	
Always closed	0.2	0.6
Always open		
Open before		
Open during	3	14
Open after		

## 2.5. Exposure model data

The intake fractions for one occupant are calculated using equation (4-2) and a breathing rate  $BR$  of  $16 \text{ m}^3/\text{d}$  (Hodas et al. 2016) and an exposure duration of 24 hours.

## 2.6. Input data: effect (IER) model

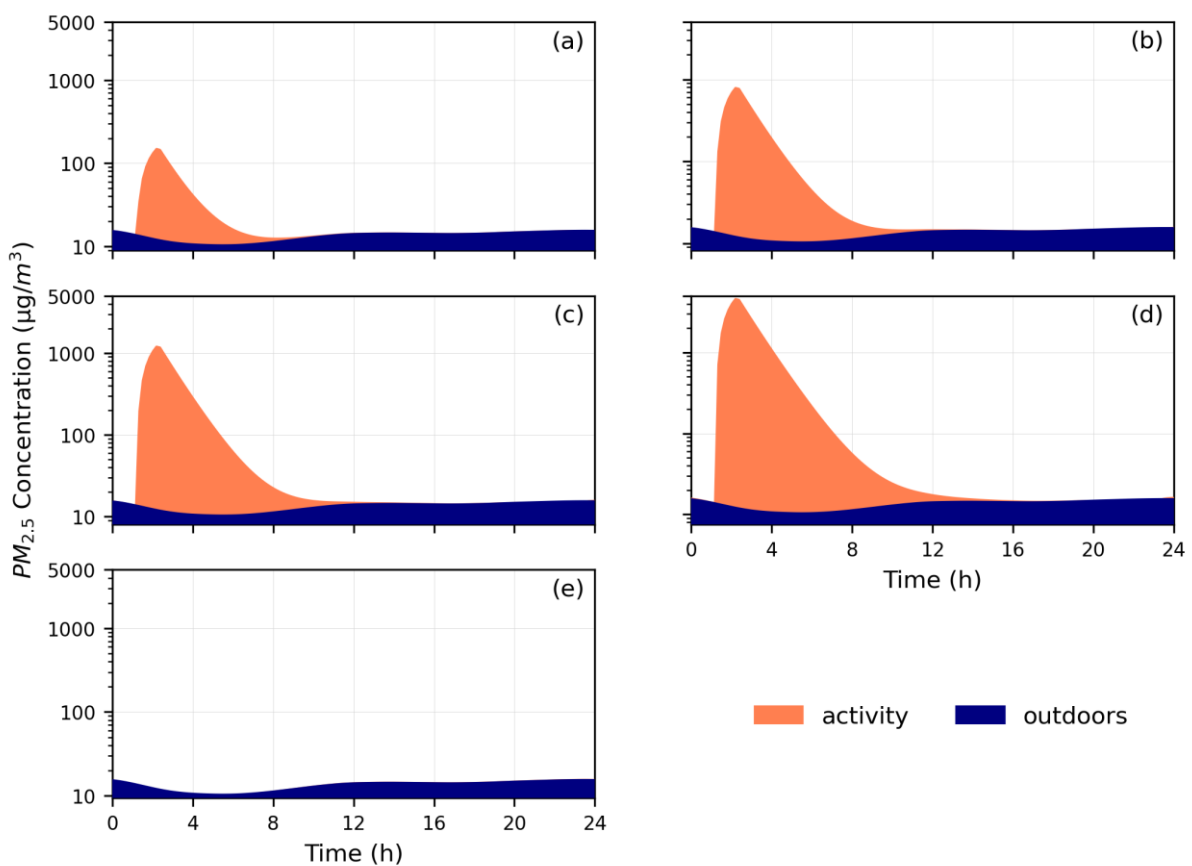
Global population data are obtained from world population prospects (United Nations, Department of Economic and Social Affairs, and Population Division 2019) and age-specific global mortality rates,  $M$  (deaths/year), for the five diseases outcomes are obtained from the GBD Collaborative Network for 2019 (GBD Global Burden of Disease Collaborative Network 2019). An annual average reference ambient  $\text{PM}_{2.5}$  level of  $16 \mu\text{g}/\text{m}^3$ , corresponding to an average European city (WHO 2016), is considered and average exposure concentrations  $\bar{C}_{in}$  ( $\mu\text{g}/\text{m}^3$ ) are calculated over 24 hours. The calculated effect factor  $EF$  only corresponds to exposure to one activity and ambient  $\text{PM}_{2.5}$  concentrations, without considering the occurrence of several activities at the same time. It results in a conservative approach, with effect factors due to potentially lower exposure concentrations.

### 3. Results

#### 3.1. Indoor PM<sub>2.5</sub> concentrations

##### 3.1.1. Dynamic PM<sub>2.5</sub> concentrations

Dynamic PM<sub>2.5</sub> concentrations from four indoor activities in a reference room of 30 m<sup>3</sup> with closed windows at 0.6 ACH are presented in Figure 4-4 over 24 hours, a duration that allows to evacuate most activity-related PM<sub>2.5</sub> from indoor air. Indoor and outdoor concentrations are dynamic. Indoor concentrations increase after emission, and decrease due to evacuation with air renewal. Outdoor concentrations vary independently of the indoor activity, due to change in outdoor emissions (e.g. fuel burning) or wind speeds.



*Figure 4-4: Indoor PM<sub>2.5</sub> concentrations from outdoors (navy) and increment from one hour activities (orange) per day: a) toasting or cooking on an electric stove, b) grilling (low), c) smoking or lighting a candle with essential oil diffusion d) use of a coal heating stove and (e) occupant direct emissions*

PM<sub>2.5</sub> concentration increments are higher for higher emission rates: the use of a coal heating stove can lead to a peak of 4500 µg/m<sup>3</sup>, while toasting or cooking on an electric stove lead to a peak of 150 µg/m<sup>3</sup>. The area under the curve gives the concentration to which occupants are exposed over a given duration (µg.s/m<sup>3</sup>), which is important to consider in health impact assessment. It is ultimately

linked to the decay rate, mainly determined by the air change rate: higher ACH lead to higher decay rates.

Since emission rates of occupants are 2 to 4 orders of magnitude lower than activity emission rates, resulting increase in concentrations are negligible compared to background concentrations. Occupant emissions are thus not considered in this study. Furthermore, the dynamic model allows to calculate concentrations in next-door rooms. The increment in the next-door room represents 1.6% of the increment in the room with activity, and is thus neglected in this study.

### 3.1.2. 24-h average $PM_{2.5}$ concentrations for all activities

24-h average concentrations for the different activities and ventilation scenarios, calculated from results of the dynamic model, are presented in [Figure 4-5](#) for the room with small volume and high occupancy  $HO$  of  $30 \text{ m}^3/\text{occupant}$ . Concentrations in the room with higher volume and low occupancy  $LO$  of  $67 \text{ m}^3/\text{occupant}$  correspond to the average ventilation rate of 0.6 ACH.

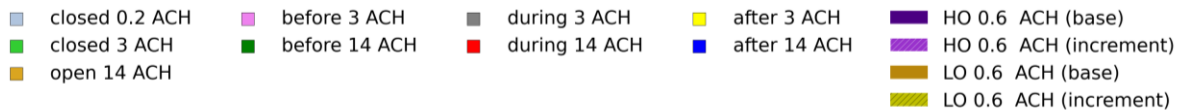
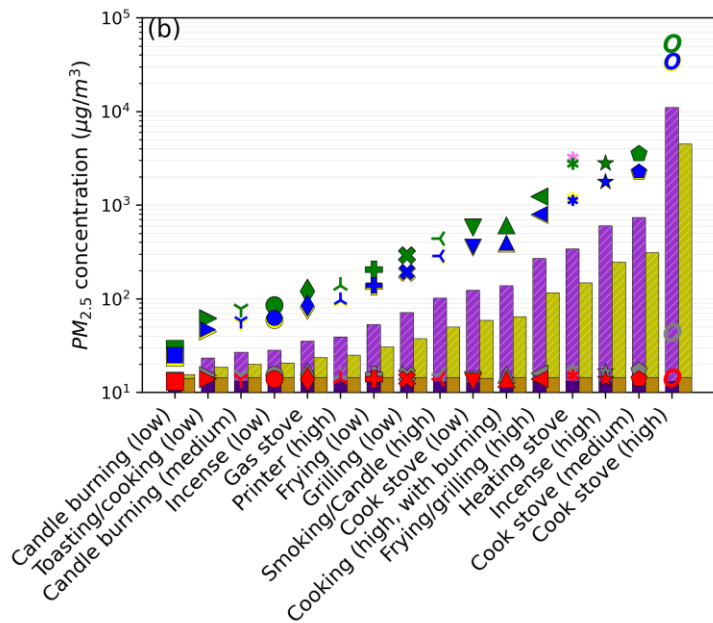
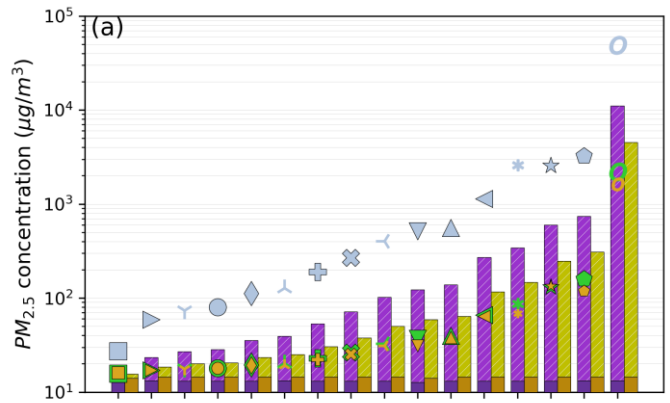


Figure 4-5: Bars represent  $PM_{2.5}$  concentrations for an average scenario with 0.6 ACH without indoor emission (base) and with emissions from activities (increment) for a room with small volume and high occupancy (HO) and a room with high volume and low occupancy (LO). Markers represent concentrations for (a) three other ACH in the small room: 0.2 ACH (always closed), 3 ACH and 14 ACH (always open) and (b) two ACH: 3 ACH and 14 ACH for different opening scenarios (open before, open during and open after)

The increment in the larger room is on average 2.5 times lower than that in the smaller room, and the ratio of their volumes is 2.2. The difference can be explained by a higher deposition rate in the larger room due to larger available surface area. Higher ventilation rates lead to a decrease in concentrations if windows are always open or closed. We note from the other scenarios presented in Figure 4-5 (b), that least concentrations are linked to opening windows during the activity, since

ventilation rates during emission are much higher, while opening before the activity does not affect concentrations: they are equal to the closed window scenario. Opening after the activity allows for partial evacuation of substances and hence a slight decrease in concentrations (see Annex Figure 8).

### 3.1.3. Parametrised model PM<sub>2.5</sub> concentrations

The average PM<sub>2.5</sub> concentration over 24 hours are calculated for each activity with the parametric model described by equation (4-4). Though the parametric model with a mean ACH throughout the day can provide a good estimation of 24-h average indoor air concentrations (e.g. for fixed mechanical ventilation), under certain conditions, the dynamic model is more precise. In most cases, the ACH changes throughout the day according to opening/closing of windows and meteorological conditions. The latter determines natural air flow rates through openings and infiltration. If variations in ACH occur during or right after emissions, indoor PM<sub>2.5</sub> concentrations are affected. The lowest and highest possible concentration for each activity, based on the different scenarios, are illustrated with uncertainty bars in Figure 4-6.

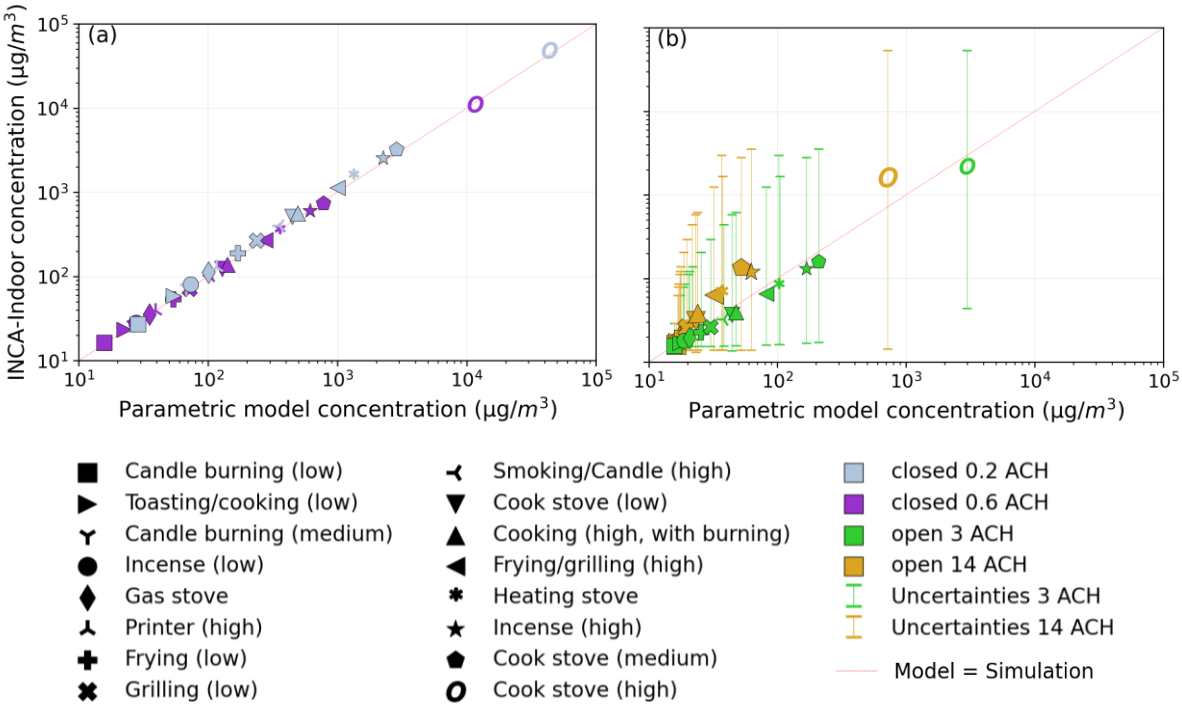


Figure 4-6: INCA-Indoor v/s parametric model 24-h average indoor PM<sub>2.5</sub> concentrations from different activities for (a) closed windows at 0.2 and 0.6 ACH and (b) air renewal of 3 and 14 ACH for windows always open, and uncertainties linked to window opening scenarios

Activities in airtight buildings with windows closed (0.2 ACH) or open before the activity lead to the highest concentrations. In Figure 4-6 (a), we note that there is no uncertainty bar related to scenarios with 0.2 and 0.6 ACH since windows are considered to be closed. In scenarios with 3 and 14 ACH, windows can be always open, or open before, during or after the activity. Lowest concentrations (lower end of the error bar) correspond to scenarios where windows are open during the



emission, evacuating almost all emitted particles and leading to a concentration approximately equal to  $\bar{C}_{\text{out}}$ . Highest concentrations are linked to windows always being closed, with an infiltration rate of 0.2 ACH, or scenarios where the windows are open before the activity, hence not affecting activity-related concentrations. The uncertainty factors between the parametric model and the dynamic simulation, based on the root mean squared log of error (RMSLE), are 1.18, 1.00, 1.03 and 1.14 for the following scenarios respectively: closed window at 0.2 ACH and 0.6 ACH and window always open at 3 ACH and 14 ACH. The average percentage error is <3%. The uncertainties are linked to the variations in ACH due to meteorological conditions affecting natural ventilation rates, which are not considered by the parametric model (see [Annex Figure 8](#)).

#### 3.1.4. Contribution of outdoor PM<sub>2.5</sub> levels on indoor concentrations

Dynamic and parametric model concentrations were calculated for an average European city with an average background PM<sub>2.5</sub> level of 16 µg/m<sup>3</sup>, while, for different cities in the world, outdoor PM<sub>2.5</sub> concentrations can range from 4 µg/m<sup>3</sup> to 200 µg/m<sup>3</sup> (WHO 2016). [Figure 4-7](#) illustrates the effect of different outdoor PM<sub>2.5</sub> levels on indoor concentrations for different ventilation scenarios and three activities representing low, medium and high emission rates: candle burning (low), frying (low) and cook stove (high).

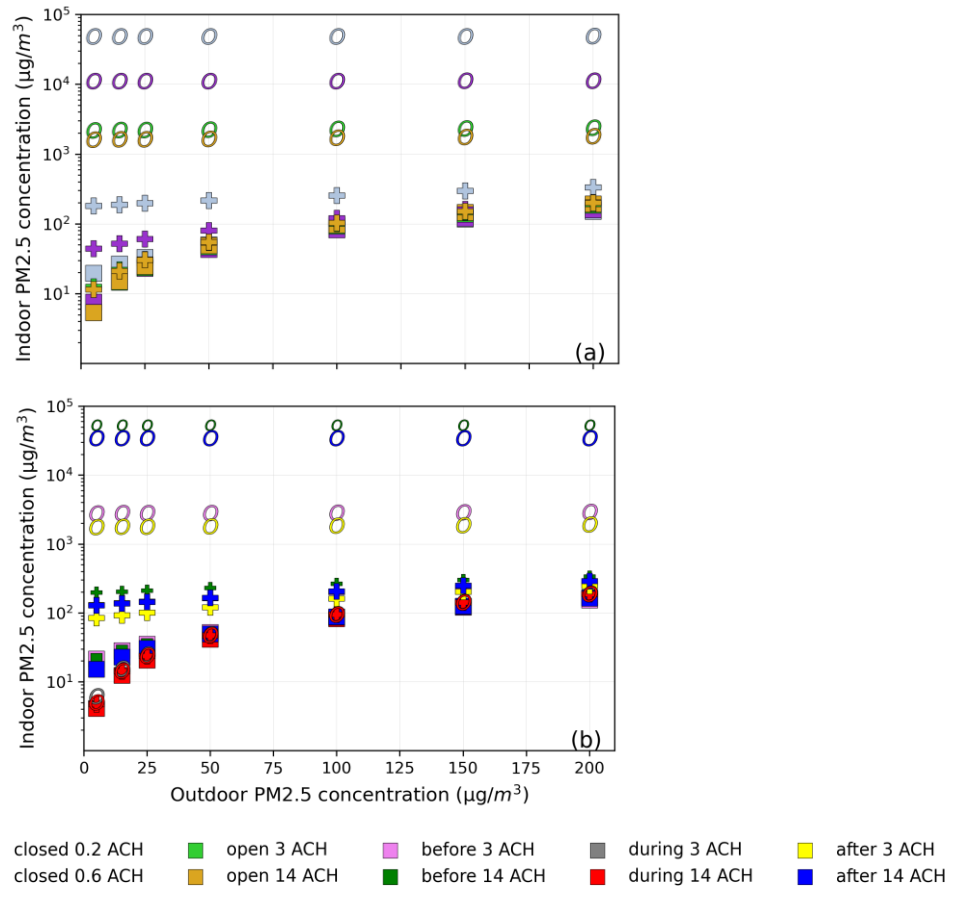


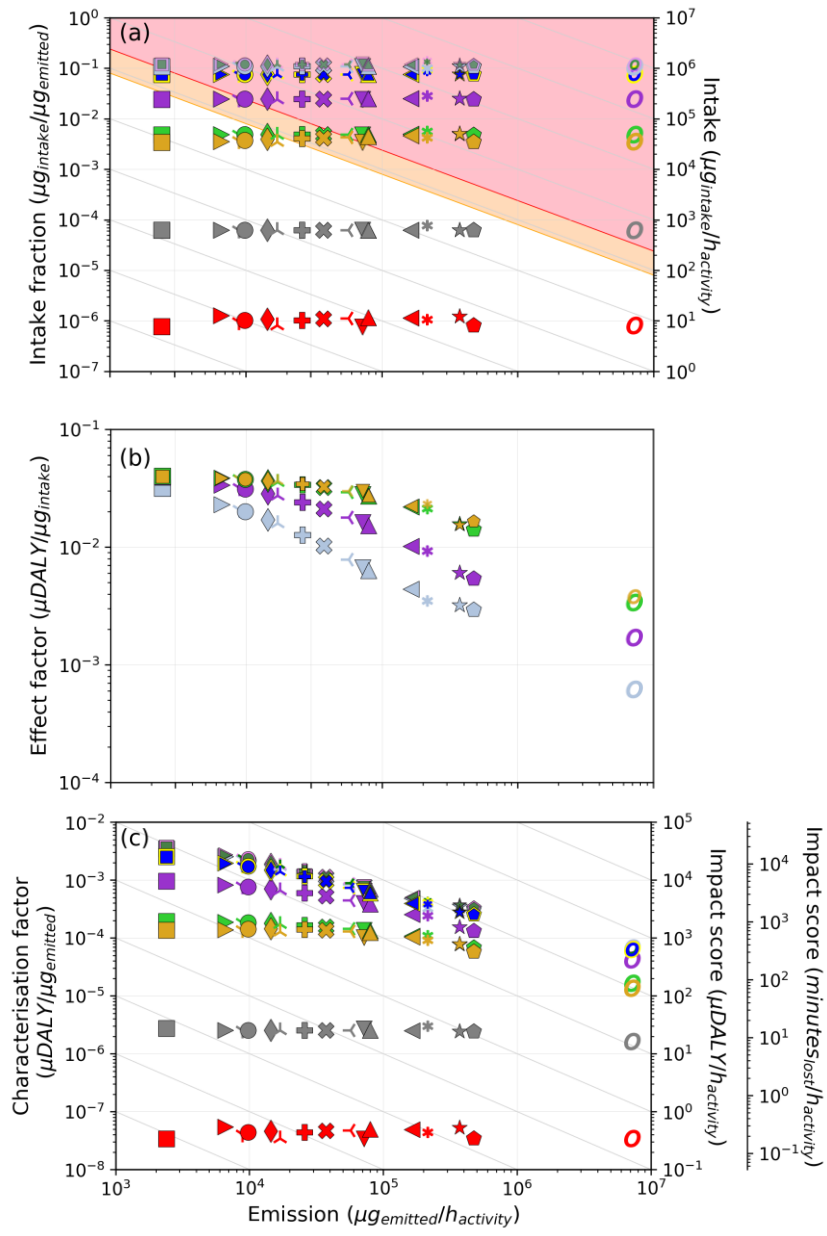
Figure 4-7: Indoor  $PM_{2.5}$  concentrations for three different activities and seven outdoor concentrations ( $4 - 200 \mu\text{g}/\text{m}^3$ ) for a) four standard ACH: (a) 0.2 ACH and 0.6 ACH with **closed** windows and 3 ACH and 14 ACH with windows **always** open, and (b) 3 ACH or 14 ACH for different opening scenarios (**open before**, **open during** and **open after**)

We note from Figure 4-7 that, for activities with high emissions such as incense burning, outdoor concentrations have low relative effect on indoor concentrations (only up to 6% increase), while they can lead to an 8-fold increase for low-emission activities such as candle burning (low). At very high outdoor concentrations and for low emission rates such as candle burning, average concentrations are lower for closed window scenarios since the highest contribution to indoor  $PM_{2.5}$  is outdoor air. Outdoor  $PM_{2.5}$  concentrations over one year (2018) are presented in Annex Figure 9, showing important variations which can range from a minimum of  $1 \mu\text{g}/\text{m}^3$  to a maximum of  $103 \mu\text{g}/\text{m}^3$  during episodes of high air pollution (especially in urban areas during summer).

### 3.2. Intake fractions, effect factors and characterisation factors

Intake quantities and impacts for each activity are calculated **per occupant** and **per hour** of activity. The uncertainty factor between the intake ( $\mu\text{g}$ ) calculated by parametric and the dynamic model are 1.0, 1.05, 1.04 and 1.37 respectively for the standard scenarios with 0.2 ACH, 0.6 ACH, 3 ACH and 14 ACH.

Intake fractions ( $\mu\text{g}_{\text{intake}}/\mu\text{g}_{\text{emitted}}$ ) and intake rates ( $\mu\text{g}_{\text{intake}}/\text{h}_{\text{activity}}$ ) calculated using the dynamic concentrations for all activities and ventilation scenarios are shown in [Figure 4-8 \(a\)](#). Orange and red lines represent the annual and daily exposure recommendations from the WHO air quality guidelines respectively (WHO 2021).



- |                                  |                                |                       |                 |
|----------------------------------|--------------------------------|-----------------------|-----------------|
| — iso-intake/impact lines        | ▲ Printer (high)               | * Heating stove       | ■ open 14 ACH   |
| — annual exposure recommendation | ✚ Frying (low)                 | ★ Incense (high)      | ■ before 3 ACH  |
| — daily exposure recommendation  | ✘ Grilling (low)               | ● Cook stove (medium) | ■ before 14 ACH |
| ■ Candle burning (low)           | ↖ Smoking/Candle (high)        | ○ Cook stove (high)   | ■ during 3 ACH  |
| ▶ Toasting/cooking (low)         | ▼ Cook stove (low)             | ■ closed 0.2 ACH      | ■ during 14 ACH |
| γ Candle burning (medium)        | ▲ Cooking (high, with burning) | ■ closed 0.6 ACH      | ■ after 3 ACH   |
| ● Incense (low)                  | ◀ Frying/grilling (high)       | ■ open 3 ACH          | ■ after 14 ACH  |
| ◆ Gas stove                      |                                |                       |                 |

Figure 4-8: (a) Intake fractions ( $\mu\text{g}_{\text{intake}}/\mu\text{g}_{\text{emitted}}$ ) for all activities and ventilation scenarios on the primary y-axis and total intake ( $\mu\text{g}_{\text{intake}}/\text{h}_{\text{activity}}$ ) on the secondary y-axis, with iso-intake diagonal lines in grey and annual and daily recommendations represented by yellow and red lines, (b) effect factors ( $\mu\text{DALY}/\mu\text{g}_{\text{intake}}$ ) for all activities and four standard ventilation scenarios and (c) characterisation factors ( $\mu\text{DALY}/\mu\text{g}_{\text{emitted}}$ ) on the primary y-axis and health damages ( $\mu\text{DALY}/\text{h}_{\text{activity}}$  and  $\text{minuteslost}/\text{d}$ ) on the two secondary y-axes (left and right) with iso-impact diagonal lines in grey.

Intake fractions ( $\mu\text{g}_{\text{intake}}/\mu\text{g}_{\text{emitted}}$ ) calculated by equation (4-2) are different for each scenario but independent of the emission rate: they depend on breathing rate, occupancy, particle deposition rate and ACH (Fantke et al. 2017). Given a ventilation scenario, the total intake ( $\mu\text{g}_{\text{intake}}/\text{h}_{\text{activity}}$ ), represented by diagonal grey iso-intake lines, is higher for activities with higher emission rates. Markers in the orange and red zone indicate activity and window-opening combinations that lead to concentrations, and consequently intake quantities, beyond WHO guidelines. These include low-emission activities such as lighting a candle in a closed airtight building at 0.2 ACH, which is a possible scenario. Unless having a very high ventilation rate (windows open only during activity, with a 24-hour average of 14 ACH) during the use of a very high-emission cook stove, all scenarios lead to intake well above guidelines.

Figure 4-8 (b) shows that effect factors decrease with increasing emission rate for each scenario, since they depend on indoor  $\text{PM}_{2.5}$  concentrations. Characterisation factors  $CF$ , product of  $iF$  and  $EF$ , also vary across activities and scenarios. Least impacts occur when windows are open during activities, especially if ventilation rates are very high (e.g. with cross ventilation). We also note that indoor fuel burning for cooking (high, using maize straw) can lead to very high health impacts of  $484 \mu\text{DALY}/\text{h}_{\text{activity}}$  (about 4  $\text{hours}_{\text{lost}}$ ) in closed buildings at 0.2 ACH. However, these ACH are unlikely for this activity, occurring in rural homes where buildings with potentially high infiltration rates. Furthermore, occupants are might ventilate during the use of the cook stove, which is a source of heat. Air renewal rates are more likely to be around 3 to 14 ACH, resulting in 98 to  $119 \mu\text{DALY}/\text{h}_{\text{activity}}$  (52 to 63  $\text{minutes}_{\text{lost}}/\text{h}_{\text{activity}}$ ). On the other hand, candle burning can potentially occur in airtight buildings with closed windows, at 0.2 or 0.6 ACH, leading to 7 to  $20 \mu\text{DALY}/\text{h}_{\text{activity}}$  (4 to 11  $\text{minutes}_{\text{lost}}/\text{h}_{\text{activity}}$ ). Results for the larger room are given in Annex Figure 10.

Table 4-5 presents impact scores for all activities and all scenarios in the  $30 \text{ m}^3$  room.

Table 4-5: Impact scores ( $\mu\text{DALY}/h_{\text{activity}}$ ) for  $\text{PM}_{2.5}$  emission from 19 activities for 10 ventilation scenarios in a small room ( $30 \text{ m}^3$ )

Window ACH	Impact score ( $\mu\text{DALY}/h_{\text{activity}}$ )									
	closed 0.2	closed 0.6	open 3	open 14	before 3	before 14	during 3	during 14	after 3	after 14
Candle burning (low)	8.24	2.31	0.47	0.33	8.84	8.81	0.01	0.00	6.06	6.00
Toasting/cooking electric (low)	16.85	5.46	1.24	0.91	18.15	17.95	0.02	0.00	12.90	12.74
Candle burning (medium)	20.23	6.95	1.64	1.23	21.59	21.54	0.02	0.00	15.82	15.69
Incense (low)	21.52	7.52	1.80	1.38	22.93	22.86	0.02	0.00	16.90	16.78
Gas stove	26.88	10.13	2.57	2.08	28.43	28.35	0.04	0.00	21.60	21.50
Printer (high)	29.21	11.31	2.93	2.16	30.77	30.77	0.04	0.00	23.77	23.69
Frying (low)	35.83	15.41	4.30	3.61	37.40	37.35	0.07	0.00	29.98	29.92
Grilling (low)	41.60	19.61	5.85	5.07	43.19	43.13	0.09	0.00	35.65	35.61
Smoking/Candle (high)	48.64	25.36	8.25	7.40	50.39	50.31	0.14	0.00	42.40	42.40
Cook stove (low)	52.28	28.54	9.74	7.37	56.12	56.04	0.19	0.00	46.35	46.42
Cooking (high, with burning)	55.24	30.39	10.68	9.90	57.47	57.35	0.20	0.00	47.96	48.01
Frying/grilling (high)	78.62	41.71	17.76	17.05	82.90	82.71	0.41	0.01	64.39	64.73
Heat stove	61.06	45.68	20.82	17.25	81.11	85.37	0.57	0.01	77.63	74.94
Incense burning (high)	130.6	56.91	29.01	29.34	138.1	137.7	0.91	0.02	103.3	104.1
Cook stove (average)	151.5	62.56	32.33	27.27	159.0	158.9	1.14	0.02	120.1	121.2
Cook stove (high)	484.0	302.2	118.6	97.81	481.4	481.2	11.79	0.25	457.7	459.1

Impacts range from  $8.2 \times 10^{-5} \mu\text{DALY}/h_{\text{activity}}$  (low emission and very high ventilation rate during activity) to  $4.84 \times 10^2 \mu\text{DALY}/\text{FU}$  (high emission and very low ventilation rate). In comparison, spending one full day in the centre of the French city of Lille leads to an average impact of  $1.03 \times 10^1 \mu\text{DALY}$  (see Annex Figure 9 for outdoor  $\text{PM}_{2.5}$  concentrations in Lille). Tables summarising effect factors, characterisation factors and impact scores for each activity and scenario for both small and large rooms ( $30 \text{ m}^3$  and  $67 \text{ m}^3$ ) are given in Annex Table 11 to Annex Table 15.

### 3.3. Individual Lifetime Risk

The individual lifetime risk (*ILR*) (DALY/person/lifetime), representing the number of life years lost from the five disease outcomes, are calculated for each activity for an average scenario of 0.6 ACH air change rate from equation (4-8) and are presented in Figure 4-9. It is considered that an individual is exposed PM<sub>2.5</sub> resulting from daily one-hour activities over a lifetime of 86 years (WHO 2020). The ILR for 30-minute and 2-hour activity durations are also given.

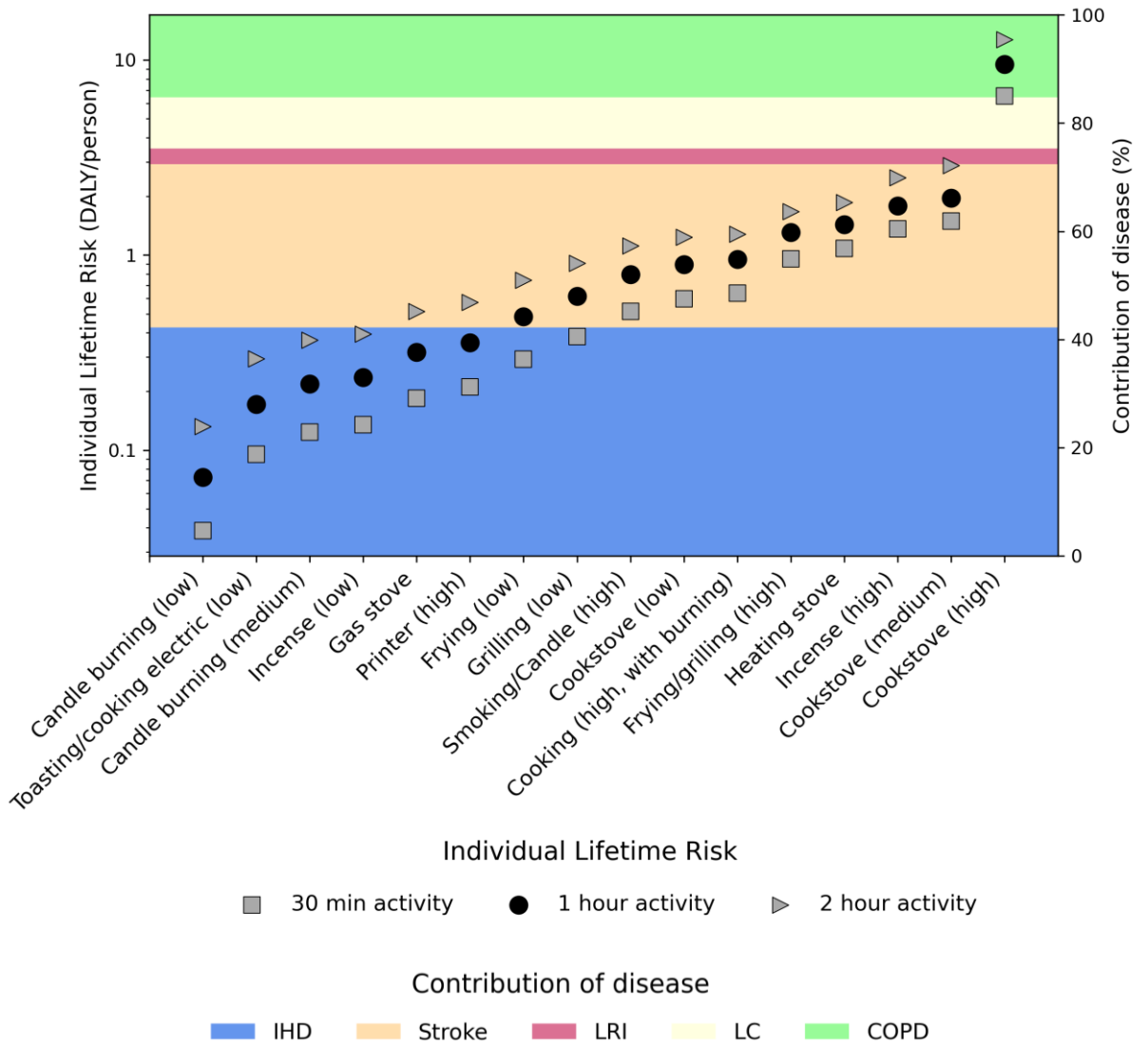


Figure 4-9: Total individual lifetime risks (DALY/person) given by black markers for one-hour activities at 0.6 ACH and grey markers for 30-minute and two-hour durations on the left y-axis, with contributions of each disease (ischaemic heart disease IHD, stroke, lower respiratory infections LRI, lung cancer LC and chronic obstructive pulmonary disease COPD) given on the right y-axis

Activities with very high emissions can lead to a loss of up to 10 years of life, distributed as follows for the use of cook stove (high): 42% of the risk is related to IHD, 30% to stroke, 3% to LRI, 9% to LC and 15% to COPD. We note variations in the distance between the ILR for the default activity

duration (one hour) and that of each two additional durations (30 minutes and 2 hours) for different activities. These are explained by two non-linearities, one associated with the effect factor and one associated with the intake. First, the exposure-response model underlying the effect factors is supra-linear and dependent on indoor  $PM_{2.5}$  concentrations (see equation 3), which are a function of both indoor activity emissions and outdoor  $PM_{2.5}$  levels. Second, intake considers the concentration increment associated with a specific activity, and is hence non-linearly dependent on indoor  $PM_{2.5}$  concentrations (especially at very low emission rates, where outdoor  $PM_{2.5}$  have a higher relative influence on indoor  $PM_{2.5}$  concentrations).

#### 4. Conclusions and perspectives

In this chapter, we derived a set of intake fractions, effect factors, characterisation factors and impact scores for 19 one-hour activities and 10 different ventilation scenarios. We note that, at very low ACH, all activities induced exposure concentrations beyond WHO recommendations. High or very high ventilation *during* all activities allowed to reduce concentrations well below these recommendations. Characterisation factors for the one-hour activities (or any other activities with corresponding emission rates) provided in this study can be integrated to LCIA methods and the framework proposed can help to devise optimal ventilation strategies in building design. The derived impact scores ( $CF \times ER_{activity}$ ) for an activity unit of one hour can be scaled by activity duration to obtain actual activity impacts. Indoor concentrations depend on indoor settings (e.g. window layout and opening scenarios), outdoor  $PM_{2.5}$  level and activity duration. These parameters influence the indoor exposure  $PM_{2.5}$  level used to calculate effect factors and ultimately the characterization factor. The emission rate depends on the activity but is independent of indoor settings.

The parametric model agrees with the dynamic model when the ACH shows little variation during the day, but cannot integrate ventilation scenarios, which are given uncertainties of this type of model. The current version of the dynamic model considers window opening scenarios with one-hour time steps, while in reality, occupants can open windows for only a few minutes, especially during cold weather. Ventilation durations are thus potentially overestimated, leading to underestimations in concentrations and impacts, in particular for the *open during* and *open after* scenarios. In the particular case of wood stove heating, opening windows during the activity is counterproductive, leading to higher heating needs. Furthermore, all heat-generating activities induce convection, and height could play an important role in exposure concentrations (Ainiwaer et al. 2022), but the model used in this study does not account for spatial differentiations. Finally, the factors provided do not consider multiple occurrences of different activities at the same time, which could increase concentrations and lead to variations in effect factors and impact scores. They may be applied to LCIA for representative archetypes, but simulations are needed for specific case studies. The effect factor model includes some limitations. First of all, we do not consider the composition of  $PM_{2.5}$  and the toxicity of the substances



they contain (e.g. the presence of metals: Mainka and Fantke, 2022). Moreover, the model does not consider the years of life lived with a disability for each disease outcome in the characterisation of impacts: only the number of years lost due to premature death are considered.

Perspectives include the study of the influence of PM toxicity if compositions of emitted particles for specific activities are known, and the calculation of characterisation factors and impact scores for a wider range of activities. Studies focusing on fugitive emissions are few and fairly recent. With growing awareness around health impacts of indoor solid fuel burning for heating or cooking, more data could be available and allow to derive impact scores for different types of stoves. Secondly, effect factors were calculated based on ambient PM<sub>2.5</sub> concentrations, though occupants are exposed mainly to indoor concentrations which are often higher than outdoors: non-linear effect factors are hence possibly overestimated. Representative building or activity scenario archetypes can be modelled to calculate annual exposure concentrations considering the fractions of time spent indoors and outdoors.

Indoor activities that generate PM<sub>2.5</sub> are also likely to emit other pollutants such as VOCs, CO or NO<sub>x</sub> (Salthammer, Mentese, and Marutzky 2010), and further study to include them in the calculation of impact scores is recommended. Finally, with a life cycle perspective, though increased ventilation rates help to decrease impacts related to indoor PM<sub>2.5</sub>, they might also increase energy consumption for heating. It is thus important to consider heating as a source of impacts and identify best trade-offs that allow to reduce damages from PM<sub>2.5</sub> and heating altogether. Heating needs of the building and impacts generated can be calculated using an energy simulation and LCA software, such as Pleiades (IZUBA ÉNERGIES 2001a). However, despite these needs for additional research, our framework and related factors constitute a valuable starting point for addressing different indoor activities and integrate their PM<sub>2.5</sub>-related emissions, exposures and health effects into LCA and environmental footprint studies.

# 5. General framework integrating IAQ impacts to building LCA: application to a case study

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Dans ce chapitre, nous proposons une méthode prenant en compte la qualité de l'air intérieur (QAI) dans l'analyse de cycle de vie (ACV) des bâtiments. Cette méthode associe les modèles décrits dans les chapitres 2, 3 et 4 afin de quantifier les impacts des polluants intérieurs (COV et  $PM_{2.5}$ ), émis par des matériaux et activités, sur la santé des occupants. Nous démontrons son applicabilité avec des cas d'étude : trois pièces dans un bâtiment tertiaire et deux pièces dans un bâtiment résidentiel. Les impacts liés à la consommation d'énergie (du chauffage et des ventilateurs) sont couplés aux impacts de la QAI pour différents taux de ventilation dans chaque pièce, et le taux minimisant le total des impacts est identifié.

Les  $PM_{2.5}$  et le chauffage sont les principales sources d'impacts dans ces cas d'étude, responsables respectivement de 40% à 94%, et 1% à 31% des impacts totaux de chaque pièce sur son cycle de vie (1100  $\mu$ DALY/an à 2570  $\mu$ DALY/an en considérant QAI et ACV). En fonction des activités et de l'occupation des pièces, le taux idéal de ventilation est différent : 0.2 ACH dans la salle de réunion, 0.9 ACH dans le bureau, et 5.8 ACH dans la cuisine (avec de forts taux d'émissions). Les taux de ventilation optimaux sont aussi dépendants de la source d'énergie qui influence aussi bien les impacts ACV que ceux de la QAI : le chauffage au charbon et le chauffage électrique avec ventilateur émettent des  $PM_{2.5}$ , augmentant ainsi les impacts de la QAI de jusqu'à 2.5 fois par rapport au chauffage au gaz. D'après une étude de la disposition des fenêtres dans la cuisine résidentielle, nous concluons que les cuisines devraient, si possible, être placées dans des pièces avec de la ventilation traversante ou avec deux fenêtres perpendiculaires. Nous explorons aussi des solutions visant les principales sources d'impacts : une ventilation double-flux avec échangeur de chaleur pour réduire le besoin énergétique, et des filtres pour réduire significativement les concentrations de  $PM_{2.5}$ . Une combinaison des deux permet de réduire de 56% les impacts totaux du bâtiment étudié.

Ce chapitre montre que la méthode développée est applicable à l'écoconception des bâtiments afin d'élaborer des stratégies de ventilation optimales. Elle peut aussi servir à faire évoluer la réglementation en matière de santé publique, à travers des archétypes représentatifs du secteur tertiaire ou résidentiel, en fournissant des recommandations en termes de conception et de ventilation.

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Life cycle assessment (LCA) methods are used in building ecodesign, but do not currently consider indoor air quality (IAQ). Since we spend about 85% of our time indoors, and are exposed to potentially hazardous substances, IAQ is of particular importance to human health. Its consideration in LCA could help make adequate design choices (e.g. materials, window layouts or ventilation rates) and reduce the building's impacts, while avoiding their transfer to other life cycle stages.

To address this gap, we propose a methodology, based on the previous chapters, combining building LCA and models that encompass the whole pollutant pathway, from emission to quantified impacts on human health using the disability-adjusted life years (DALYs) indicator. We account for volatile organic compounds (VOCs) and fine particulate matter (PM<sub>2.5</sub>), emitted from materials and indoor activities. An optimal ventilation rate allowing to reduce overall IAQ and LCA impacts (namely from energy for space heating and ventilation fans) is identified. The framework's applicability is demonstrated on a case study: different rooms having distinct uses, occupancy and activity patterns, lead to different emission rates, impacts and optimal ventilation rates. The influence of heat sources (gas, electricity, wood) on optimal rates is assessed and different window layouts for natural ventilation are tested.

PM<sub>2.5</sub> and heating are the main sources of impacts, respectively ranging from 40% to 94%, and 1% to 31% of total impacts of each room, which range from 2500  $\mu$ DALY/year to 14200  $\mu$ DALY/year. Rooms with higher indoor emissions have higher optimal ventilation rates: 1.2 ACH (air changes per hour), 2.9 ACH and 13.2 ACH in the meeting room, office and kitchen respectively. These rates also vary for different heat sources due to their different IAQ and LCA impacts: 2.7 ACH, 5 ACH and 15 ACH for coal (still a common fuel in rural Asian countries), gas and electric fan heating respectively in the living room. The combined use of double-flow ventilation to lower heating needs and filters that reduce PM<sub>2.5</sub> concentrations leads to a 56 % decrease in total impacts of the meeting room.

This chapter shows the applicability of the framework to building ecodesign. For instance, distinct optimal ventilation strategies can be devised, depending on the room or building use. The framework can also have a regulatory application in public health, through representative archetypes, by providing general recommendations in the tertiary and residential sectors.

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## 1. Introduction

The integration of indoor air quality (IAQ) into the life cycle assessment (LCA) of buildings necessitates the development of models that characterise impacts of indoor air pollutants on occupant health with LCA indicators. We study two main pollutant categories commonly present indoors with quantified effects on health: volatile organic compounds (VOCs) and fine particulate matter (PM<sub>2.5</sub>). The main sources linked to the release of IAPs (indoor air pollutants) are materials and activities for VOCs, and, for PM<sub>2.5</sub>, outdoor air pollution and activities. The need for frameworks covering the entire pollutant pathway were identified, linking materials and indoor activities to human health impacts considering the endpoint unit used in LCA: Disability-Adjusted Life Years (DALYs).

To address these gaps, in the previous chapters, frameworks to evaluate health impacts of VOCs and PM<sub>2.5</sub> emitted by materials, occupants and their activities in indoor environments were developed. Important conclusions were made regarding each pollutant type and source, and are considered in the present chapter. We observed that indoor air chemistry was an important factor to consider, since chemicals are both consumed and produced, having direct effects on occupants. Outdoor PM<sub>2.5</sub> pollution, dependent on the building's location, affected indoor concentrations. On the other hand, direct occupant sources of VOCs and PM<sub>2.5</sub> (skin, breath and clothes) were found to be negligible and will not be studied in this chapter. We also assessed the importance of capturing ventilation and emission dynamics which can lead to substantial uncertainties on indoor air concentrations: up to a factor 780 for very high emission rates. It is hence important to study indoor air concentrations with simulations that capture temporal variations in emission rates, air flow rates and outdoor temperatures, coupled with dynamic occupant presence in different rooms to model their intake.

While high ventilation rates during emissions led to the decrease of indoor pollutant concentrations, if outdoor temperatures are low, occupant comfort could be affected. This can lead to an increase in energy consumption for heating. If mechanical ventilation is used, higher ventilation rates also lead to higher electricity consumption for the fans. Since IAQ impacts decrease, but impacts of heating and electrical fans increase, an optimal ventilation rate that yields lowest impacts can be identified.

The aim of this chapter is to develop a methodology integrating IAQ into building LCA, in order to determine optimal building design parameters. Its applicability is demonstrated on a case study to identify optimal ventilation rates for different rooms and scenarios. The different steps for reaching the objective are to:

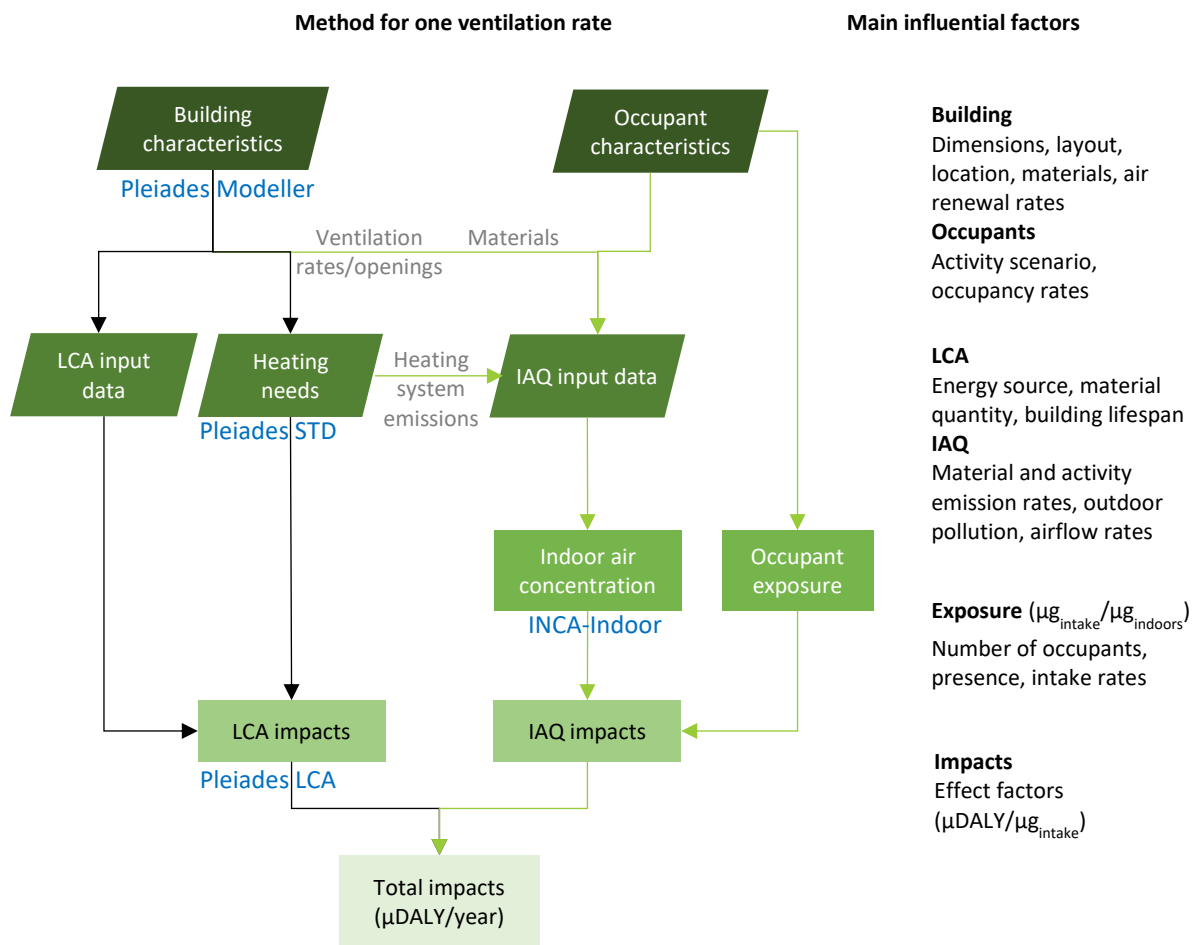
1. Calculate LCA impacts of the case study building on all life cycle stages

2. Calculate health impacts linked to IAQ, accounting for VOCs and PM<sub>2.5</sub> emitted by materials and occupant activities
3. Couple IAQ and LCA impacts for a variety of ventilation rates to determine optimal values allowing to reduce overall health impacts for different rooms

Using different variants of a case study, we aim at demonstrating the applicability of the methodology developed to decision-making in building construction.

## 2. Materials and methods

The general approach followed to integrate IAQ impacts to building LCA is presented in [Figure 5-1](#). The different parts of the method (for a given ventilation rate) are shown with main influential factors at each step and the method is described with fuller details in the next sections. As an example, the methodology is applied using existing software: Pleiades modeller allowing to create a building model, Pleiades STD to perform dynamic thermal simulation (Peuportier and Blanc Sommereux 1990), Pleiades LCA for life cycle assessment (Polster et al. 1996), and INCA Indoor for IAQ calculations (Mendez et al., 2015).



*Figure 5-1: General approach followed to integrate IAQ impacts to building LCA for a given ventilation rate*

We model the building under study using Pleiades Modeller (IZUBA ÉNERGIES 2001a) which allows to define building characteristics: design, layout, materials, mechanical ventilation rates, openings (windows and doors), location and occupancy. Heating needs are calculated with Pleiades STD (Peuportier and Blanc-Sommereux 1990) based on outdoor climate, thermal properties of building materials and use characteristics (e.g. occupancy, internal heat gains, temperature set point). For the building's LCA, we use Pleiades LCA (Polster et al. 1996; Thiers and Peuportier 2012), considering all *stages* of its life cycle: *construction*, *operation* (heat and electricity consumption based on calculated heating needs and hot/cold water consumption based user inputs), *renovation* and *deconstruction*. The LCA database ecoinvent v3.4 cutoff (Frischknecht et al., 2004) is used.

Material VOC emissions are calculated considering a material age of one year, after which emissions are stabilised (see Annex Figure 11). Activity VOC and PM<sub>2.5</sub> emissions ( $\mu\text{g}/\text{h}$ ) are obtained from literature, and summarised in Chapters 3 and 4. The heating system PM<sub>2.5</sub> emissions also depend on the calculated heating energy needed, since more fuel is burnt for higher

consumption, leading to higher emissions. Using the INCA-Indoor model (Octopus Lab 2017), which considers the building specifications obtained from Pleiades Modeller (room geometry, mechanical ventilation rates, opening of windows and doors), and pollutant emission rates based on materials and activity scenarios, indoor air concentration of pollutant  $x$   $C_{in,x}$  ( $\mu\text{g}/\text{m}^3$ ) are calculated. These are coupled with occupant exposure ( $\mu\text{g}_{\text{intake}}/\mu\text{g}_{\text{indoors}}$ ) through different pathways according to dynamic occupancy. Using the effect factors ( $\mu\text{DALY}/\mu\text{g}_{\text{intake}}$ ) of each pollutant, a total IAQ-related health damage is calculated and added to LCA impacts in  $\mu\text{DALYs}$ .

Variations in ventilation rates are tested on a case study in order to identify optimal trade-offs that allow to reduce overall health impacts. In order to demonstrate possible applications of the developed methodology in the tertiary and residential sector, we study virtual variants of a case study, with particular attention to different heat sources and window layouts.

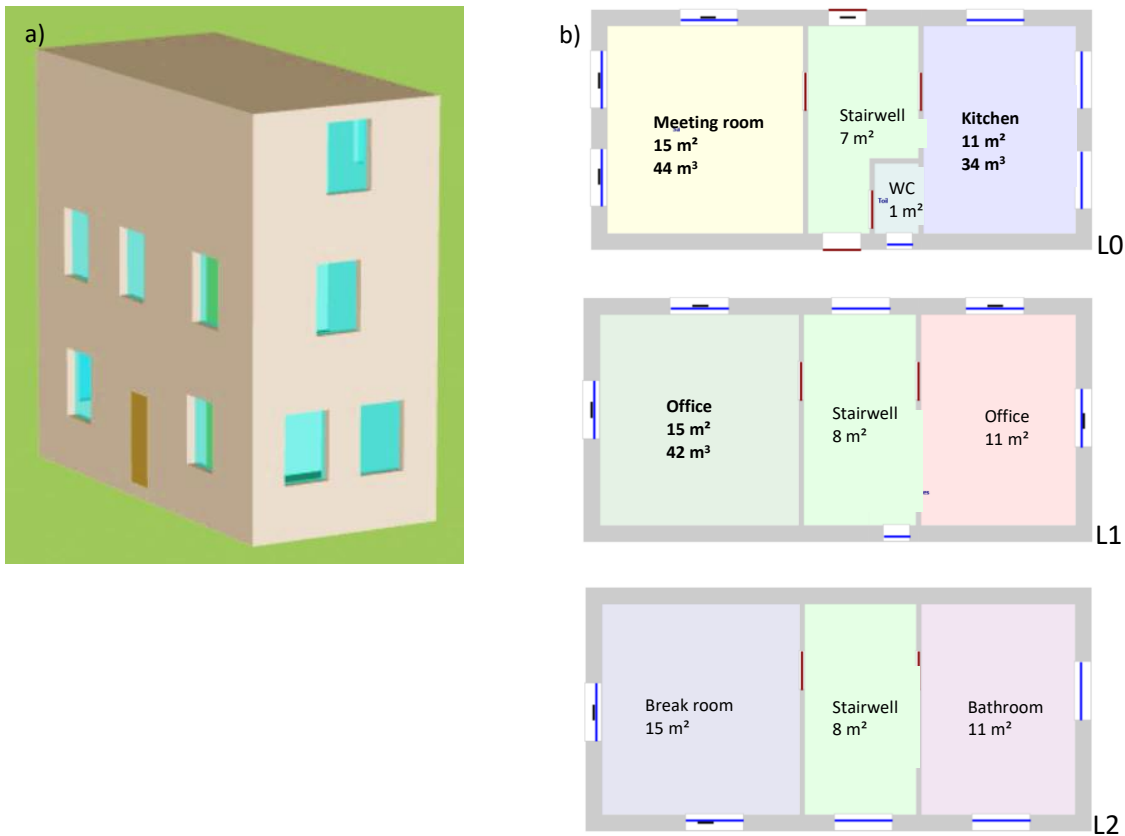
## 2.1. Building characteristics

The case study used to illustrate the methodology corresponds to the office building of Octopus Lab (developer of INCA-Indoor) situated at La Madeleine in the north of France.

### 2.1.1. Structure

The building has three levels and divided into different rooms as illustrated in [Figure 5-2](#).



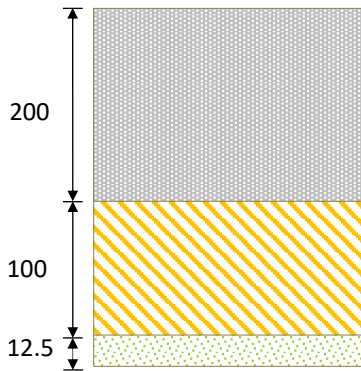


*Figure 5-2: a) 3D representation of the case study building and b) floor plan of different levels and room surface areas*

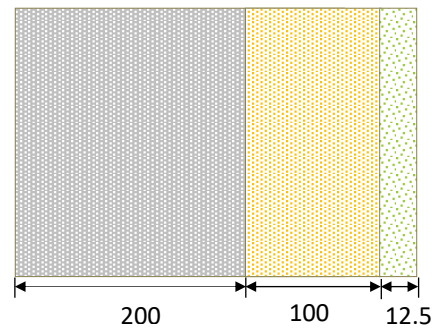
Three out of the ten rooms are selected: meeting room, kitchen and the 15 m<sup>2</sup> office. The selected rooms represent distinct building uses, which not only lead to different choices of materials (in particular flooring: bamboo flooring for the meeting room, PVC flooring for the kitchen and carpet for the office, as shown in [Figure 5-3](#)), but also room-specific occupant activities. These result in different VOC and PM<sub>2.5</sub> emission/concentration dynamics, occupancy rates, and ultimately different health impacts.

The material compositions of the ceiling, bearing walls, partition walls and flooring for the three rooms are illustrated in [Figure 5-3](#). The compositions of different material layers were obtained from the occupants, but estimations had to be made for some thicknesses based on common building compositions.

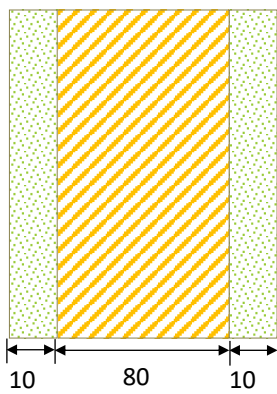
a) Ceiling



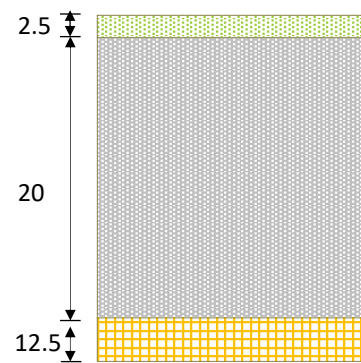
b) Bearing walls



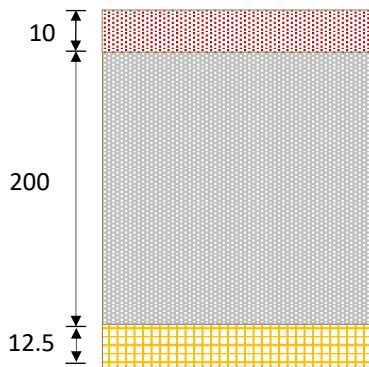
c) Partition walls



d) Kitchen floor



e) Meeting room floor



f) Office floor

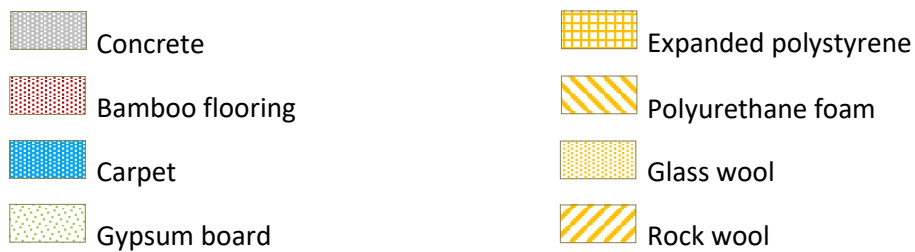
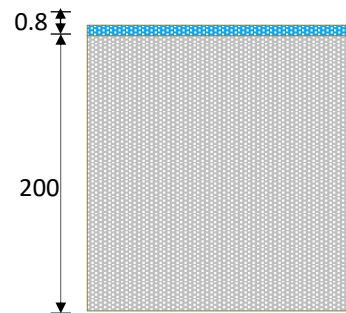


Figure 5-3: Material composition of the a) ceiling, b) bearing walls, c) partition walls, d) kitchen floor, e) meeting room floor, and f) office floor

### 2.1.2. Occupancy

The kitchen is occupied by six persons from 8:30 to 9:00, 12:30 to 13:30 and from 16:00 to 16:30. The office is occupied by three persons for longer periods: from 9:00 to 12:30 and from 13:30 to 18:00. The building is heated by a collective gas boiler (located in a different compartment than those studied), but two alternatives are also studied: a portable electrical fan heater, and a coal stove. Coal heat stoves are still a common practice in rural India or China (Shen et al. 2020; Li et al. 2022).

### 2.1.3. Ventilation rates

All rooms are subject to a normalised infiltration rate  $I_4$  of 1.7 m<sup>3</sup>/h/m<sup>2</sup> under 4 Pa, which is on the low-end of infiltration rates in tertiary buildings (Carrié et al. 2006). The normalised infiltration rate defines the airflow rate at a pressure difference of 4 Pa between the inner and outer sides of the envelope. The rooms are fitted with mechanical ventilation. The air renewal rate considers both infiltrations and mechanical ventilation. It is expressed in air changes per hour (ACH) – the number of times the room’s air volume is renewed in an hour. Ranges within which the air renewal rate is varied are based on values reported in literature: from 0.2 ACH for closed, airtight buildings (Persily et al. 2010) to 15 ACH for non-OECD countries (Rosenbaum et al. 2015). The average for houses in OECD countries is 0.64 ACH (Rosenbaum et al. 2015).

The minimum air renewal rate for offices and residences is 18 m<sup>3</sup>/h per occupant according to the French regulation on health in indoor environments (decree of 20 November 1979). It can be converted into ACH using the room’s volume and occupancy rate. This results in a minimum air renewal rate of 90 m<sup>3</sup>/h (2 ACH) for the meeting room, 108 m<sup>3</sup>/h (3 ACH) in the kitchen and 54 m<sup>3</sup>/h (1.3 ACH) in the office.

## 2.2. IAQ input parameters

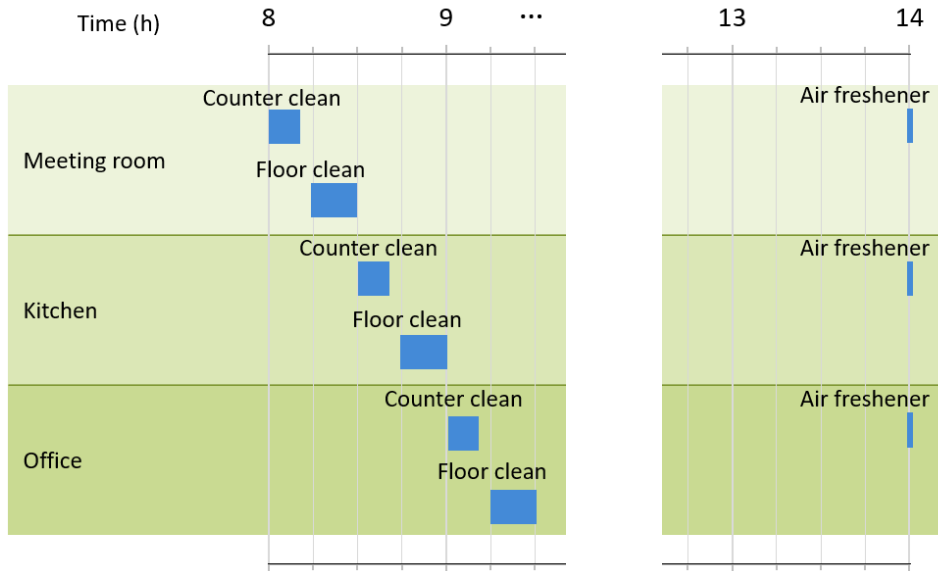
### 2.2.1. Material emissions

Material VOC emissions were calculated using the multilayered emission model described in A1.1, with mass fractions obtained from Pharos (Healthy Building Network 2000) and diffusion and material-air partition coefficients obtained from Huang et al. (2017; 2019). For two materials, namely bamboo flooring and gypsum board, optimal parameters calculated in Chapter 2 (Bhoonah et al. 2023) are used. The multilayered model allows to consider the buffer effect of material layers which affects the emission profile. The age of materials considered is one year and most material thicknesses are estimated. Sources of uncertainty lies in these parameters, since they affect emission rates, as seen in the Sensitivity analysis in chapter 2). However, after one year, material emission rates are stabilised, except for isopentane from expanded polystyrene (see Annex Figure 11). Model data (emitted substances, mass fractions, diffusion coefficients and material-air partition coefficients) are summarised in A4.1.

### 2.2.2. Activity VOC emissions

The activity scenario for one day is presented in [Table 5-1](#), indicating the starting time and duration of activities emitting VOCs. These activities occur once a week, both in the tertiary and residential setting. Substances released by these activities and their emission rates are presented in [Annex Table 6](#) to [Annex Table 8](#).

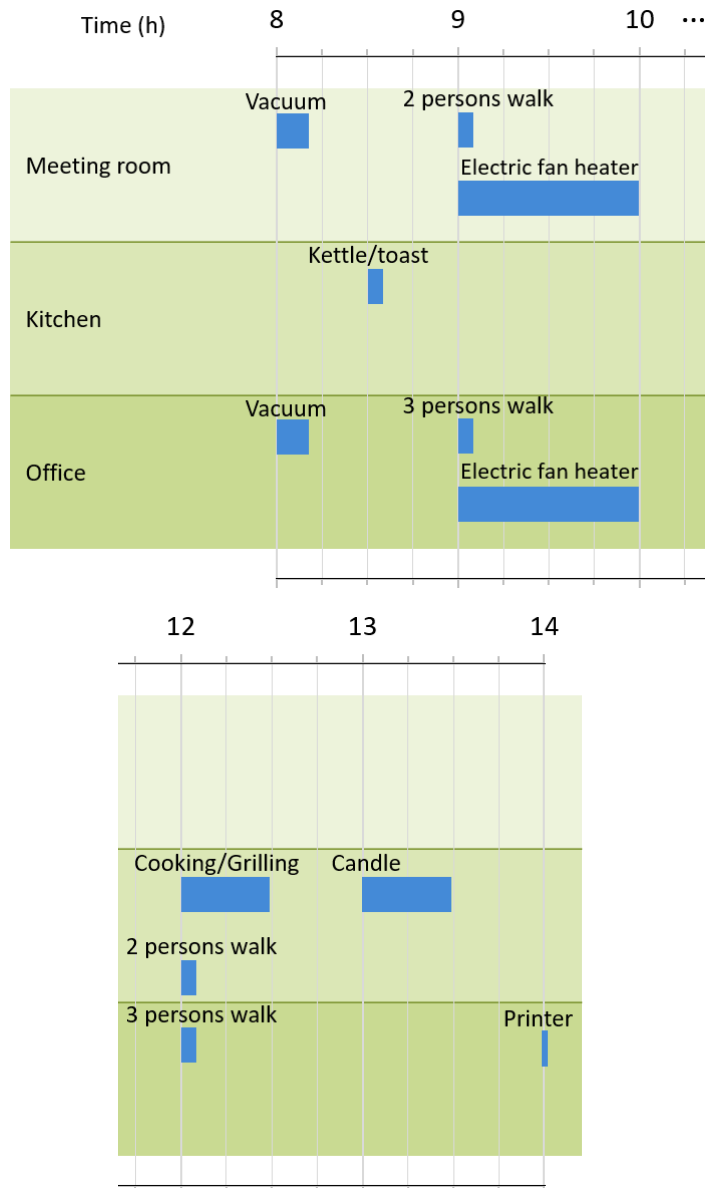
*Table 5-1: VOC-emitting activity scenario in the meeting room, kitchen and office*



### 2.2.3. Emissions and resuspension of PM<sub>2.5</sub>

The activity scenarios for one working day in each room of the tertiary building, indicating the starting time and duration of activities leading to PM<sub>2.5</sub> emission or resuspension are presented in [Table 5-2](#). Emission rates are presented in [Table 4-2](#) and resuspension rates in [Annex Table 9](#).

Table 5-2: Activity scenario in the rooms of the tertiary building (meeting room, office and kitchen)



Activities are given for one day, but some variations are considered throughout the week: grilling occurs once a week instead of cooking on an electric stove and kitchen candle is burnt twice a week. In the office, printing occurs thrice a week. Heaters are used only in winter (from the 1<sup>st</sup> of October to the 30<sup>th</sup> of April).

In this case study, since vacuuming occurs once per week, resuspension rates never reach the maximum value found in literature. They are at a low rate after vacuum cleaning, and reach a medium value for remaining week days (see Annex Table 9). Because the resuspension rate depends on both the dust coverage of the room ( $g_{\text{dust}}/m^2$ ) and the intensity of the activity, we consider a decrease during a few hours after vacuuming. It does not reach zero: though dust coverage is reduced, the vacuuming efficiency ranges between 20% and 75% for PM<sub>2.5</sub> (Xi et al.

2020). We also consider emission rates for vacuum cleaners, measured by Ferro et al. (2004). The exhaust port can contain leaked particles, or emit particles which are produced through the mechanical wear of spinning motor components (Lioy et al. 1999). These particles are again deposited: after about four hours for  $PM_1$  (Wolkoff et al. 1998) and faster for  $PM_{2.5}$  due to their higher mass and diameter. Moreover, the high exhaust air velocity can force deposited particles (especially smaller ones) back into indoor air.

### 2.3. IAQ impact assessment method

Indoor air concentrations of  $PM_{2.5}$  and VOCs are calculated using the INCA-Indoor multizone model for different ventilation rates through the mass balance of pollutants in each room under study. Influential parameters include the building's characteristics (location, dimensions and openings), occupancy rates, meteorological conditions and outdoor pollution (determined by the building's geographical location), and indoor activities and their emission rates in each room. Airflows are simulated using CONTAM (Dols and Polidoro 2020), considering infiltration, exfiltration and flows between rooms based on outdoor and indoor pressures and temperatures. A constant indoor temperature of 20°C is considered in all rooms by INCA-Indoor for the calculation of airflows and indoor pollutant concentrations.  $PM_{2.5}$  concentrations also depend on their deposition rates (K. Lai and Nazaroff 2000) and VOC concentrations depend on their sorption to material surfaces and chemical reactions between indoor substances (Mendez et al. 2015). Air is treated as an ideal gas, obeying to the ideal gas law.

#### 2.3.1. Intake of pollutants

The presence of occupants varies according to the day of the week: there is no presence during weekends in offices, while the presence of occupants is more important during these two days in residential buildings. The total intake of occupants through inhalation ( $m^3/s$ ) is shown in [Annex Figure 12](#). The intake of VOCs by the four pathways (inhalation, ingestion, dermal contact and gaseous dermal uptake) are considered, while only inhalation is considered for  $PM_{2.5}$  since it is mainly absorbed through the respiratory system (Thangavel, Park, and Lee 2022) and current effect factor models consider only inhalation as intake pathway (Burnett et al. 2014; Fantke et al. 2019).

#### 2.3.2. VOC effect factors

The intake of VOCs by the four pathways (inhalation, ingestion, dermal contact and gaseous dermal uptake) are considered and health impacts are calculated based on intake quantities ( $\mu g_{\text{intake}}$ ) and effect factors  $EF$  ( $\mu\text{DALY}/\mu g_{\text{intake}}$ ). Effect factors are calculated based on  $ED_{10}$  or  $ED_{50}$  (the lifetime doses per person that causes a disease probability of 10% and 50% after intake) as described in [A1.3](#), for cancer diseases, reproductive/developmental non-cancer diseases and general non-cancer diseases.

### 2.3.3. PM<sub>2.5</sub> effect factors

PM<sub>2.5</sub> intake by inhalation and consequent health impacts are calculated according to the methods described in Chapter 4 (section 2.1). Effect factors are calculated for each room based on the average exposure to the indoor PM<sub>2.5</sub> concentration  $\bar{C}_{in}$  ( $\mu\text{g}/\text{m}^3$ ) and the ambient PM<sub>2.5</sub> concentration of the city  $\bar{C}_r$  ( $\mu\text{g}/\text{m}^3$ ). Population data for France were obtained from the world population prospects (United Nations 2019) and age-specific French mortality rates,  $M$  (deaths/year), for the five diseases outcomes are obtained from the GBD Collaborative Network for 2019 (GBD Global Burden of Disease Collaborative Network 2019). In this chapter, the calculated effect factor  $EF$  corresponds to exposure to ambient (background) and activity-related PM<sub>2.5</sub> concentrations, considering multiple occurrence of activities according to the scenario.

### 2.3.4. Health impacts

Health impacts are calculated as the product of the total mass intake ( $\mu\text{g}_{\text{intake}}$ ) of substances and their effect factor ( $\mu\text{DALY}/\mu\text{g}_{\text{intake}}$ ). In the case of VOCs, we distinguish between the different intake pathways and related effect factors (see A1.3). For PM<sub>2.5</sub>, we calculate the total impacts as a product of the total intake by inhalation and the effect factor linked to average indoor concentrations,  $EF(\bar{C}_{in})$ .

## 2.4. LCA input parameters

### 2.4.1. Functional unit

The functional unit for each room is summarised in Table 5-3. Variants of the same room should correspond to the same functional unit.

Table 5-3: Quantitative and qualitative functions of each room studied

Room	Surface area (m <sup>2</sup> )	Use	Comfort temperature	Number of occupants
Meeting room	14.8	Host meetings		Up to 10
Kitchen (tertiary or residential)	11.6	Cook and eat	Above 19°C	Up to 5
Office	14.8	Work on computers		Up to 3
Living room	14.8	Family gathering		Up to 5

The building's lifespan is 100 years, and indoor temperatures are maintained at above 19°C in all rooms. In this case study, different ventilation rates are studied without compromising occupant comfort in winter (indoor temperatures). This potentially leads to increase in energy consumption for heating. We do not consider the effect of ventilation on active cooling in summer (air conditioning).

#### 2.4.2. Materials and construction

The materials used in the building and their respective masses, obtained from Pleiades LCA based on the building model on Pleiades Modeleur, are given in [Annex Table 17](#). The distance considered between the production site and the construction site is 100 km.

#### 2.4.3. Operation

We consider that each occupant consumes 10 L of cold water per day in an office building. For the residential variant, it changes to 40 L of hot water and 100 L of cold water per day. The heating power (W) and energy (kWh) needed to achieve a minimum of 19°C temperature in all rooms during working hours are calculated using Pleiades STD, based on outdoor temperatures, air flow rates, insulation, and occupancy. For gas heating, an efficiency on lower heating value (LHV) of 1.02 is considered corresponding to a modern condensing boiler. A prospective attributional hourly mix based on Frapin et al. (2022) was considered to evaluate impacts of electricity consumption. Concerning electric heating, it finally comes to a mix of 57% nuclear, 21% hydro and renewable energies, 20% gas and 2% coal.

The ventilation fan electricity use is also calculated according to ventilation rates, considering a consumption of 0.02 kWh/m<sup>3</sup>. Gas is indicated as the heating source for the actual building, and variants are studied for electrical heating and residential coal heating. The transport of occupants (from office to home, for example) and the waste generated during the operation stage are excluded.

#### 2.4.4. Renovation

The interior/exterior doors and windows are replaced every 30 years, the paints every 10 years and the equipment is changed every 20 years.

#### 2.4.5. End of life

Waste treatments of the different building materials at their end-of-life are given in [Annex Table 27](#). We consider a distance of 20 km between of the construction site to their waste treatment sites.

### 2.5. LCA method

We use Pleiades LCA for the building's life cycle assessment. End-point health impacts in LCA can be calculated using Impact World+ (short-term and long-term impacts), or using ReCiPe 2016 (hierarchist impacts). In this study, health impacts are calculated using ReCiPe 2016 (Huijbregts et al. 2017; Johan Lammerant et al. 2019), which is one of the most recent and updated endpoint methods. The database used is ecoinvent v3.4 (ecoinvent 2017).



## 2.6. Optimal ventilation rates

IAQ and LCA results (hierarchical health impacts) are coupled for all ventilation rates. The rate that yields the lowest total impact is identified for each room. We compare the ventilation speeds to reference comfort air speeds  $v_{\text{air}}$  (m<sup>2</sup>/s). The comfort air renewal rate  $Q_{\text{air}}$  (ACH) is given by equation (5-1).

$$Q_{\text{air}} = \frac{v_{\text{air}} \times A_{\text{section}}}{V_{\text{room}}} \quad (5-1)$$

Where  $A_{\text{section}}$  (m<sup>2</sup>) is the section through which air enters the room (e.g. window or ventilation inlets) and  $V_{\text{room}}$  (m<sup>3</sup>) is the room volume. Several reference values of  $v_{\text{air}}$  can be obtained from literature, since they depend on occupant comfort, local climate and outdoor temperatures: they range from 0.5 to 3 m/s (Baldwin and Maynard 1998; Aynsley 2006; Candido 2010; J. Zhou et al. 2023). We select a reference value of 1 m<sup>2</sup>/s above which comfort is reduced in office type spaces since it can disturb loose papers (Aynsley 2006). Based on window sections of 1.7 m<sup>2</sup> (1.1 m x 1.55 m), we obtain  $Q_{\text{air,window}} = 179$  ACH for the kitchen. For mechanical ventilation with vent sections of 0.16 m diameter (Litiu 2012), we obtain  $Q_{\text{air,vent}} = 8$  ACH. We note that these rates are based on air speeds at the window or vent, but that air reaches a bigger room volume and its speed decreases inside.

## 3. Results and discussion

Concentration curves of VOCs (from materials and activities) and PM<sub>2.5</sub> (from activities and outdoors) are presented. Impacts related to other stages of the building's life cycle are obtained from Pleiades LCA. Finally, IAQ and LCA impacts are added up and presented for all scenarios considered, differentiating between the different contributors (ventilation fans, heating, material VOCs, activity VOCs and PM<sub>2.5</sub>). The ventilation rates yielding the lowest impacts are identified. Solutions targeting main sources of impacts are also proposed for the meeting room.

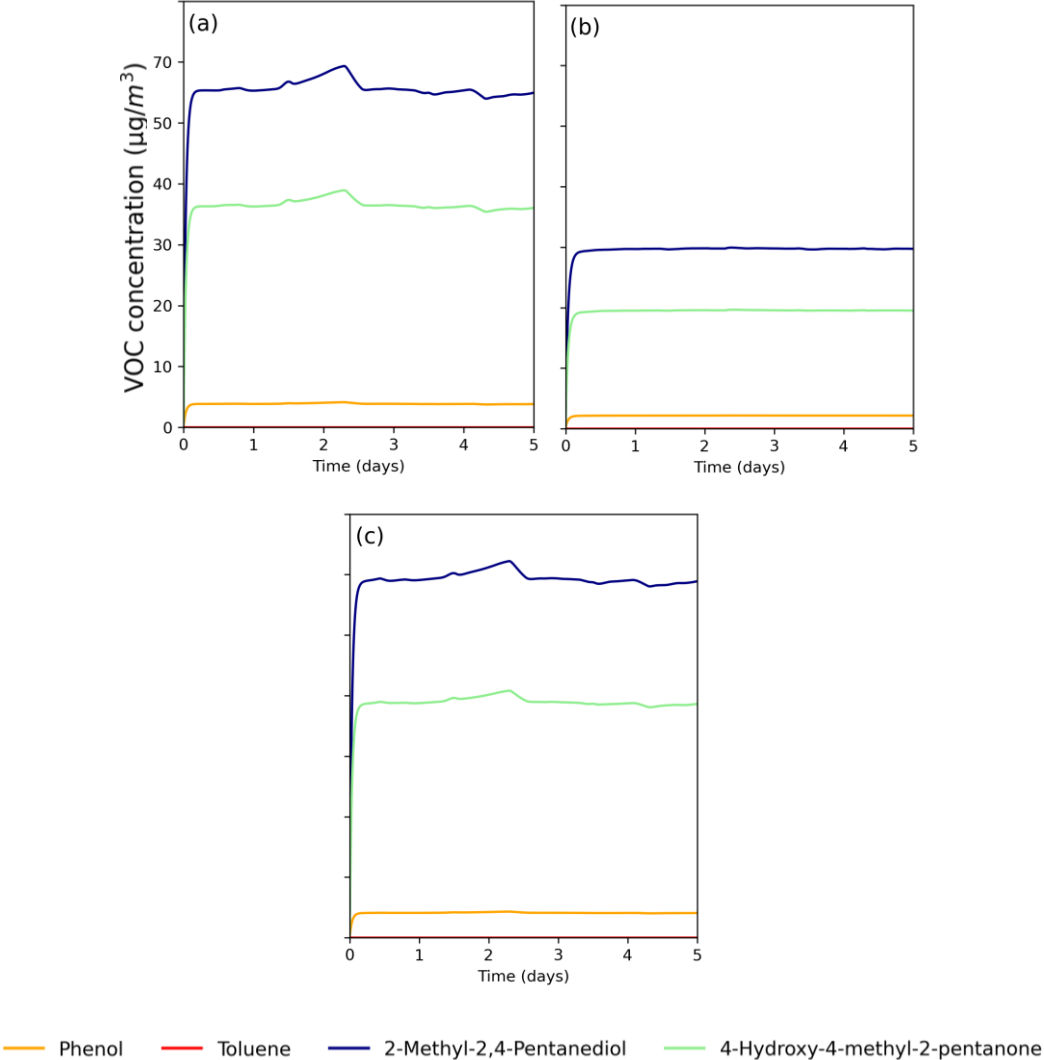
### 3.1. Concentration curves

Concentration curves for VOCs (emitted from materials and activities) and PM<sub>2.5</sub> (emitted from activities and penetrating from outdoors) are presented mainly to observe occupant exposure frequency and duration. For instance, materials emit VOCs continuously, while activities emit VOCs or PM<sub>2.5</sub> at specific points in time. Concentrations correspond to a 0.6 ACH ventilation rate, which is the average rate measured in OECD countries.

#### 3.1.1. VOCs from materials

The concentrations of VOCs emitted by the materials and those formed by indoor chemical reactions are modelled using INCA-Indoor. Out of the 51 emitted VOCs, 4 VOCs having the

highest calculated intake and impacts for each room in this case study, are presented in [Figure 5-4](#). We note that the choice of VOCs presented cannot be generalised, since they are specific to materials considered in this case study, and are likely to change for other materials. Outdoor VOC concentrations are considered to be zero.



*Figure 5-4: Concentrations of four VOCs having the highest intake and impacts in a) the meeting room, b) the kitchen and c) the office considering a 0.6 ACH ventilation rate*

These substances are all emitted by gypsum board. Phenol and toluene are also emitted by the glass wool insulation and the bamboo flooring. The curve is relatively flat, since materials emit constantly during the day. We note the increment in concentrations after about two days, which could be related to variations in ventilation rates due to air infiltration (see [Annex Figure 13](#)) which depend on meteorological conditions (in particular wind speed and direction), or chemical reactions between ozone and VOCs (see chapter 3).

Finally, INCA-Indoor considers indoor VOC concentrations and emissions to be initially at zero. However, in the case of materials, emissions are continuous. There is thus an

underestimation of concentrations <1%, which is negligible, especially in the case of the office building where occupants are not present during that period (from midnight to about 4 a.m.).

### 3.1.2. VOCs from activities

Out of the 77 VOCs considered in this case study (emitted by activities and formed by indoor air chemistry), three with the highest emission rates from floor and counter cleaning are terpinolene, d-limonene and p-cymene (Singer et al. 2006). Their concentration curves are presented in Figure 5-5 for a ventilation rate of 0.6 ACH.

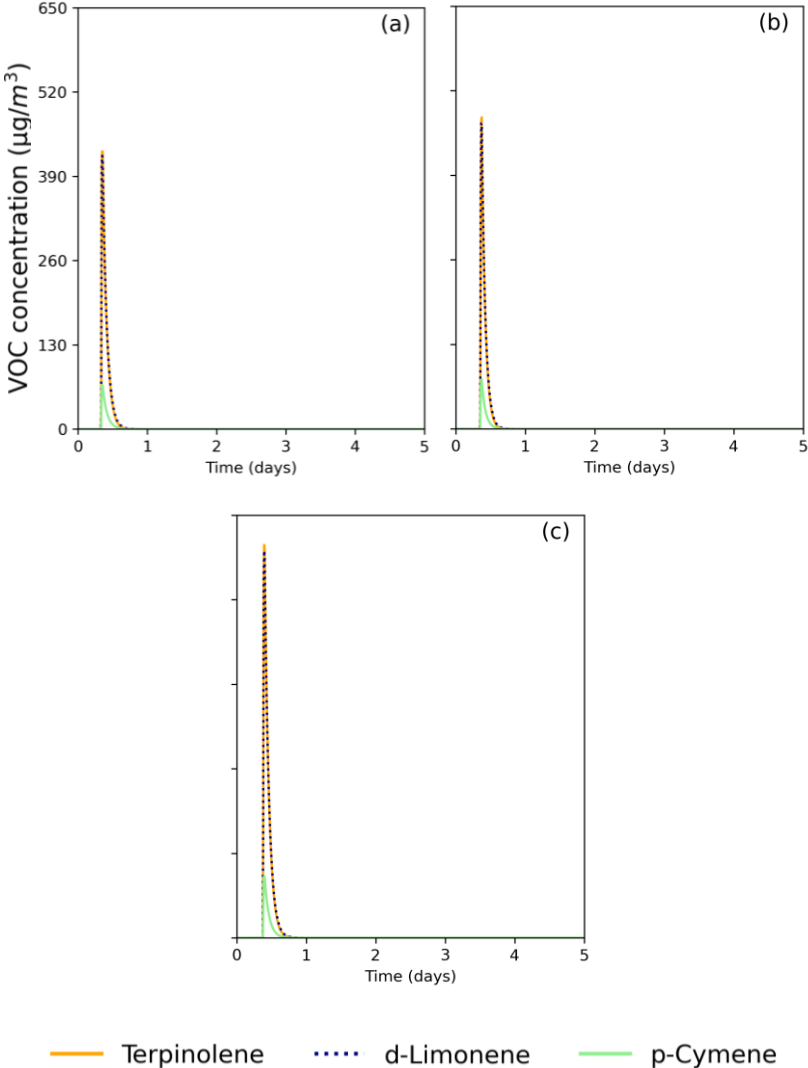


Figure 5-5: Terpinolene, d-Limonene and p-Cymene concentrations from activities in the a) meeting room, b) kitchen and c) office

The VOC concentration peaks are highest in the office, followed by the kitchen and the meeting room. Though same activities lead to their release (cleaning of countertops and floor), the surface areas cleaned and room volumes are different: the office has a cleaning area larger than the meeting room due to the presence of more furniture. The kitchen has a smaller cleaning area, but

also a smaller volume than the other rooms. We note that these VOCs are only present for a day, during which cleaning occurs. There is no exposure on the other days of the week.

### 3.1.3. PM<sub>2.5</sub> from activities

The concentration curves of PM<sub>2.5</sub> over a winter week (thus including electrical fan heating that increases particle concentrations in air through their emission by the friction of moving rotors and through their resuspension due to blown air ) are presented in [Figure 5-6](#) for the three rooms (meeting room, kitchen and office) considering an average mechanical ventilation rate of 0.6 ACH. Outdoor concentrations are represented by blue curves and indoor concentrations (from indoor activities and outdoors) are in orange.

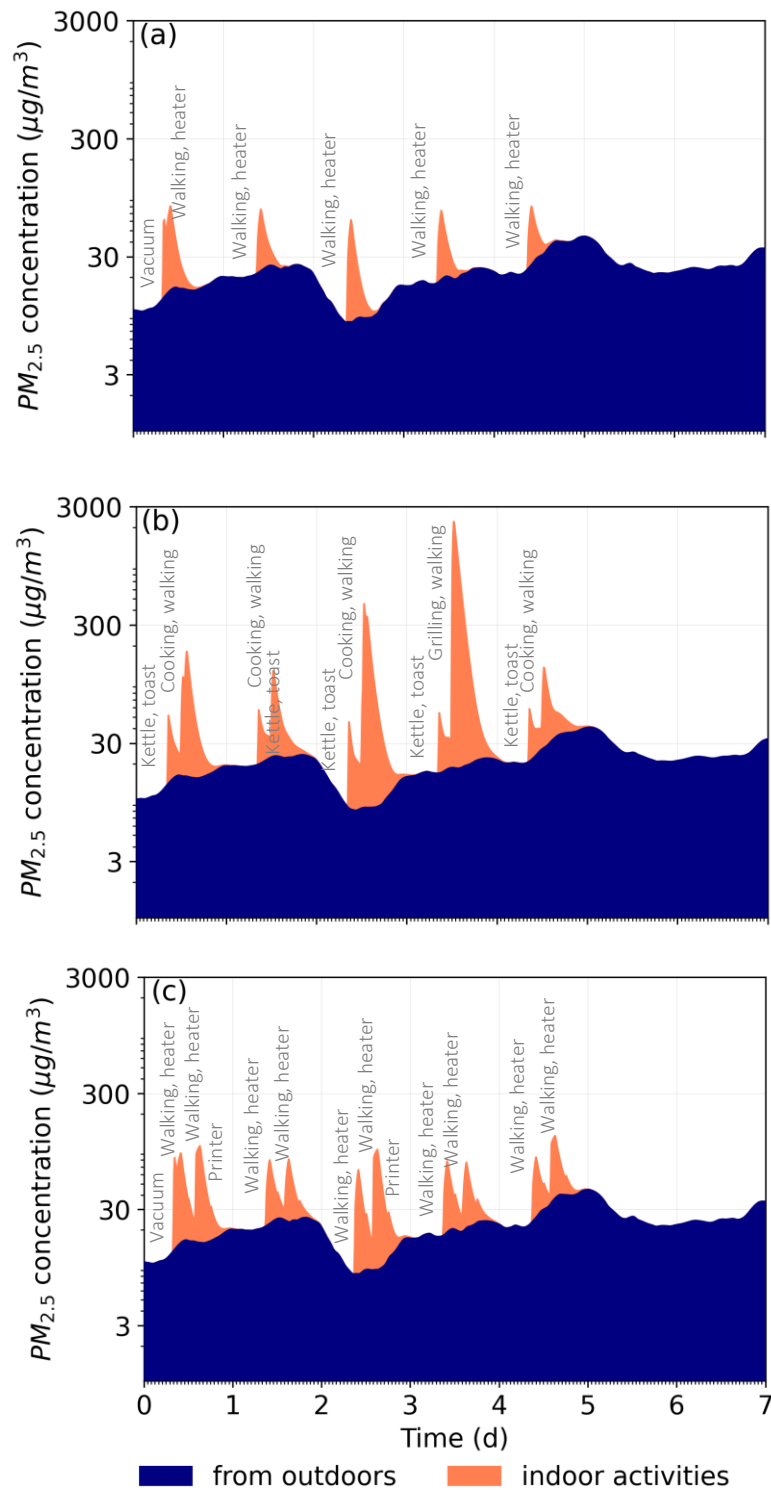


Figure 5-6: : PM<sub>2.5</sub> concentration in the (a) meeting room, (b) kitchen and (c) office over one winter week due to outdoor penetration (navy) and indoor activities (orange) at 0.6 ACH

In the case of the office, concentration increments occur during working days, with peaks around activity periods. The highest peak concentrations over the week in the three rooms are different due to the different activities: 80 µg/m<sup>3</sup> in the meeting room, 2240 µg/m<sup>3</sup> in the kitchen and 130 µg/m<sup>3</sup> in the office. Kitchen PM<sub>2.5</sub> concentrations are very high, especially due to grilling,

as compared to the other rooms where vacuuming and electrical fan heating have the highest emissions. We note that the electrical fan heater considered is portable, and not representative of fixed electrical heaters. In the absence of activities, outdoor concentrations are higher than indoors, and vary throughout the week between  $8 \mu\text{g}/\text{m}^3$  and  $50 \mu\text{g}/\text{m}^3$ , with an average of  $22 \mu\text{g}/\text{m}^3$ . While higher ventilation rates allow to decrease  $\text{PM}_{2.5}$  from indoor sources, there is an increase in penetration from outdoor sources (see Annex Figure 15) if no additional measures are taken (e.g. filter).

### 3.2. Health impacts according to LCA damage indicator

Results of the life cycle assessment are presented in Figure 5-7 for each room and for all life cycle stages (including IAQ impacts), corresponding to a ventilation rate of 0.6 ACH and the use of gas heating.

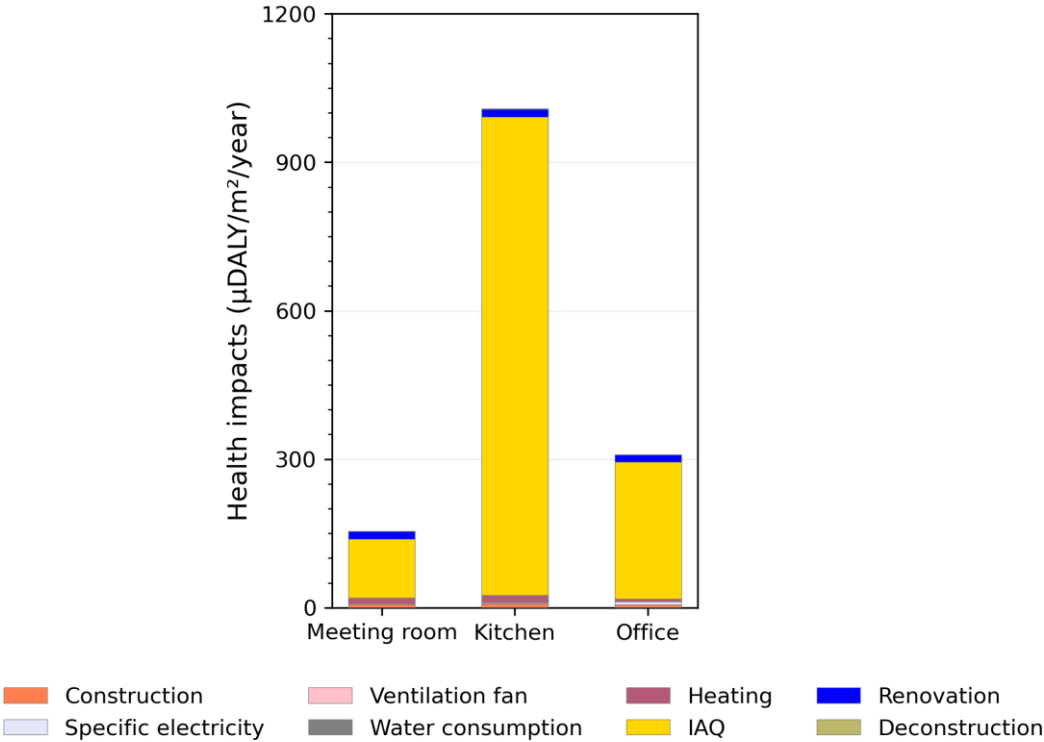


Figure 5-7: Health impacts per unit flooring area for the meeting room, kitchen and office separated into life cycle stages: construction, operation (electricity consumption – specific and from ventilation fans, water, heating and IAQ), renovation and deconstruction – ReCiPe 2016 – Hierarchist

We note from Figure 5-7 that the operation stage, including IAQ, heating and specific electricity use, has the highest impacts on human health, representing 85 to 98% of total impacts in the three rooms for the given ventilation rate, especially due to IAQ, which represents over 77% of the total impacts. However, with higher ventilation rates, IAQ impacts are lowered and heat impacts increased, as seen in the next section. Reference values for total LCA impacts of buildings on

human health are scarce but the few values found in the literature indicate a range of around 80 to 1000  $\mu$ DALY/m<sup>2</sup>/year which fit the order of magnitude found in this study and highlight the significance of IAQ impacts in buildings (Wurtz and Peuportier 2021; Saadé et al. 2022). Results using the Impact World+ method are also presented in [Annex Figure 16](#), indicating that the choice of methods can lead to different LCA results, and eventually different optimal ventilation rates.

### 3.3. Optimal ventilation rates

Results for different ventilation rates ranging from the lowest (closed, airtight building) to the highest (non-OECD countries with cross-ventilation (Rosenbaum et al. 2015)) are presented in [Figure 5-8](#). We only consider IAQ, heating and electricity consumption of ventilation fans, since they are affected by changes in ventilation rates.

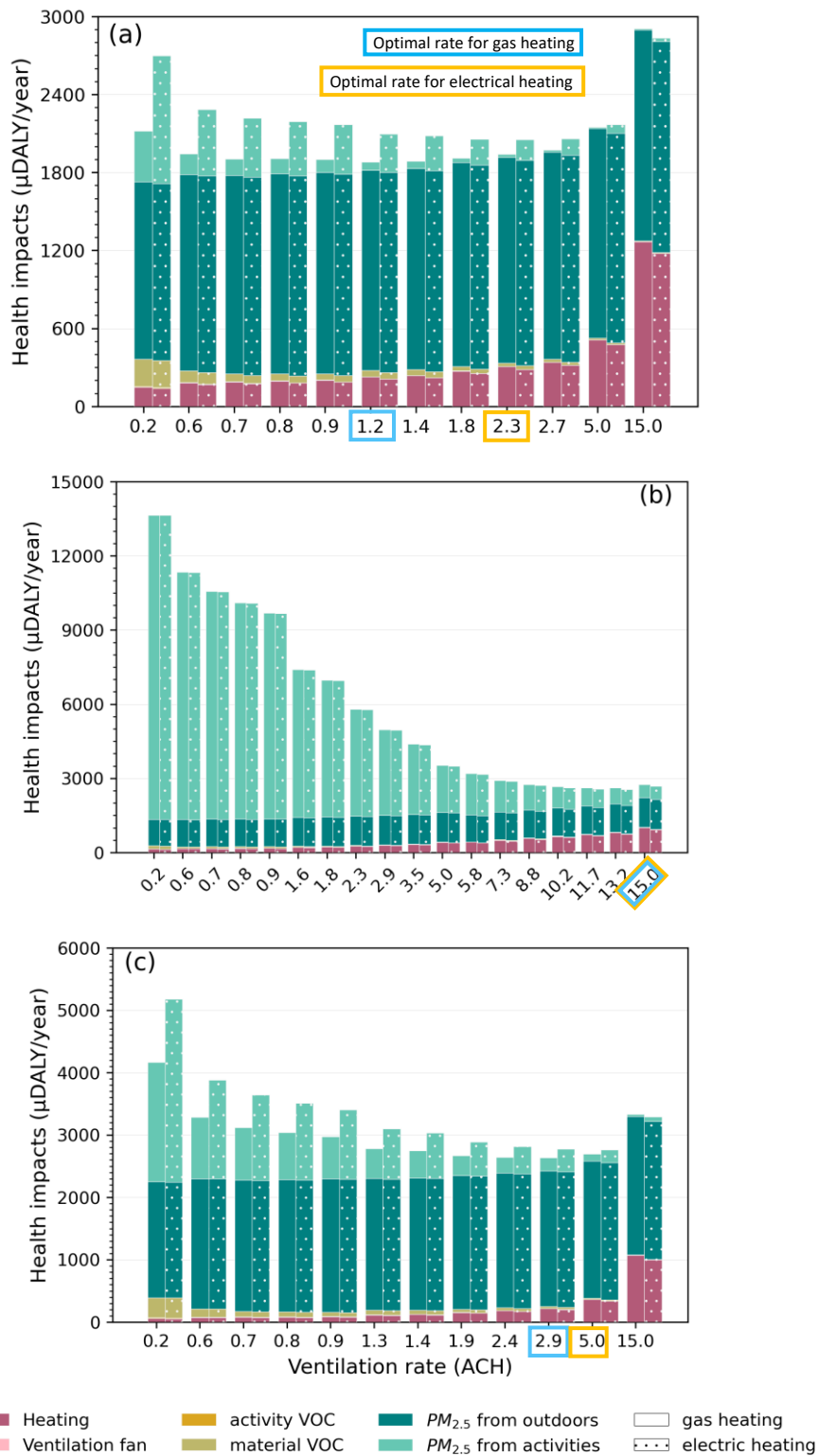


Figure 5-8: Impacts for different ventilation rates, related to heating, ventilation fan, VOCs from activities and materials and  $PM_{2.5}$  from outdoor and indoor sources in the a) meeting room, b) kitchen and c) office



Highest impacts are induced by the inhalation of PM<sub>2.5</sub> (56 to 98% of impacts considered in this section: IAQ, heating and ventilation fans) and heating (gas or electric: 1 to 44%). PM<sub>2.5</sub> impacts are both related to indoor sources and outdoor penetration (see [Figure 5-6](#)).

Impacts of electricity use from ventilation fans (2-9% of total energy consumption) and VOCs emitted by activities or materials are lower: <0.001% for ventilation fans, <0.4% for activities, <10% for materials. In the meeting room and office, PM<sub>2.5</sub> impacts are mainly due to outdoor sources while impacts from indoor sources are lower. In the kitchen, indoor sources are dominant.

We note that the rate inducing the lowest overall impacts in each room is different mainly due to the differences in volume, occupancy (exposure) and activities (emissions). The optimal rate can also be different in the same room but for different heat sources, because 1) LCA impacts of each heat source are different and 2) the electrical fan heater causes the resuspension of particles, leading to additional IAQ impacts. In the meeting room, the optimal ventilation rate is 1.2 ACH for gas heating and 2.3 ACH for electrical heating due to increased IAQ impacts related to the heater. For the same reasons, the optimal ventilation rate for gas heating is also lower than that for electrical heating in the office (2.9 ACH v/s 5 ACH). We note that these results consider only VOCs and PM<sub>2.5</sub>, but not other potentially harmful substances such as CO<sub>2</sub>, which can be above recommended limits at rates below 5 ACH (see [Figure 5-9](#)).

In the kitchen, impacts from PM<sub>2.5</sub> and heating decrease with increasing ventilation to an optimal rate of 13.2 ACH, beyond which they increase again. The rate is much higher as compared to the other two rooms because indoor PM<sub>2.5</sub> emissions are much higher, and because of the larger number of persons present (6 persons in the kitchen, 5 in the meeting room and 3 in the office). The high ventilation rate can be difficult to attain without causing occupants' discomfort due to draught and lowered indoor temperatures, especially in winter. While the optimal ventilation rates remain within  $Q_{\text{air,vent}}$  for the office and meeting room, it is exceeded in the kitchen. The calculated air speed corresponds to the inlet and would be lower inside the room. The actual room air speeds could be evaluated using CFD (computational fluid dynamics).

Impacts presented are related to heat consumption, electricity consumption for ventilation fans, and IAQ. Compared to total impacts (IAQ and all stages of the room's LCA), PM<sub>2.5</sub> represents a share of 57% to 96% in the kitchen, with lowest impacts related to highest ventilation rates. Heating impacts range from 1% to 33%, with lowest impacts related to lowest ventilation rates. For the other two rooms, PM<sub>2.5</sub> impacts range between 54% and 91%, while heat consumption represents 1% to 42% of impacts.

### 3.3.1. Carbon dioxide concentrations

Figure 5-9 shows the maximum CO<sub>2</sub> concentration in the meeting room, with a red line indicating the recommended limit and yellow markers representing the duration for which this limit is exceeded.

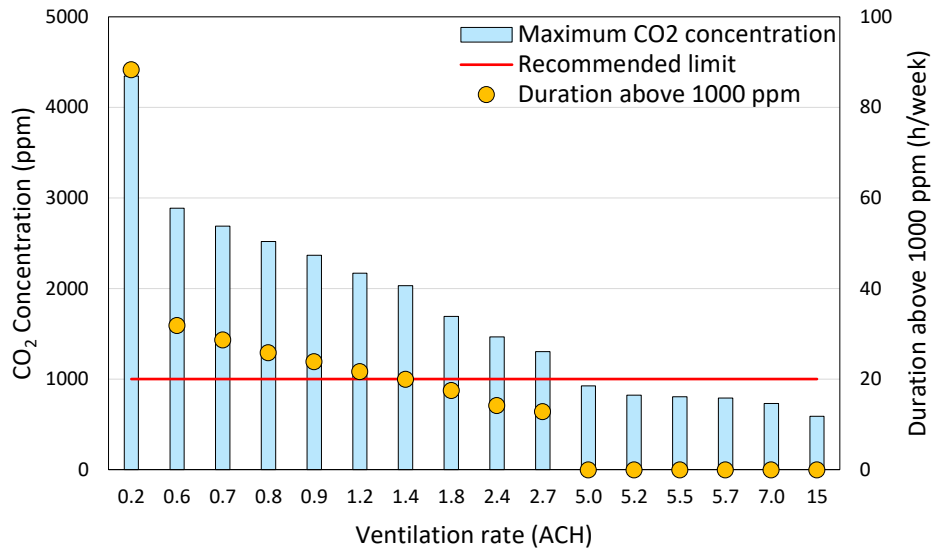


Figure 5-9: Maximum CO<sub>2</sub> concentrations in the meeting room represented by bars, recommended limit represented by a red line and the duration for which concentrations exceed this limit are represented by yellow markers for different ventilation rates

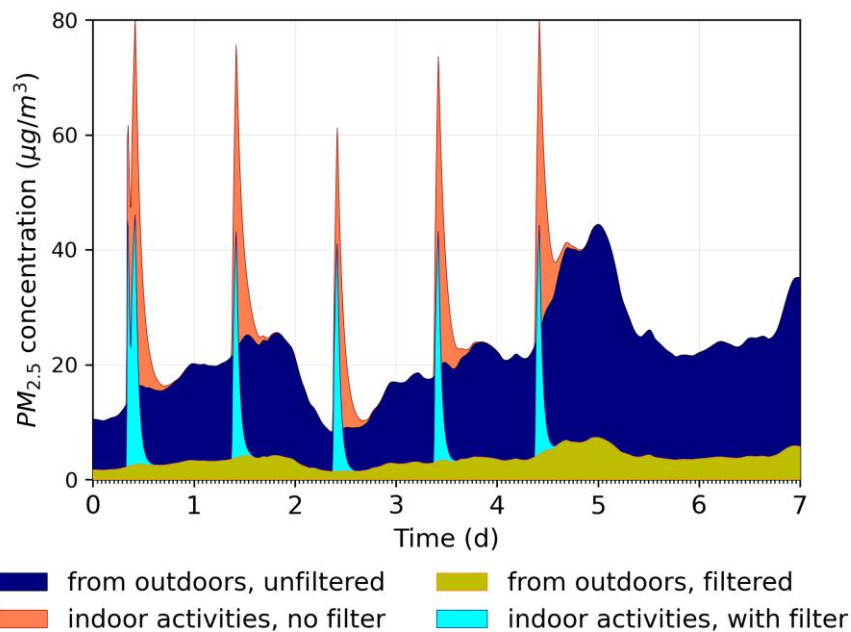
At the calculated optimal ventilation rate of 1.2 ACH, which does not account for CO<sub>2</sub> concentrations, the limit of 1000 ppm is exceeded for 22 h/week. For the recommended flow rate of 18 m<sup>3</sup>/h/person (2 ACH for five occupants in this room), the CO<sub>2</sub> concentration limit is exceeded for an estimated 16 hours per week. This suggests that the calculation of optimal ventilation rates should not only consider VOCs, PM<sub>2.5</sub> and heating needs, but also CO<sub>2</sub> concentrations.

### 3.4. Strategies further limiting health impacts

Besides finding an optimal ventilation rate, other ventilation strategies can be devised to reduce overall impacts. We identified two main sources of impacts linked to IAQ: indoor PM<sub>2.5</sub> (both penetrating from outdoors and emitted by indoor sources) and energy consumption for heating. To address each of them respectively, the effect of filters and a double-flow ventilation system with heat exchanger are tested in the meeting room heated with gas and ventilated at 1.2 ACH (optimal ventilation rate identified).

A filter type F9 is considered (equivalent to MERV 16 (ASHRAE 52.2 2017)), capturing more than 99% of particles with diameters over 1 µm and 45% to 95% of particles with smaller diameters from penetrating air. As illustrated in Figure 5-8, impacts in the meeting room are mainly

due to outdoor penetration. Figure 5-10 shows the outdoor and indoor PM<sub>2.5</sub> concentrations with and without filter in the meeting room.



*Figure 5-10: Indoor PM<sub>2.5</sub> concentrations in the meeting room with 54 m<sup>3</sup>/h ventilation (1.23 ACH) due to outdoor penetration and indoor activities, with and without filter*

Indoor concentrations linked to outdoor penetration decrease considerably with filters, hence reducing long-term exposure (an average of 23.8 µg/m<sup>3</sup> without filter v/s 5.6 µg/m<sup>3</sup> with filter). Short-term exposure due to activities, represented by concentration peaks, are slightly lower with filters due to the different background concentrations: 66.5 µg/m<sup>3</sup> v/s 46.1 µg/m<sup>3</sup>. Thus, the filter leads to a three-fold decrease in intake: 26460 µg<sub>intake</sub>/year with filter and 83890 µg<sub>intake</sub>/year without. The average air concentration is below C<sub>0</sub>, the theoretical minimum risk exposure level (TMREL) being 5.8 µg/m<sup>3</sup>, which is the safe limit below which effects have not been observed (Burnett et al. 2014). In this case, impacts related to PM<sub>2.5</sub> inhalation were considered to be zero. This assumption is based on the absence of evidence from epidemiological studies supporting effects at concentrations below this limit and should be updated if correlations between PM<sub>2.5</sub> exposure and diseases at lower concentrations are recorded. For this first assessment, no additional material where included in the LCA to represent filter manufacture, renewal and end-of-life, nor additional electricity consumption that could be induced by the filter.

With double-flow ventilation, fresh air enters through vents and flows into ducts before entering the room. Air is also extracted from the room, flowing into separate ducts. A heat exchanger allows the preheating of fresh external air entering the room by the warmer air being extracted from the room, resulting in lower need for additional heating: 34% decrease, leading to a

proportional decrease in heating impacts. We consider a double electricity consumption by the ventilation system for the double-flow due to the presence of a second fan.

Figure 5-11 summarises the impacts corresponding to a ventilation rate of 54 m<sup>3</sup>/h (1.2 ACH, identified optimal ventilation rate for the meeting room), considering no filter and single-flow ventilation, no filter and double-flow ventilation, with filter and single-flow ventilation, with filter and double-flow ventilation.

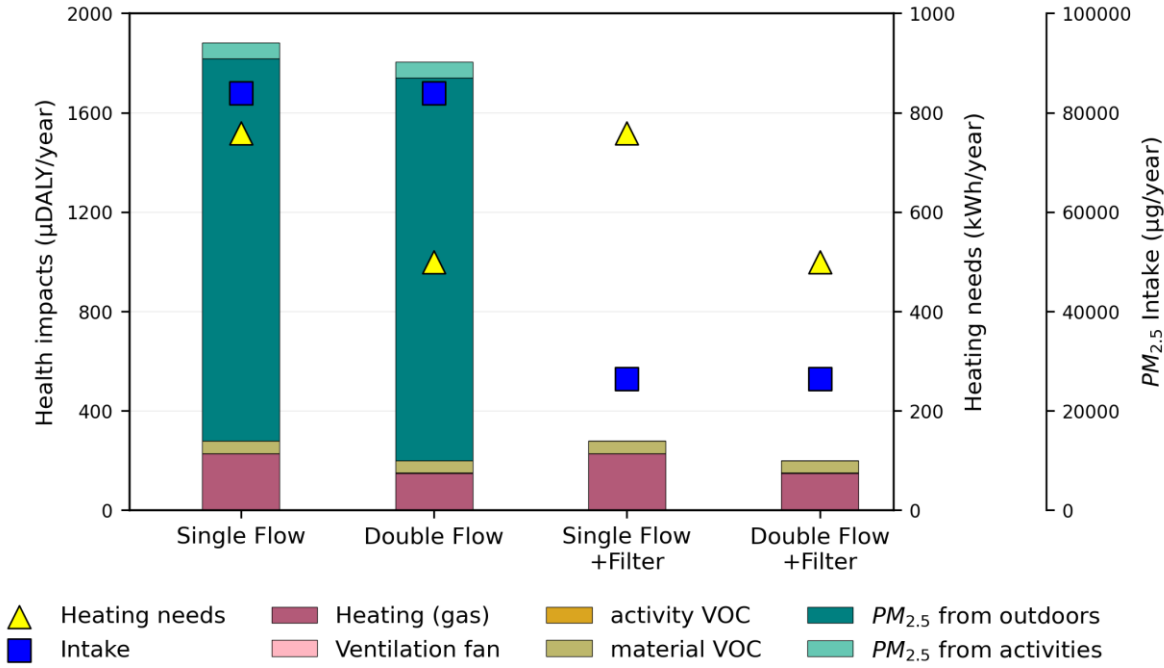


Figure 5-11: Yearly heating needs (kWh), PM<sub>2.5</sub> intake (µg) and health impacts (µDALY) for a single-flow ventilation, double-flow, single flow with filter, and double flow with filter at 1.2 ACH

The combination of double-flow ventilation and filter decreases total IAQ and heating impacts by 89% as compared to a single-flow ventilation without filters. Impacts presented are linked to heat consumption (up to 10% of total LCA + IAQ impacts), electricity consumption for ventilation fans (<0.001% of total impacts) and IAQ (74% of total impacts). Considering the whole life cycle of the building, the double-flow/filter combination leads to 75% decrease in impacts: from 2235 µDALY/year (20 hours<sub>lost</sub>/year) to 504 µDALY/year (5 hours<sub>lost</sub>/year).

Other solutions include, for instance in the kitchen, a hood which can have a PM<sub>2.5</sub> capture efficiency of 0.6 to 1 (Eom et al. 2023). An efficiency of 0.6 results in an optimal ventilation rate of 5.8 ACH, at which impacts of PM<sub>2.5</sub> emitted by cooking indoors are reduced by 60% and total IAQ+LCA impacts by 10% as compared to cooking with no hood at the same ventilation rate. An efficiency of 100% leads to an optimal ventilation rate of 0.6 ACH, with 75% decrease in total

IAQ+LCA impacts. The decrease in impacts between a scenario with and without hood for a 60% efficiency is lower since the optimal ventilation rate is high, hence leading to the partial evacuation of particles even without the use of a hood.

### 3.5. Residential variants

The case study that has been discussed is specific to the given use: office building, occupied only during working days and work hours, with related activities. We choose to explore a variant of the meeting room, used as a residential living room, and that of a residential kitchen with natural ventilation. Since the occupancy scenarios and activities are different, optimal ventilation rates are also expected to change. Both rooms are occupied by 5 persons at a time, and the regulatory air renewal rate of 18 m<sup>3</sup>/h/person yields 2 ACH for the living room and 2.6 ACH for the kitchen. Possible window layouts are studied in the kitchen. In the living room, we assess the health effects of coal heating stoves, with emission rates presented in [Annex Table 31](#). These rates correspond to practices common in rural China or India (J. Zhou et al. 2023), but are not representative of modern wood/pellet heating stoves, such as those certified by Flamme Verte (France) or ClearSkies (England), for which no indoor emission rate is provided. Activity scenarios are presented in [Annex Table 16](#).

#### Ventilation rates: living room

Impacts of the living room are studied for three heating sources: gas, electric fan and a rural heating stove using coal as fuel (Li et al. 2022). Gas heating does not directly affect indoor occupants (in the case of a collective boiler), while the heating stove leads to fugitive PM<sub>2.5</sub> emissions and electric fan heating results in particle resuspension (with the movement of air) and emission (from friction in rotors). Stove heater emissions depend on the heating power, and hence increases for higher ventilation rates where more energy is needed to keep the room at a given set point temperature. We consider average natural ventilation rates from windows, doors and infiltration, which in many countries is the main mode of ventilation in residential settings (Hodas et al. 2016).

#### Window layout: kitchen

A study on different window layouts can help to devise adequate natural ventilation strategies to lower impacts on occupants. As seen in Chapter 4, high PM<sub>2.5</sub> emissions from cooking or grilling, for example, occur in the kitchen, and opening of windows *during* the activity considerably lowered their impacts. The position of windows is also important to consider, since it affects air flows. Two window layouts are defined in [Figure 5-7](#). The second one includes windows opposite each other, leading to cross-ventilation if both are open.

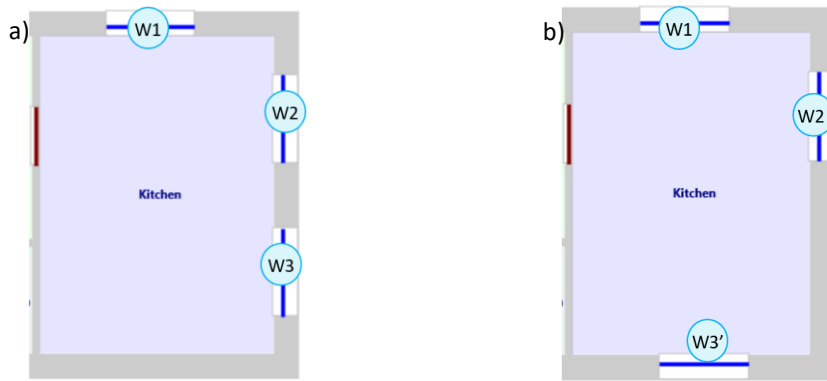


Figure 5-12: Kitchen window layouts a) without cross-ventilation and b) with cross ventilation

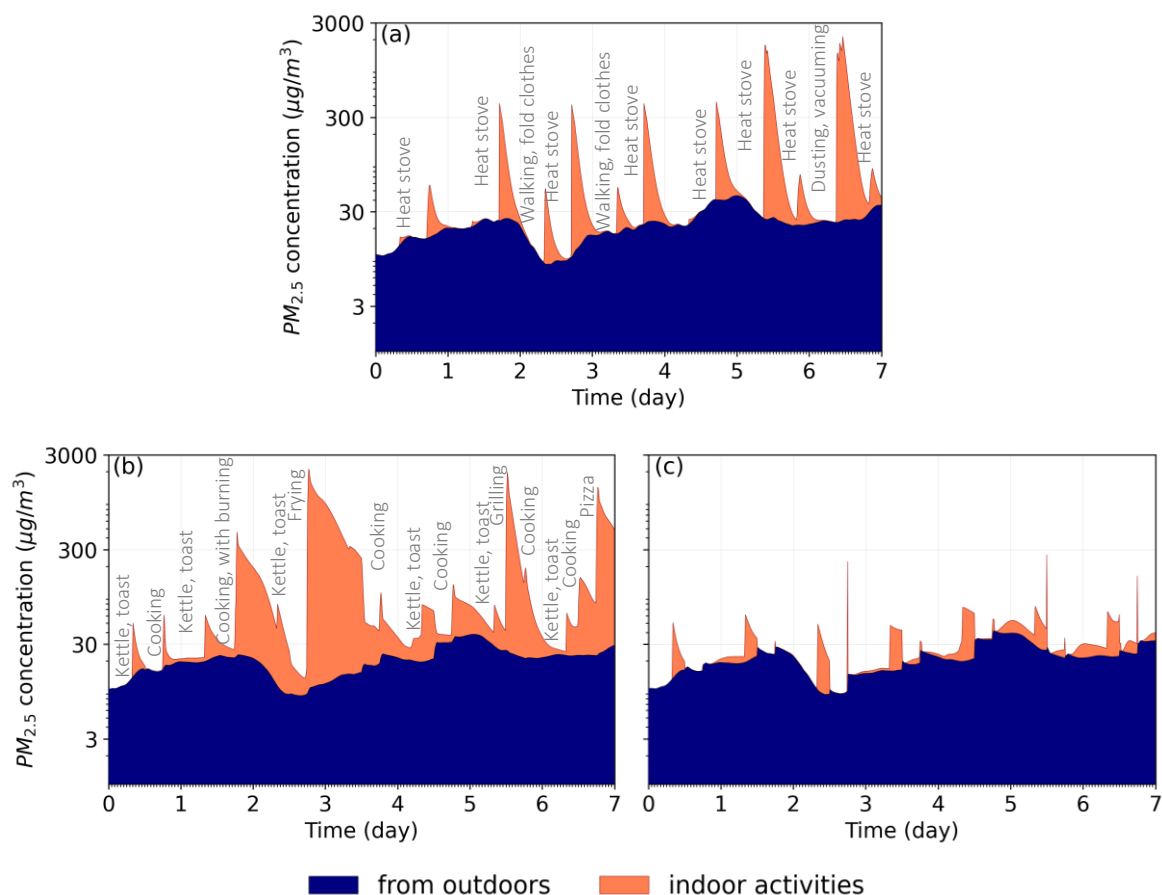
Based on the layouts, opening scenarios are presented in Table 5-4. Windows are open for one hour at 12 p.m., and one hour at 6 p.m., during cooking. The kitchen extractor, according to the building’s occupants, is set at 1 ACH throughout the day.

Table 5-4: Window scenarios in the kitchen: crosses represent absence/closure and ticks represent presence/opening

Window	Scenario 0	Scenario 1	Scenario 2	Scenario 3	Scenario 4	Scenario 5
Extractor	✗	✓	✗	✗	✗	✗
W1	✗	✗	✓	✓	✓	✓
W2	✗	✗	✗	✓	✓	✗
W3	✗	✗	✗	✗	✓	✗
W3'	✗	✗	✗	✗	✗	✓

### 3.5.1. Concentration curves

The concentration curves in Figure 5-13 correspond to a) a ventilation rate of 0.6 ACH in the living room, assuming the use of a heating stove used in rural regions, b) opening of one window for one hour during cooking in the residential kitchen and c) opening of two windows for one hour during cooking with cross-ventilation in the residential kitchen. Occupants are present mainly at night (after 5 p.m.) and during weekends.



*Figure 5-13: Indoor  $PM_{2.5}$  concentrations over a winter week penetrating from outdoors (navy) and emitted by indoor activities (orange) sources a) in a living room with a heating stove and air flow rate of 0.6 ACH, b) in a residential kitchen with one window open and b) in a kitchen with two windows open across from each other for one hour during cooking*

We notice from [Figure 5-13 \(a\)](#) high peaks of up to  $2150 \mu\text{g}/\text{m}^3$  due to wood heating in the living room, especially on weekends where occupants are present for longer periods and consequently use the stove for longer.

In the kitchen, concentrations range from  $10 \mu\text{g}/\text{m}^3$  to  $2130 \mu\text{g}/\text{m}^3$  if only one window is open. We note that exposures to high concentrations can occur for a long period of time (area under the curve in  $\mu\text{g}\cdot\text{s}/\text{m}^3$ ), leading to high intake doses, especially when frying. In the case of cross ventilation (see [Figure 5-13 \(c\)](#)),  $PM_{2.5}$  concentrations are much lower and peaks occur for very short periods as compared to having only one window open. Variations in air change rates for each scenario are presented in [Annex Figure 14](#), explaining the difference in concentrations.

### 3.5.2. Health impacts

LCA impacts of heating with gas, electricity or coal, and indoor impacts related to resuspension from activities (dusting, walking, vacuuming and folding clothes) and emission from electric fan heater and coal heater are presented in [Figure 5-14](#).

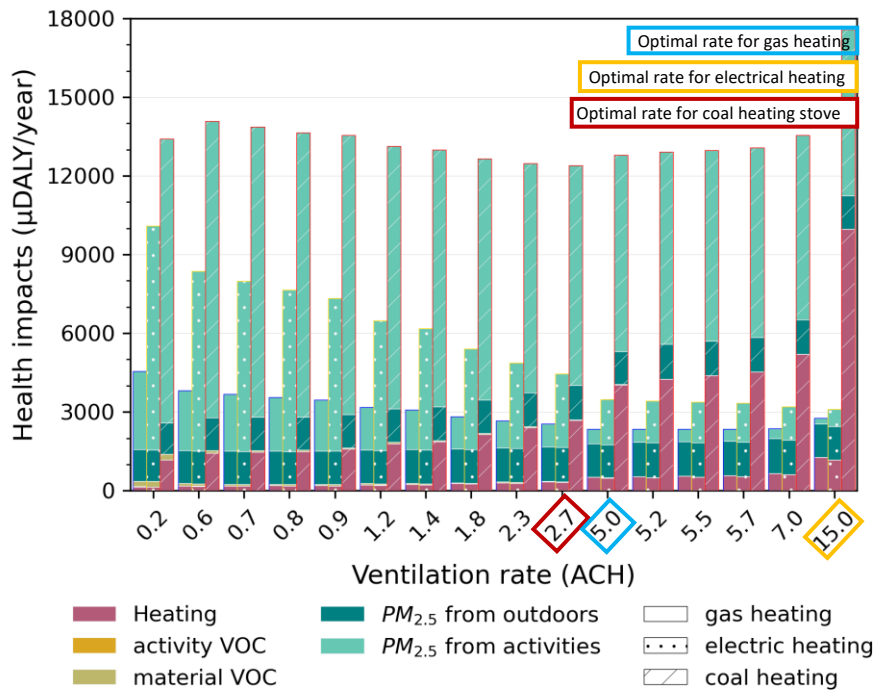


Figure 5-14: Influence of the air change rate on the health impacts of a residential living room for three heat sources: gas, electricity and coal

VOC impacts are negligible, representing <0.2% of all impacts for activities and <5% for materials. Optimal ventilation rates are 5 ACH for gas heating, 15 ACH for portable electrical heating and 2.7 ACH for coal heating. In the case of electrical heating, the duration of heating and indoor PM<sub>2.5</sub> emissions are considered to be equal for all ventilation rates and heating needs (He et al. 2004). We consider a collective gas boiler installed outside the room, which in this case does not affect the room's air quality, unlike a boiler situated inside the occupied zone. For the heating stove situated in the living room, indoor emissions depend on the heating needs. Thus, higher ventilation leads to higher evacuation of PM<sub>2.5</sub>, but also increased emissions from coal combustion. Since the effect factor model used is non-linear, effect factors decrease with increasing average indoor concentrations. The combined effect of both leads to lower health impacts at 0.2 ACH than at 0.6 ACH (see Annex Figure 19).

Electric heating leads to the lowest LCA impacts (i.e. IAQ impacts excluded) for equivalent heating needs: 93% of gas and 12% of coal heating impacts. The total PM<sub>2.5</sub> impacts with stove heating are up to 4 times higher than with electrical fan heating. The contribution of outdoor PM<sub>2.5</sub> has less relative importance in the case of coal stove and electric fan heater (except at 15 ACH for electric fan heater). However, modern heating stoves making use of dry wood or pellets might substantially decrease both LCA and IAQ impacts depending on their location in the building, their efficiency, their design and occupant habits. LCA end-point health impacts of wood are over 7 times lower than coal. According to the Stove Industry Alliance, modern heat stoves could represent



only 2.7% of indoor PM<sub>2.5</sub> emissions, while the coal stove considered in this study represent 48 to 91% of emissions from activities.

Though electrical heating has the lowest LCA impacts, if it contains a fan, indoor IAQ impacts could be non-negligible due to PM<sub>2.5</sub> resuspension and emission. Gas heating, at a ventilation rate of 5 ACH, can yield lower IAQ and heating related impact: 2350 µDALY/year. At 15 ACH (optimal rate for electrical heating) impacts amount to 3110 µDALY/year. For the scenario with a heat stove, impacts are of 12400 µDALY/year for an optimal ventilation rate of 2.7 ACH (IAQ+heating). If the living room is fitted with an electrical heater without fan, there might be different PM<sub>2.5</sub> emissions (or resuspension) from heating and the same LCA impacts as electrical fan heater. Considering no additional PM<sub>2.5</sub> emissions would lead to up to 2% decrease in impacts (heating, ventilation fan and IAQ) as compared to collective gas heating for a ventilation rate of 5 ACH.

The ideal ventilation rates are higher than the recommended value of 18 m<sup>3</sup>/h per person (2 ACH for the living room with 5 occupants). Furthermore, CO<sub>2</sub> concentrations, already above recommended limits for ventilation rates below 5 ACH (see [Figure 5-9](#)), do not include potential fugitive CO<sub>2</sub> emissions from the burning of coal. Optimal ventilation rates do not exceed  $Q_{air,vent}$ , except for electrical heating.

### 3.6. Optimal ventilation strategy and window layout: home kitchen

We test the different window layouts from [Table 5-4](#) for the home kitchen (on the first floor of an apartment in La Madeleine, in the north of France, having an average wind speed of 16.7 km/h). [Figure 5-15](#) illustrates impacts related to heating and indoor PM<sub>2.5</sub>, average air change rates ACH, and average PM<sub>2.5</sub> concentrations (µg/m<sup>3</sup>) for each scenario.

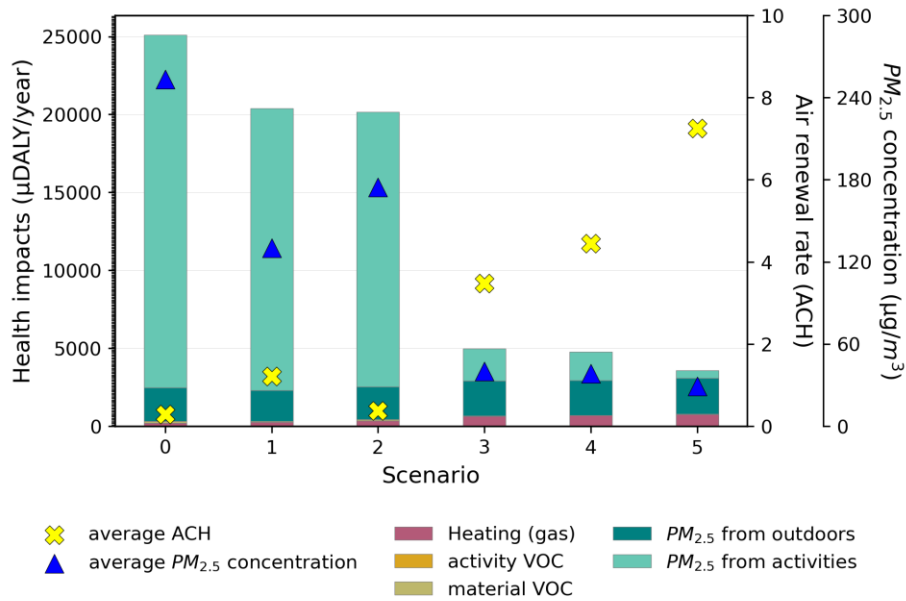


Figure 5-15: Health impacts related to heating and PM<sub>2.5</sub> indicated by bars, average ACH indicated with yellow square markers and average PM<sub>2.5</sub> concentrations (μg/m<sup>3</sup>) indicated with triangle blue markers for a residential kitchen with window opening scenarios 0: infiltration only, 1: extractor, 2: one window open, 3: two perpendicular windows open, 4: three windows open (2 perpendicular) and 5: cross-ventilation

Average and dynamic air renewal rates do not exceed  $Q_{\text{air,window}} = 179$  ACH (see Annex Figure 14). The lowest health impacts (3580 μDALY/year, i.e. 31 hours<sub>lost</sub>/year from heating and IAQ), linked to the highest ventilation rate, is achieved by scenario 5 with cross-ventilation. Scenario 3 and 4 present similar health impacts (4964 μDALY/year and 4762 μDALY/year) due to perpendicular windows.

This study shows that the kitchen should be placed, if possible, in a room either with the presence of cross-ventilation or with perpendicular windows. A kitchen hood should also be installed if possible, since 60% to 100% of particles can be extracted (Eom et al. 2023), leading to 1% - 85% decrease in heating and IAQ impacts altogether. However, the electricity consumption of the hood should then also be included in the study.

#### 4. Discussion

This methodology presents several strengths. First, it is a comprehensive framework evaluating the total impacts of a building, considering a global scale through LCA and a local (indoor) scale through IAQ impact assessment. It can help in decision-making early at the design phase in order to avoid additional health impacts on occupants. Some limitations and linked perspectives have been identified, and the possible applications are discussed in the following sections.

#### 4.1. Limitations and perspectives linked to LCA modelling

We note that optimal ventilation rates are dependent on the calculated impacts, and are potentially not similar for different impact assessment methods. This is because impacts can be different for different methods, and the ventilation rate yielding lowest LCA + IAQ impacts is also likely to change. Further research is needed to investigate those discrepancies. In the case of electrical heating, we considered an average electricity mix according to a prospective attributional LCA approach. However, considering a marginal mix corresponding to consequential LCA (the consequence of adding an electricity demand is an additional electricity production fulfilled by marginal technologies), coal or gas thermal plants would be the main electricity production techniques (in France), thus increasing generated impacts (Roux, Schalbart, and Peuportier 2016). The effect of different LCA methods or electricity mix on the optimal ventilation rate could be assessed in order to evaluate the uncertainties of the results. Regarding the use of filters, their LCA impacts were not considered, and their efficiency relies on their regular maintenance and replacement. Actual filter efficiency can vary if users do not ensure replacement at adequate times.

#### 4.2. Limitations and perspectives linked to IAQ modelling

For rooms with low indoor PM<sub>2.5</sub> emissions, such as the meeting room, outdoor PM<sub>2.5</sub> has the highest contribution to indoor concentrations. Thus, calculated optimal ventilation rates are also low in order to avoid the penetration of particles. However, very high CO<sub>2</sub> concentrations are observed at ventilation rates below 5 ACH in the meeting room, but no health effect factor is available. A consideration of the presence of pollutants whose effects on human health have not yet been quantified would improve the robustness of this methodology. Moreover, several activities such as cooking or lighting of candles can release other pollutants than PM<sub>2.5</sub>, namely VOCs. Their concentrations and effects should be considered when possible. In the case of a heating stove, the only fugitive emission rates were obtained from a study in rural China for coal fuel. Further study on recent heat stove technologies and wood fuel (e.g. Flamme Verte in France, for which no fugitive emission rate was obtained) can be made for more representativeness. As a more general perspective, a wider range of heating devices should be assessed using this integrated framework, including other types of electrical heating and the effect of heat power on emission rates (e.g. radiant heaters, electric oil filled heater, heat pump), wood heating (e.g. pellet or log stove, boiler) or gas heating (condensing and or modulating, possibly inside the occupied zone).

In the case of dust, resuspension depends on dust coverage and type of flooring. For instance, carpets can trap more particles than hard floorings. Furthermore, vacuum cleaners both trap and release particles (decrease in dust coverage and thus future resuspension, for e.g. due to walking, but emission/resuspension while vacuuming). Thus, a thorough study on resuspension values for different building uses and flooring types can be realised, considering dust coverage and the net gain of vacuuming.

A thorough focus on uncertainties of IAQ modelling and resulting impacts can be realised. A main source of uncertainty lies in the activity scenario, as different activities and durations can lead to large differences in pollutant concentrations and resulting impacts. Through a sensitivity analysis, main parameters of influence can be identified in order to simplify the model. Uncertainties related to LCA (fabrication of elements, use and end of life stages) could also be addressed.

Finally, the framework is based upon a total damage on human health obtained by adding the LCA result and the IAQ contribution. The equivalence of LCA and IAQ related DALYs could be discussed, as could the addition of different environmental indicators into a unique LCA damage indicator (for e.g. DALYs corresponding to toxic substances are treated as equivalent to DALYs corresponding to climate change). In this study, we treat the DALY as a single unit and assume that they can be added together.

### 4.3. Applications of the integrated IAQ and LCA framework

We have demonstrated that the methodology enables the identification of optimal mechanical ventilation rates for different case studies according to the building/room's characteristics, location (which determines outdoor pollution and meteorological conditions), and function (which determine indoor activities and occupancy). In addition, optimal natural ventilation strategies can be devised, such as window layouts or the organisation of rooms based on indoor activities. For instance, we concluded that a kitchen should be placed in a room with cross-ventilation or perpendicular windows if possible. Solutions such as double-flow ventilation, filters or kitchen hoods can be assessed using this method.

Additional solutions to reduce the impacts of IAQ can also be explored, since very high ventilation rates are not always realistic. For example, the effect of the frequency of cleaning surfaces (through sweeping, vacuuming or mopping) on PM resuspension rates or cooking habits (e.g. types of oil used, type of food cooked, and presence of lids) on kitchen emissions can be studied to recommend best practices. The increase in ventilation rates are particularly inconvenient when outdoor temperatures are lower or higher than comfort levels (e.g. in winter or during heat waves). Thus, intelligent ventilation systems could be devised, such as increased ventilation rates when outdoor temperatures are within a comfort range, or the adaptation of these rates according to the presence of certain pollutants in indoor air (PM<sub>2.5</sub>, CO<sub>2</sub> or VOCs).

The methodology can also be used to update existing regulations on ventilation in different sectors. For instance, in some cases, optimal ventilation rates exceeded both the French regulatory value (decree of 20 November 1979) and the maximum recommendation of 5 ACH by the International Energy Conservation Code (International Energy Conservation Code (IECC), ICC Digital Codes 2021). Using representative archetypes for different building categories, associated

with reference activity and occupancy scenarios, optimal ventilation rates can be identified. Activity scenarios can be obtained from surveys, or stochastic models (Vorger et al. 2014) and the building stock can be simulated based on national statistics.

## 5. Conclusions

In this chapter, we developed a methodology allowing to link IAQ and building LCA. We saw that impacts of IAQ are very important at the building's scale, especially at low ventilation rates, and that they should not be neglected in building ecodesign. Two main pollutant categories were studied: VOCs and PM<sub>2.5</sub>, emitted from materials or indoor activities. We showed the applicability of this framework to decision-making in building design, construction or planning through a case study. Material and activity VOC emissions were responsible for lower health damages than energy use for heating and PM<sub>2.5</sub>. Particles are emitted by indoor activities (especially cooking and heating stove), penetrate from outdoors or resuspend from surfaces. For different variants of the case study, optimal ventilation rates or window layouts yielding lowest total impacts (of heating and PM<sub>2.5</sub>) were identified. We noted that optimal ventilation rates were highest for rooms with high PM<sub>2.5</sub> emissions, namely, the kitchen: 13.2 ACH (without hood above the cooking stove). Natural ventilation through perpendicular or opposite windows can also result in considerable decrease of impacts if open during kitchen activities. The use of a coal stove, still used in Asian rural areas, had high impacts on occupants due to fugitive emissions. Additional solutions were proposed to reduce impacts from increased ventilation rates: double-flow ventilation with heat exchanger and the use of filters, led to a decrease by 56% of overall impacts (LCA+IAQ).

Main perspectives of this study include detailed sensitivity analysis, first on impact assessment methodology, including the LCA hypotheses (e.g. marginal vs attributional electricity mix), and second on the IAQ impact assessment developed. A wider range of heating devices should also be included since they influence both LCA and IAQ impacts. Finally, building archetypes can be developed in order to facilitate the integration of the framework into building ecodesign tools.



# Conclusion

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Le résumé des travaux réalisés dans les différents chapitres est d'abord présenté. Le premier chapitre présente l'état de l'art sur le sujet de la qualité de l'air (QAI), la quantification de ses impacts et quelques travaux qui ont visé à faire le lien entre la QAI et l'analyse de cycle de vie (ACV). Le deuxième chapitre traite les émissions de composés organiques volatils (COV) des matériaux et leurs impacts sur la santé. Dans le troisième chapitre, nous étudions les impacts des COV émis par des activités intérieures et les réactions chimiques entre les substances présentes dans l'air. Les activités sont aussi étudiées dans le chapitre 4, en ce qui concerne l'émission de particules fines ( $PM_{2.5}$ ). Les impacts de celles-ci sur la santé des occupants sont évalués pour plusieurs scénarios de ventilation. Finalement, dans le chapitre 5, le lien entre les différents modèles permet de calculer l'impact total de la QAI sur la santé des occupants, qui est couplé aux impacts évalués par l'ACV du bâtiment. La méthode développée est testée sur des cas d'études afin d'identifier des débits optimaux de ventilation permettant de réduire la somme des impacts liés au chauffage et à la QAI.

La méthode développée pour intégrer la QAI à l'ACV des bâtiments peut être utilisée comme outil d'aide à la décision dans le secteur du bâtiment, pour intégrer les impacts de la QAI dans les résultats d'ACV des bâtiments et des projets urbains, ou même développer des recommandations ou règlements dans le secteur de la santé. Cette méthode peut être introduite dans la plateforme logicielle Pleiades, qui regroupe les modules suivants : 1) modélisation thermique dynamique, 2) ACV et 3) QAI. Ce dernier ne considère actuellement pas les émissions de polluants par les activités des occupants, et ne permet pas de calculer les impacts sur leur santé. L'intégration de notre méthode à ce module permettrait d'étudier tous les impacts sur une unique plateforme.

Les perspectives sont ensuite détaillées. Tout d'abord, les concentrations simulées pourraient être validées par des mesures expérimentales sur un cas d'étude. Les modèles utilisés qui décrivent le parcours du polluant devraient être développés pour améliorer leur fiabilité et évaluer leurs incertitudes. Par exemple, la resuspension des  $PM_{2.5}$  par les activités des occupants en fonction du taux de poussière déposé sur les surfaces pourrait être étudiée, étant donnée l'influence de ce phénomène sur les concentrations dans l'air. Les perspectives comprennent de plus la définition d'archétypes basés sur des statistiques nationales ou globales, ainsi qu'une étude poussée sur les solutions visant à réduire les impacts de la QAI.

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## Summary

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In this section, the different findings in each chapter are summarised. The applicability of the general framework developed in this thesis is discussed: it can be used as a decision-making tool in the building construction sector, update current LCA practice in order to consider IAQ impacts of materials and processes, or help to devise regulations in the health sector. Possible perspectives of this study are outlined. These include the further development of individual models that describe the pollutant pathway in order to increase their robustness and evaluate their uncertainties, a thorough study on solutions that can reduce IAQ impacts and the definition of archetypes (activity/emission scenarios) based on national or global statistics that can be included directly to building LCA.

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## 1. Conclusions

The aim of this PhD thesis is to integrate indoor air quality (IAQ) in the life cycle assessment (LCA) of buildings. This objective was reached through the development of a framework that considers different indoor pollutant sources (materials, occupants and activities) as well as outdoor penetration of VOCs and PM<sub>2.5</sub> in the calculation of IAQ health impacts on building occupants.

Through a literature review presented in Chapter 1, we identified the need for models that treat the entire pollutant pathway, from their emission to their impacts on human health, considering two main pollutant categories: volatile organic compounds (VOCs) and fine particulate matter (PM<sub>2.5</sub>). VOCs, emitted namely by materials and occupant activities (e.g. use of detergents or air fresheners), are taken in through inhalation, ingestion, dermal contact or gaseous dermal uptake, and can be highly hazardous to health – even carcinogenic. Primary PM<sub>2.5</sub> come from indoor activities (e.g. cooking, candle burning or smoking) or penetrate from outdoors. When inhaled, they can lead to serious cardiopulmonary diseases. Secondary VOCs and PM<sub>2.5</sub> can also be formed by indoor chemical reactions between precursors, namely ozone and organic compounds. The integration of IAQ in building LCA requires the calculation of their impacts. We chose the Disability-Adjusted Life Years (DALYs) unit, an indicator used in LCA methods to characterise endpoint damages on human health.

The framework presented in Chapter 2 was developed to: 1) combine a material VOC emission model with measured data in order to fix model parameters that have substantial uncertainties of a factor 10 to 100, leading to predicted results within a factor 1.1 of measured concentrations, 2) extrapolate chamber air concentration measurements at different time steps, 3) predict indoor air concentrations in terms of building characteristics and 4) calculate long-term impacts on occupant health. The framework was applied on two monolayered materials: gypsum board and bamboo flooring, and tested on a case study. Long-term health impacts related to the VOC off-gassing in a 15 m<sup>2</sup> office with bamboo flooring and gypsum-covered walls, occupied by three persons during working hours, was of 70 μDALY, or 0.15 μDALY/year per m<sup>2</sup> of flooring area. As a comparison, LCA impacts (material extraction and end-of-life) related to gypsum walls are of 0.46 μDALY/year per m<sup>2</sup> of flooring area. The sensitivity of health impacts to different parameters was also tested: ventilation rate, material thickness, temperature and material age. Long-term impacts were mostly affected by ventilation rates and material thickness, while other parameters mainly affected short-term impacts. The applicability of the framework to multilayered materials was demonstrated through an exploratory step, but information on layer thicknesses is required to obtain conclusive results.

In Chapter 3, a framework was proposed to evaluate health impacts related to VOCs from indoor activities on occupant health. Emission data collected from literature were coupled with the INCA-Indoor model in order to calculate indoor air concentrations of released VOCs, based on the room volume and air renewal rate. The framework was tested on the same case study as in Chapter 2, where,

once a week, the floor and counter were cleaned with a reference detergent and air freshener was used. We also included direct VOC emissions from occupants (skin, breath and clothes), but their contribution to indoor pollutants was negligible, except for CO<sub>2</sub> from exhaled air. Impacts amount to 0.16  $\mu$ DALY/day for activities but are potentially underestimated since only 51 substances out of 79 were treated by INCA-Indoor. We also studied the effect of indoor air chemistry on the concentration of VOCs and consequent health impacts, which increased by 31% in summer due to the formation of toxic chemicals such as formaldehyde. The formation of secondary organic aerosols (SOA) led to an increase by up to 2.9  $\mu$ g/m<sup>3</sup> in PM concentrations. Our framework, based on INCA-Indoor, thus not only allows to consider direct VOC emissions, but also the indirect effects of ozone on the formation and consumption of substances in chemical reactions.

Fine particulate matter was studied in Chapter 4. A framework linking activities (and their PM<sub>2.5</sub> emission rates) to indoor air concentrations, intake by inhalation and health impacts was developed. INCA-Indoor was used to model indoor air concentrations resulting from 19 one-hour activities over a duration of 24 hours and different ventilation scenarios. Concentration results were used to validate a parametric model for an average scenario. Effect factors ( $\mu$ DALY/ $\mu$ g<sub>intake</sub>), characterisation factors ( $\mu$ DALY/ $\mu$ g<sub>emitted</sub>) and impact scores for a functional unit of one-hour activity ( $\mu$ DALY/h<sub>activity</sub>) were derived for each activity and scenario, ranging from  $8.2 \times 10^{-5}$   $\mu$ DALY/h<sub>activity</sub> for low emission rates (e.g. from candle lighting) with high ventilation during the activity, to  $4.84 \times 10^2$   $\mu$ DALY/h<sub>activity</sub> for very high emission rates (e.g. from a wood stove) in closed, airtight buildings. We note that increased ventilation rates reduce IAQ impacts, but could lead to a higher energy demand for heating, and consequently higher impacts.

In Chapter 5, we combined models of the previous chapters that treat the pollutants pathway from emission to health impacts, considering PM<sub>2.5</sub> and VOC emissions from materials and occupant activities. Results were integrated to building LCA and the applicability of the general framework was tested on case studies to identify optimal ventilation rates that allow to reduce overall health impacts (IAQ + LCA). Optimal ventilation rates range from 1.2 ACH (low indoor emissions) to 13.2 ACH (high emissions, such as in a kitchen). The effect of different types of heater (gas, electric fan and coal heating stove) was also assessed in a residential variant. The heating stove was responsible for highest indoor impacts due to fugitive PM<sub>2.5</sub> emissions, as well as the electric fan heater due to resuspension and emission of PM<sub>2.5</sub>. A combination of two solutions was proposed: the use of filtered, double-flow ventilation with heat exchanger, allowing to reduce impacts by 75%.

More generally, we conclude that impacts from heating and PM<sub>2.5</sub> were highest, while those of VOCs were negligible at the scale of the building's life cycle. Outdoor PM<sub>2.5</sub> penetrating from unfiltered ventilation had a significant contribution to health impacts of occupants, especially for high ventilation rates in rooms with low indoor emissions, representing 9% to 100% of total PM<sub>2.5</sub> impacts. In rooms with high indoor PM<sub>2.5</sub> emissions, for instance in a kitchen, outdoor PM<sub>2.5</sub> has a lower influence on IAQ,

and high ventilation rates are recommended. The results of this thesis highlight the high relative importance of IAQ health impacts at the scale of a building's life cycle, and the fact that it should not be disregarded in construction projects or in the public health sector. These impacts reflect the high share of global diseases and deaths attributed to air pollution according to different studies such as the Global Burden of Diseases (Murray et al. 2020; Tran, Park, and Lee 2020; Ali et al. 2021). Our framework, aiming to achieve a more complete building LCA by considering global and local/indoor impacts to avoid the displacement of pollution (in this case from outdoors to indoors), has a number of possible applications.

## 2. Applicability

The general framework can be applied to building and urban projects as an aid for decision-making, thus helping to prevent impacts right at the design phase, as currently conducted by LCA practitioners. Architects and building constructors can associate the two types of tools described (air flow/pollutant concentration tool such as INCA-Indoor and building simulation/LCA tool such as Pleiades LCA) which can be integrated in a single user interface. Results can be provided together as elements of the *operation* stage. One possibility would be to integrate this thesis's works into INDALO, which is the user interface of INCA-Indoor, already available as a module of Pleiades, integrating outdoor PM<sub>2.5</sub> and indoor material VOCs and CO<sub>2</sub> emissions. Further development of the interface could lead to full LCA (integrating IAQ) results, namely by allowing users to indicate sufficient information on occupant activities, and coupling resulting pollutant concentrations to exposure and effect factors. Thus, users could study design alternatives in order to opt for materials or types of heating, calculate adequate ventilation rates and define optimal window layouts that lead to lowest health impacts.

Regulations can also be devised at a national level for public health, by identifying optimal ventilation rates in different building categories represented by building-activity archetypes (e.g. 1.4 vol/h for a typical office and 5.8 vol/h for a shared kitchen) and using calculated health impacts further encourage the use of double-flow ventilation systems with filters. Current regulations, which date from 1979 in France (decree of 20 November 1979), could thus be updated by considering effects of energy consumption and IAQ. These representative archetypes can also include outdoor PM<sub>2.5</sub> levels, which vary according to geographical location, leading to different health impacts. Thus, for different geographical locations (e.g. urban or rural), optimal ventilation rates may be different. The spatialisation of PM<sub>2.5</sub> impacts is also linked to local mortality rates, hence underlying information on the vulnerability of populations is needed: PM<sub>2.5</sub> can be more or less toxic (depending on the major emission sources of the region), populations can be at higher or lower risk due to other factors (e.g. smoking, alcohol drinking and diet), or healthcare can be more or less accessible.

The influential parameters identified in the case of material VOC emissions (temperature, thickness, age, ventilation rate) could be used to devise strategies to lower their influence and

uncertainty, for instance through artificial ageing of the material with heat by material manufacturers. These strategies could also be added to the regulatory labelling scheme in France (decree of 19 april, 2011). The quantified benefit of the reuse of construction or furniture materials, could be further enhanced with the addition of IAQ impacts because emissions is lower from old materials, thus encouraging a circular economy.

Frameworks developed in each chapter can be applied to LCIA. For instance, IAQ health impacts for a broader range of materials and products (cleaning detergents, air fresheners, candles, incense sticks) can be calculated and integrated to their LCA inventory (as an approximation of actual impacts for given use scenarios, but dynamics are important for more precise results) according to defined indoor settings. These could help in decision making by opting for solutions that have the least impacts linked to their whole life cycle (including their extraction/transformation, effects on energy demand of the building and indoor pollution). Impact scores and characterisation factors of different PM<sub>2.5</sub>-emitting activities can be integrated to LCIA for different activity scenarios.

### 3. Perspectives

**Identification of influential parameters.** The framework developed is based on a large number of parameters and different models treating each step of the pollutant pathway for different sources and types of pollutants. The complexity of these models can be questioned. First, through a sensitivity analysis, parameters of highest influence can be identified. Their underlying models can thus be further enhanced (for e.g. following the perspectives described below), while remaining parameters can be simplified.

**Uncertainties.** For different steps of the framework, uncertainties can be calculated whenever possible. Main parameters of each model have to be identified in order to calculate the uncertainty of the final results (i.e. health impacts). Different ways to calculate these uncertainties are given under the specific model categories in the next paragraphs.

**Material VOC emissions.** A material VOC emission model is used to calculate emission rates, and the uncertainty linked to these rates are evaluated using measured data, but for only a limited number of materials. Better estimation of the uncertainty can be obtained with more measured data if they are available. Using the error indicators (calculated at the validation step of the calibration methodology in Chapter 2), an uncertainty can be calculated on the indoor concentrations, and ultimately on health impacts. If more measured data becomes available for the same material category, the dispersion of results can be used to derive uncertainties.

We also demonstrated the applicability of the framework developed in Chapter 2 to multilayered materials, but the data available was insufficient. The framework should be tested on a multilayered material with complete information regarding the compositions and thicknesses of different layers.

Material VOC emissions are dynamic, since they depend on a number of factors, including the mass of substance remaining in the material. In the final case study in Chapter 5, we consider a unique emission rate based on an average material age of one year. We saw that it was a good estimation for most substances emitted by flooring in the case study, except for one of them: isopentane, which was emitted from the insulation in lower layers. Dynamic emission rates can be introduced to INCA-Indoor for higher precision in indoor VOC concentrations.

**Activity VOC emissions.** We use emission data measured in literature for activity VOCs. However, according to the amount of detergent applied, temperature and ventilation rates, emission rates (linked to the evaporation of liquid substances) are expected to change (Wei et al. 2022). Particular attention to activity VOC emission models would increase the robustness of the framework. In this chapter, we also consider chemical reactions between substances (e.g. VOCs and ozone). In order to evaluate uncertainties linked to these reactions, multiple simulations can be run using the same parameters and the dispersion of results can be quantified.

In the case of PM<sub>2.5</sub>-emitting activities studied in Chapter 4 (e.g. due to indoor combustion such as cooking or heating stoves and the burning of candles), the consideration of other pollutants generated, including VOCs, CO or NO<sub>x</sub> are recommended to obtain more precise impact evaluation.

**Particle resuspension and emission.** A set of impact scores and characterisation factors was derived for different indoor activities under different ventilation scenarios. The variability of these results gives an indication of uncertainties, which can be added to the final framework. These variations are mainly linked to user behaviour: opening of windows before/during/after activity, type of cooking (for e.g. electric/gas, frying/grilling...).

The current INCA-Indoor model does not consider resuspension, and its introduction could increase the reliability of simulated concentrations. For this, further study on the parameters affecting the resuspension of particles by activities (for instance by walking) are required. More specifically, the link between dust coverage of surfaces, air change rates and activity intensity can be investigated.

In the case of particle emissions from vacuum cleaners, emission rates used in this study were measured in 2004 by He et al., and since then, newer technologies have been developed, which could lead to a difference in rates. If new data is available, they should be updated. Vacuum cleaners are used to trap deposited dust, and subsequently lower the resuspension of particles from occupant activities in the hours after vacuuming. The model could be adapted to take this factor into account: vacuum cleaners act both as a “sink” that evacuates particles and a “source” that emits or resuspends particles into air.

**Quantification of health impacts.** The general framework allows to account for two pollutant categories: VOCs and PM<sub>2.5</sub>. Indirect effects of ozone, through the formation and consumption of VOCs, were considered. However, direct effects of ozone and other substances (e.g. CO<sub>2</sub>, radon or NO<sub>x</sub>) were not considered since their impacts are not yet quantified. Further study on the quantification of their

potential health hazards are to be encouraged, due to growing evidence of their long-term health effects. Furthermore, toxicity data are still lacking for a number of VOCs, and the model should be updated accordingly when available.

Impacts of PM<sub>2.5</sub> were calculated using the GBD-IER model coupled with mortality data. Thus, calculated DALYs represent the risk of premature death. The number of years lost living with a disability should also be included for each of the five disease outcomes identified. The model relies on outdoor ambient PM<sub>2.5</sub> levels to calculate the attributable risk fraction (ARF, which determines the mortality attributed to fine particulate matter), while exposure occurs both outdoors and indoors. Representative indoor archetypes can be used (for instance for specific countries) and according to the time spent indoors/outdoors, the annual average exposure concentration can be estimated. Finally, the model does not account for the composition and toxicity of particles. Further study on the presence of different types of particulate matter (e.g. metals, plastic, elemental carbon, water droplets...) could help increase the reliability of calculated effect factors.

Children and adults are exposed differently. Young children have a higher probability of being in dermal contact and ingesting toxic substances, and their metabolism is less mature, being potentially more at risk than adults for a given dose. A specific focus on children exposure and health impacts in indoor environments can be made, based on previous studies such as Aurisano et al. (2021) and Nguyen et al. (2019). Studies support the fact that biological differences, including sex, race or age, could influence the exposure, intake of or reaction to certain pollutants (Braun 2015). These parameters could also be included in the model. Finally, the uncertainties calculated for different steps of the pathway (e.g. linked to material VOC emission measurements, indoor air chemistry or occupant behaviour) can be included to the study. This can, for instance, result in a range of ventilation rates possible for a given case study.

**User v/s occupant.** Indoor occupants are exposed to diluted concentrations of substances in air. However, users who carry out activities can be exposed to very high concentrations. For instance, people who clean or cook are potentially exposed to hazardous amounts of toxic substances, especially if they conduct the activity regularly. Our model accounts for occupant exposure, but not user exposure. In order to account for both, two-compartment models could be used, representing the smaller volume of air near the user, and the rest of the room, with the diffusion of substances between the two compartments. The smaller volume close to the person is called the breathing zone. This concept is also used to study the concentration of particles emitted by individuals through the shedding of skin or dust into their “personal cloud”. Though their contribution is negligible in the room, the concentrations within this zone are considerable (Licina, Tian, and Nazaroff 2017).

**Indoor and outdoor spatial differentiation.** In the literature review, we saw that spatial variations could affect exposure to air pollutants. For heat-generating activities, for instance, emitted pollutants

tend to rise with convection. This could help to identify solutions that reduce impacts. In the case of cooking, the effect of kitchen hoods can be assessed by considering the vertical movement of particles released. Optimal window or room layouts can also be defined, as a function of pollution sources present indoors, outdoors or in intermediary areas (e.g. car or gas boiler in a garage), to encourage evacuation or avoid indoor penetration (Brown et al. 2015 Szczepanik and Scislo 2021). This could also help to integrate individual gas boilers for space and water heating (unlike a collective gas boiler considered in chapter 5) which release harmful substances such as CO<sub>2</sub>, NO<sub>x</sub> and VOCs that can enter occupied spaces.

**Archetypes.** In order to define general recommendations in ventilation strategies for different types of building/room use, representative archetypes can be defined. They should be based on local or global statistics, such as 1) the building stock, e.g. tertiary/residential, apartments/detached houses, number of occupants, floor area, 2) activity schedules obtained from surveys and 3) outdoor pollution (rural/urban setting). Such data can be obtained, in the French context, from INSEE, the National Institute of Statistics and Economic Studies, which carries surveys on the budget and activity schedules of households.

**Validation of simulated concentrations.** For a given case study, air concentration measurements of different pollutants and ventilation rates can be recorded over a given period. The case study can be modelled using INCA-Indoor, replicating occupancy, activity scenarios and average emission rates. Thus, simulated concentrations can be compared to measurements, allowing the model to be validated experimentally, and uncertainties of the model can be estimated.

Outdoor air pollution data (concentrations of pollutants including PM<sub>2.5</sub>, SO<sub>2</sub>, NO<sub>x</sub>, O<sub>3</sub>) can be obtained with high resolution. For instance, in the Parisian area, Airparif predicts air pollution with a resolution of 10 m to 50 m. This type of resolution can be added to our general framework in order to define locations with lowest health risks, and include this information to define the building orientation (e.g. facing courtyard or garden).

**Reducing IAQ impacts.** A thorough study on possible solutions to reduce health impacts of IAQ can be realised using the developed framework in order to develop recommendations. For instance, the effect of filters that treat VOCs, SO<sub>2</sub>, NO<sub>x</sub>, CO<sub>2</sub>, besides PM, can be studied. The role of revegetation on outdoor and indoor air quality could also be investigated, using existing studies as a starting point (Yibo Yao et al. 2022; Grylls and van Reeuwijk 2022; Z. Li et al. 2023).

#### 4. Fundamental questions

We conclude this thesis with a fundamental question: to which extent can we combine IAQ DALYs to LCA DALYs? On a technical note, further improvement of this framework and other LCA methods could ensure better coherence between underlying models.



On a social note, can we consider 1 DALY<sub>LCA</sub> as equivalent to 1 DALY<sub>IAQ</sub>? LCA impacts could affect large populations, while IAQ impacts affect a small number of occupants. In this thesis, we choose to treat all DALYs equally, and as one unit. However, we should still bear in mind that LCA impacts can often be on a more global scale, generating health impacts on populations (especially workers) of a given country or region, for instance where raw materials are extracted, for the construction of a building in another location, in which occupants bear IAQ impacts. Through the regionalisation of LCA, models can give more information on these issues, which remain a complex environmental, health and social dilemma.



### A1.1. Multilayered emission model

The mass balance equations for each node are given in equations (A-1), (A-2), (A-3) and (A-4) for an element  $i$  (e.g. wall or floor) under study (Micolier 2019):

$$V_{\text{room}} \frac{dC_{\text{in,g}}(t)}{dt} = \sum_{j=1}^n S_j h_{m,j}(t) \left( \frac{C_{s_i,j}(t)}{K_{\text{ma},j}(t)} - C_{\text{in,g}}(t) \right) \quad (\text{A-1})$$

$$- Q(C_{\text{in,g}}(t) - C_{\text{out}}(t)) - XF_{\text{inh}} V_{\text{room}} C_{\text{in}}(t)$$

$$- XF_{\text{derm,gas}} V_{\text{room}} C_{\text{in}}(t)$$

$$S_i dL_i \frac{dC_{s_i}(x,t)}{dt} = \frac{2D_{m_i}(t)S_i}{dL_i} (C_{1_i}(t) - C_{s_i}(t)) - S_i h_{m_i}(t) \left( \frac{C_{s_i}(t)}{K_{\text{ma},i}(t)} \right. \quad (\text{A-2})$$

$$\left. - C_{\text{in}}(t) \right) - XF_{\text{ing,dust}} V_{\text{room}} C_{s_i}(t)$$

$$- XF_{\text{derm,cont}} V_{\text{room}} C_{s_i}(t)$$

$$S_i dL_i \frac{dC_i(x,t)}{dt} = \frac{2D_{m_i}(t)S_i}{dL_i + dL_{i-1}} (C_{i-1}(t) - C_i(t)) \quad (\text{A-3})$$

$$- \frac{2S_i}{\frac{dL_i}{D_{m,i}(t)} + \frac{K_{\text{ma},i}(t)}{K_{\text{ma},i-1}(t)} \times \frac{dL_{i-1}}{D_{m,i-1}(t)}} \times \left( C_i(t) - \frac{K_{\text{ma},i}(t)}{K_{\text{ma},i-1}(t)} C_{i-1}(t) \right)$$

$$S_i dL_i \frac{dC_i(x,t)}{dt} = \frac{2D_{m_i}(t)S_i}{dL_i + dL_{i-1}} (C_{i-1}(t) - C_i(t)) \quad (\text{A-4})$$

$$- \frac{2D_{m_i}(t)S_i}{dL_i + dL_{i+1}} (C_i(t) - C_{i+1}(t))$$

$V_{\text{room}}$  is the volume of air inside the room,  $C_{\text{in}}$  the concentration of the pollutant indoors,  $S$  represents the surface area of the material,  $h_m$  the indoor convective mass transfer coefficient,  $K_{\text{ma}}$  the material-air partition coefficient expressed in equation (2-2),  $D_{m_i}$  the diffusion coefficient through the material, expressed in equation (2-1),  $C_{s_i}$  the concentration at the surface,  $Q$  ( $\text{m}^3/\text{s}$ ) the air flow rate,  $C_{\text{out}}$  the outdoor concentration of the substance (usually considered to be equal to zero for VOCs) and  $L_i$  the layer of element  $i$  of surface  $S_i$  and thickness  $dL_i$  in contact with the boundary layer of air.  $XF_{\text{inh}}$ ,  $XF_{\text{derm,gas}}$ ,  $XF_{\text{ing,dust}}$ ,  $XF_{\text{derm,cont}}$  ( $\text{kg}_{\text{intake}}/\text{kg}_{\text{compartment}}$ ) are the exposure factors through inhalation,

gaseous dermal uptake, ingestion and dermal contact as expressed in equations (1-2), (1-3), (1-4) and (1-5). Index  $j$  represents the surfaces of the compartment (e.g. 1 to 6 for the surfaces of a closed room). At equilibrium,  $\frac{C_{si}(t)}{K_{ma_i}(t)}$  is equal to  $C_{b,n}$ , the concentration of the pollutant at the boundary layer.

The material layers are divided unevenly: the thickness of the layers increase further from air since mass and heat transfers are more important in layers closer to air (Guo 2013). The thickness of layer  $L_i$  is given by (A-5) and (A-6) as a function of the total thickness of the material  $L_{mat}$  and the ratio between the layer thicknesses  $q$  (Guo 2013):

$$dL_i = \begin{cases} \Delta x q^{i-1} & \text{for } i \in [1, \frac{L+1}{2}] \\ \Delta x q^{L-i} & \text{for } i \in [\frac{L+2}{2}, L] \end{cases} \quad (A-5)$$

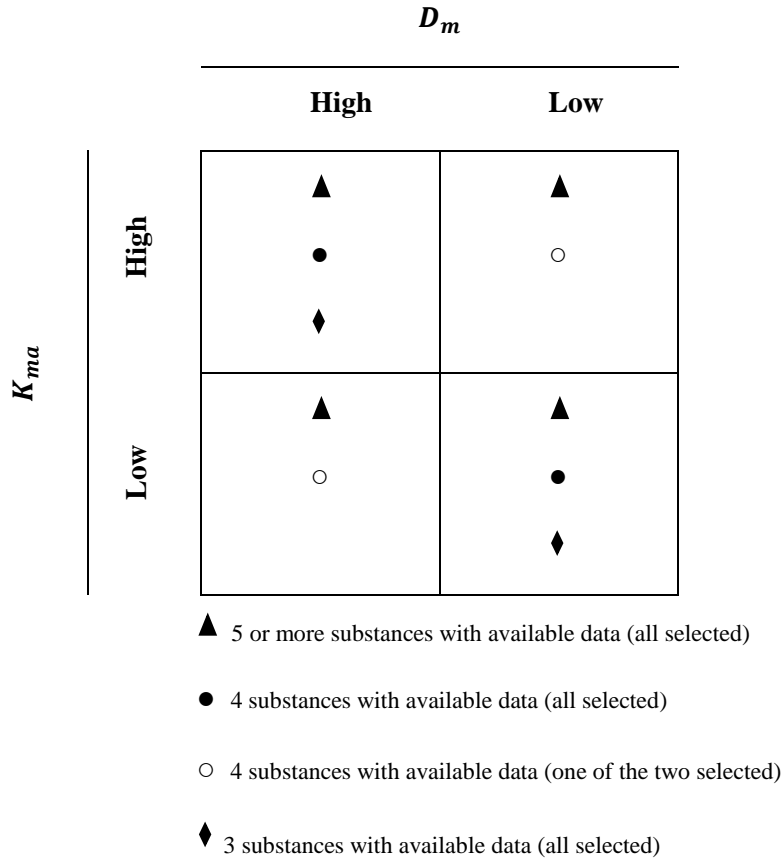
$$\Delta x = \frac{L_{mat}(1-q)}{2(1-q^{\frac{L}{2}})} \quad (A-6)$$

## A1.2. Criteria for selection of substances for main parameter calibration

The cut-off criteria defined by Huang et al. (2020), which defines whether the substance is D-limited or K-limited, is used in the selection of test substances based on their emission dynamics.

$$D_m^{0.61} \cdot K_{ma} < 0.4 \quad (A-7)$$

Substances having different emission dynamics are selected. The criteria, based on diffusion and material-air partition coefficients and the number of substances with available concentration data are summarised in Annex Figure 1.



*Annex Figure 1: Selection criteria for substances based on their diffusion and material-air partition coefficients as a function of the number of available data*

Different symbols represent cases where data are available for different numbers of substances: 1) triangle for 5 or more, 2) dot for 4 and 3) diamond for 3 substances and their position in the grid represents the criteria for selection based on the  $D_m - K_{ma}$  combination. In all cases, substances with high  $D_m -$  high  $K_{ma}$  and low  $D_m -$  low  $K_{ma}$  combinations are selected. In case 2), a third substance is selected: the one with the highest absolute difference between its  $D_m$  and  $K_{ma}$ , which can either be the high  $D_m -$  low  $K_{ma}$  or low  $D_m -$  high  $K_{ma}$  combination, represented with unfilled dots in the table. In case 1), all possible combinations are selected.

### A1.3. Health impact calculation

The effect factor,  $EF$  ( $cases/kg_{intake}$ ) for each disease category are given by the expressions below (Fantke, Bijster, et al. 2017):

$$EF_{rep/dev} = 0.1/ED10_{rep/dev} \quad (A-8)$$

$$EF_{general\ non-cancer} = 0.1/ED10_{general\ non\ cancer} \quad (A-9)$$

$$EF_{cancer} = 0.5/ED50_{cancer} \quad (A-10)$$

$ED10$  and  $ED50$  (kg/lifetime) represent the lifetime doses per person that causes a reproductive/developmental (rep/dev) or general non-cancer and cancer disease probability of 10% and 50% after intake. The  $ED10_{rep/dev}$  and  $ED10_{general\ non-cancer}$  data is obtained from the ToxVal database ('CompTox Chemicals Dashboard' 2018) and  $ED50_{cancer}$  from USEtox. They are separated into  $ED_{rep/dev}$  for reproductive/developmental effects damage and  $ED_{gen,non-cancer}$  for general non-cancer damage.

The endpoint result is based on the sum of all cancer and non-cancer effect factors. One reproductive/developmental case is equivalent to  $SF_{rep/dev} = 40$  DALYs/non cancer case, one general, non-cancer case is equivalent to  $SF_{gen,non-cancer} = 2.4$  DALYs and one cancer case is equivalent to  $SF_{cancer} = 11.5$  DALYs. The total health impacts  $HI_{tot}$  related to all  $N$  substances  $s$  emitted by materials on building occupants are given by equation (A-11).

$$HI_{tot} = \sum_{s=1}^N HI_{rep/dev,s} + HI_{gen,non-cancer,s} + HI_{cancer,s} \quad (A-11)$$

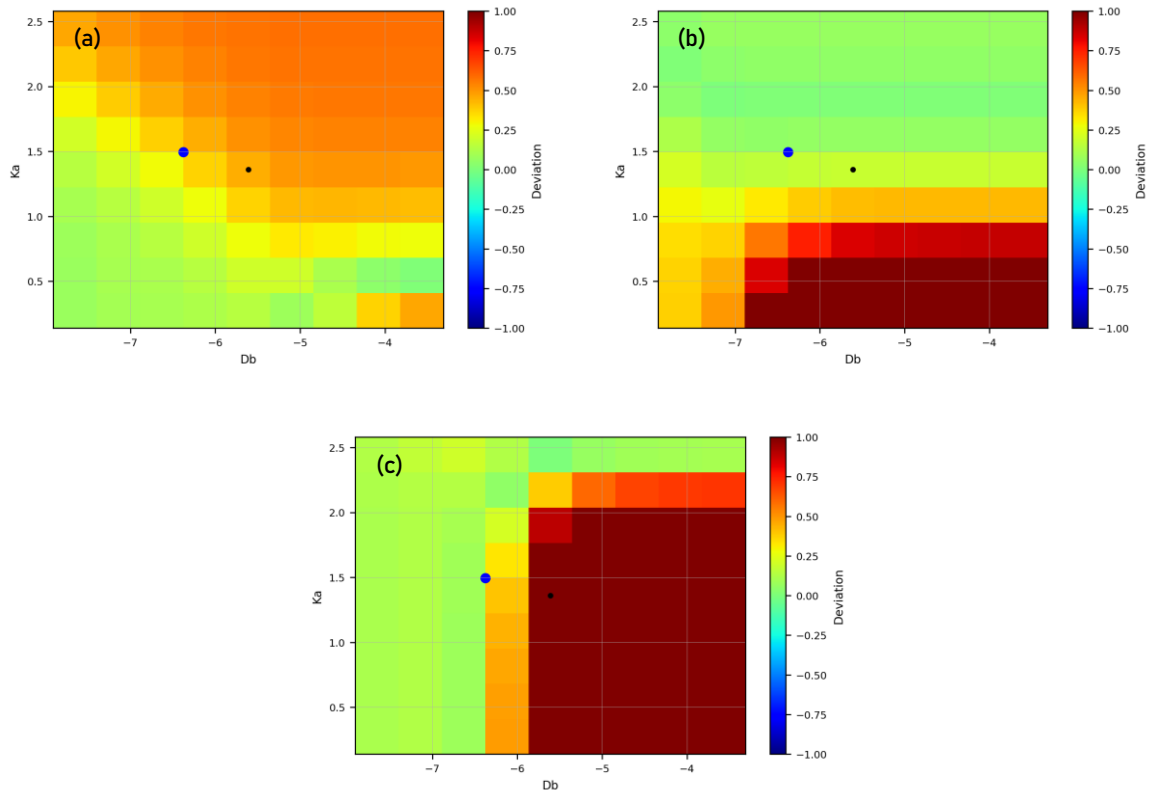
The effect factors (DALY/kg<sub>intake</sub>) for substances emitted by gypsum board and bamboo flooring are summarised in Annex Table 1.

*Annex Table 1: Effect factors for substances emitted by gypsum board and bamboo flooring*

<b>CAS</b>	<b>EF ingestion (DALY/kg<sub>ingested</sub>)</b>	<b>EF inhalation (DALY/kg<sub>inhaled</sub>)</b>
66-25-1	-	-
79-09-4	-	-
108-88-3	7.38E-03	1.62E+01
50-00-0	4.65E-02	4.02E+00
108-95-2	8.88E-03	1.27E+00
106-46-7	6.50E-02	8.77E-01
127-18-4	3.38E-01	7.74E-01
1330-20-7	6.16E-03	6.67E-01
64-19-7	-	1.42E-01
75-07-0	1.90E-01	1.36E-01
111-76-2	5.23E-02	1.23E-01
100-42-5	1.23E-03	1.04E-01
123-38-6	-	7.57E-02
104-76-7	1.40E-02	5.44E-02
95-63-6	3.03E-03	3.12E-02
629-62-9	-	2.79E-02
544-76-3	-	2.79E-02
100-41-4	1.01E-02	2.39E-02
629-59-4	-	2.02E-02
100-52-7	2.05E-02	1.94E-02
71-36-3	5.49E-03	1.24E-02
103-09-3	-	5.35E-03
67-63-0	-	3.73E-04

#### **A1.4. Variation of air change rates: unvarnished plywood**

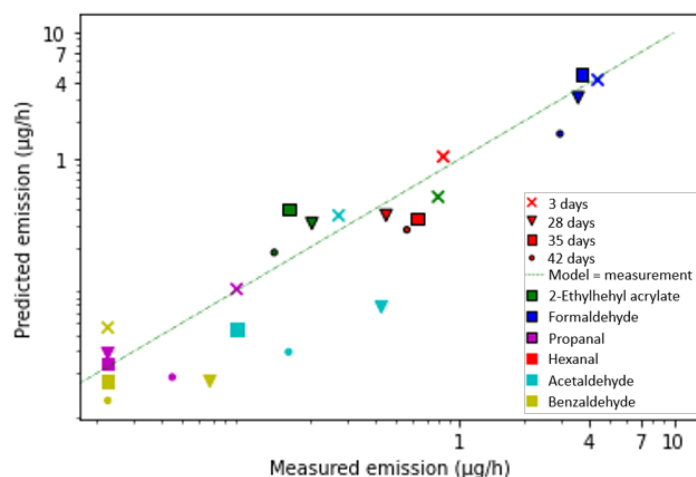
The material is placed in a 0.509 m<sup>3</sup> test chamber with glass walls, 22.4 °C air temperature and 47.7 % relative humidity. The substances selected to determine the optimal parameters are 2-ethylhexyl acrylate, formaldehyde and propanal. The heat maps for all the selected substances are presented in Annex Figure 2. The deviation is calculated according to the first and second measurements only (3 days and 28 days).



*Annex Figure 2: Heat maps of the deviation of predicted concentrations from observed values for (a) 2-ethylhexyl acrylate, (b) formaldehyde, (c) propanal with the reference values indicated by a black dot at the centre and the optimal values in a larger blue dot*

Model predictions with calculated parameters ( $b$ ,  $\beta$  and  $MF_0$ ) are compared to measurements in Annex Figure 3.





*Annex Figure 3: Measured emission v/s predicted emission values for the six substances given off by the plywood board for four given time periods: 3 days, 28 days, 35 days and 42 days*

The model agrees with the measurements with MALD and RMSLE uncertainty factors of 10.3 and 4.6 for all substances and 9.0 and 4.5 for only test substances. These factors are quite high, especially compared to factors within the range 1-2 for monolayered materials tested previously.

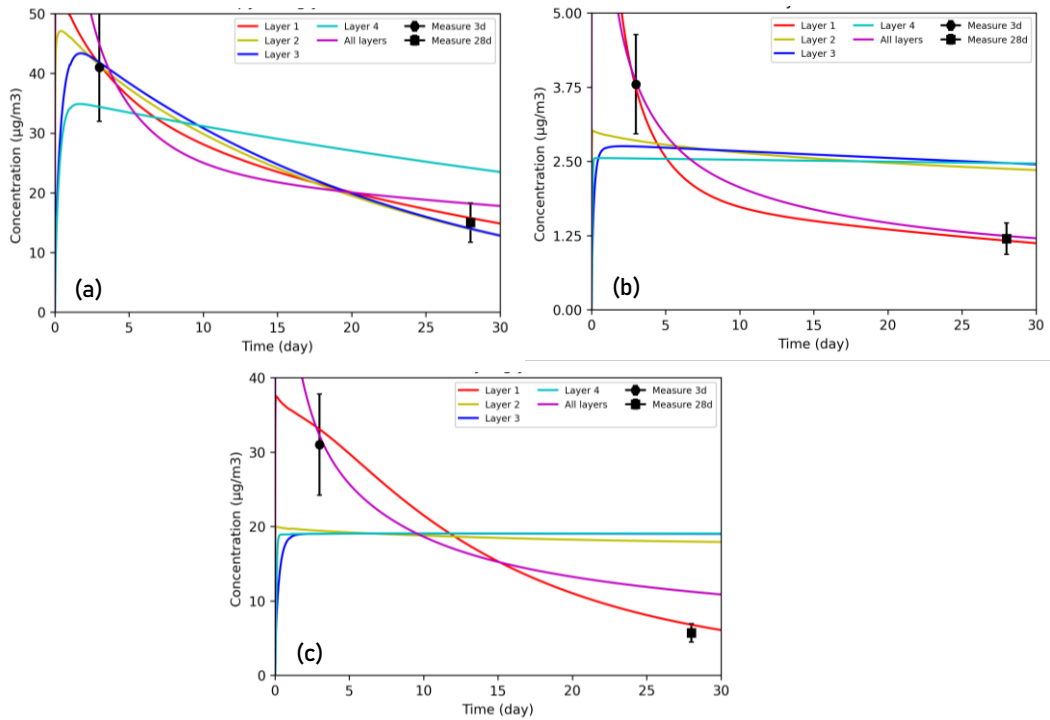
### A1.5. Multilayered material: PVC flooring

The flooring contains different layers: 1) a wear layer of PVC, 2) a layer of glass fibre, 3) another layer of PVC for resistance and 4) a polyurethane foam backing. No information is available on the thicknesses of the layers, except for the first wear layer that is 0.8 mm thick. For the other layers, average proportions were considered from products on the market, and presented in Annex Table 2.

*Annex Table 2: Material composition and thickness of different layers of PVC flooring*

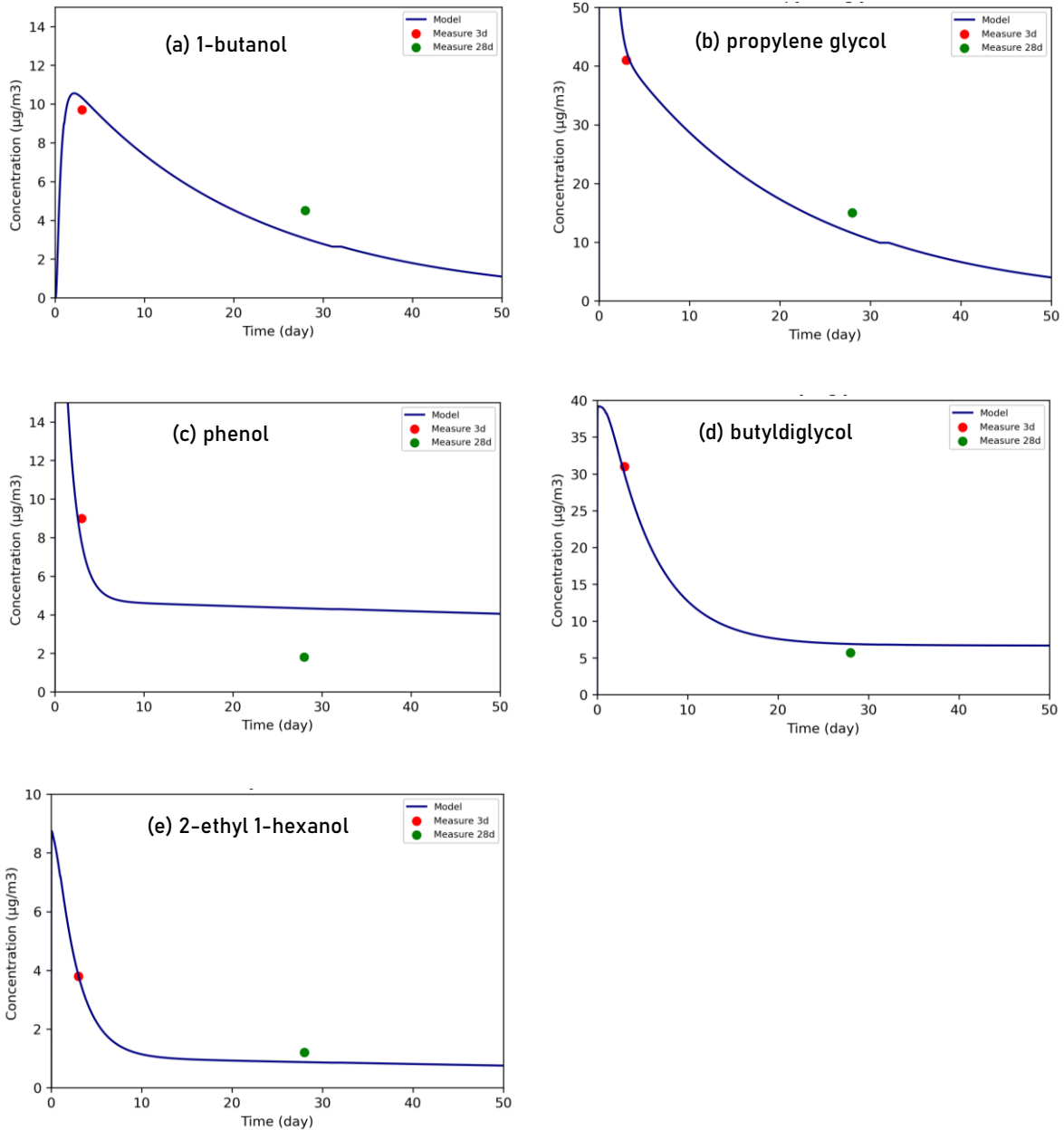
Material	Thickness (mm)
PVC wear layer	0.8
Glass fibre	0.2
PVC	2
PU foam	0.25

Annex Figure 4 shows concentrations of three substances emitted by PVC flooring considering their initial presence in each layer, or in all layers.



*Annex Figure 4: Concentration curves for (a) propylene glycol, (b) 2-ethyl 1-hexanol and (c) butyldiglycol considering emissions from different layers of PVC flooring*

From Annex Figure 4, we note that propylene glycol, 2-ethyl 1-hexanol and butyldiglycol are most likely initially present in layer 1. 2-ethyl 1-hexanol could also be initially present in all layers, though the deviation is slightly higher than for presence in layer 1. Based on these assumed positions, optimal parameters are calculated for layers 1 and 3. Predicted and measured concentrations of emitted substances are shown in Annex Figure 5.



*Annex Figure 5: Predicted air concentration curves and measured points of the different substances emitted by PVC flooring: (a) 1-butanol, (b) propylene glycol, (c) phenol, (d) butyldiglycol and (e) 2-ethyl 1-hexanol*

The predictions agree with measurements, MALD and RMSLE uncertainty factors being 1.6 and 2.8 respectively for all substances. This exploratory study demonstrates that the position of the substance within the material influences its emission profile and is expected to affect consequent health damages. It also shows that the proposed method can help identify layers in which substances are likely to be initially present. However, this study is mainly for the purpose of exploring the applicability of the methodology to multilayered material, but there are substantial uncertainties relative to layer thicknesses.

## A2.1. VOCs emitted by occupants and activities

*Annex Table 3: VOCs emitted by exhaled air and their emission rates per occupant (Moser et al. 2005)*

Substance	CAS number	Emission rate ( $\mu\text{g}/\text{h}/\text{occ}$ )
Formaldehyde	50-00-0	2.00
Methanol	67-56-1	74.30
Acetaldehyde	75-07-0	29.50
Ethanol	64-17-5	23.20
Acetone	67-64-1	524.20
Acetic acid	64-19-7	55.70
Isoprene	78-79-5	53.20
Benzene	71-43-2	1.90
Toluene	108-88-3	13.50
Phenol	108-95-2	8.50
Xylene (o,m,p)	1130-20-7	4.20
Hydrogen sulfide	7783-06-4	0.20
Acetonitrile	75-05-8	6.10
Dimethyl sulfide	75-18-3	10.70
Carbon dioxide	124-38-9	28

*Annex Table 4: VOCs emitted by shoes and their emission rates per occupant (Kurosawa et al. 2008)*

Substance	CAS number	Emission rate ( $\mu\text{g}/\text{h}/\text{occ}$ )
Toluene	108-88-3	140
Xylene	1330-20-7	26
Styrene	100-42-5	2

*Annex Table 5: VOCs emitted by skin and their emission rates per occupant (Mochalski et al. 2014)*

Substance	CAS number	Emission rate (µg/h/occ)
Acetaldehyde	75-07-0	1.16E-03
2-Propenal	67-63-0	1.27E-04
Butanal, 3-methyl-	590-86-3	1.25E-03
n-Propanal	123-38-6	7.54E-05
2-Propenal, 2-methyl-	78-85-3	1.28E-04
Propanal, 2-methyl-	78-84-2	8.84E-05
n-Hexanal	66-25-1	4.40E-04
n-Nonanal	124-19-6	8.51E-04
n-Octanal	124-13-0	3.99E-04
n-Heptanal	111-71-7	2.62E-04
2-Butenal, (E)-	123-73-9	1.64E-05
n-Pentanal	110-62-3	5.67E-05
Benzaldehyde	100-52-7	1.20E-03
n-Butanal	123-72-8	4.49E-05
2-Hexenal, (E)-	505-57-7	2.33E-05
2-Butenal, 3-methyl-	107-86-8	1.00E-04
Butanal, 2-ethyl-	97-96-1	1.07E-05
1-Heptene	592-76-7	1.90E-05
n-Heptane	142-82-5	3.18E-05
n-Octane	111-65-9	8.81E-05
Isoprene	78-79-5	3.28E-05
n-Nonane	111-84-2	1.63E-04
1-Octene	111-66-0	3.30E-05
1-Nonene	124-11-8	4.60E-05
Propene	115-07-1	3.64E-05
1-Pentene	109-67-1	7.63E-06
n-Pentane	109-66-0	1.58E-05
Acetone	67-64-1	6.90E-03
2-Butanone	78-93-3	4.98E-05
2-Pentanone	107-87-9	1.80E-05
5-Hepten-2-one, 6-methyl-	110-93-0	1.76E-03
3-Buten-2-one	78-94-4	5.47E-05
3-Penten-2-one, 4-methyl-	141-79-7	8.46E-05
2-Hexanone	591-78-6	1.72E-06
2-Heptanone	110-43-0	7.15E-06
Furan, 2-pentyl-	3777-69-3	2.72E-05
1,3-Dioxolane, 2-methyl-	497-26-7	2.25E-05
DL-Limonene	138-86-3	1.16E-04
p-Cymene	99-87-6	2.02E-05
γ-Butyrolactone	96-48-0	1.83E-04
β-Pinene	127-91-3	8.13E-06
Eucalyptol	470-82-6	1.68E-05
Ethyl Acetate	141-78-6	1.73E-04
Isopropyl acetate	108-21-4	1.24E-04
n-Butyl acetate	123-86-4	6.01E-03
Ethanol	64-17-5	9.98E-04
2-Propanol	67-63-0	9.85E-05
Dimethyl sulfide	75-18-3	1.30E-05
Acetonitrile	75-05-8	2.52E-05

*Annex Table 6: VOCs emitted by air freshener and their emission rates (Singer et al. 2006)*

Substance	CAS number	Emission rate ( $\mu\text{g/h}$ )
d-Limonene	5989-27-5	1625
Dihydromyrcenol	18479-58-8	7500
Linalool	78-70-6	6167
Linalyl acetate	115-95-7	2625
beta-Citronellol	7540-51-4	1417
alpha-Citral	141-27-5	321
3,7-Dimethyl-3-octanol	78-69-3	4083
Benzyl acetate	140-11-4	13333
Bornyl acetate	76-49-3	19167

*Annex Table 7: VOCs emitted by an all-purpose counter cleaner and their emission rates (Singer et al. 2006)*

Substance	CAS number	Emission rate ( $\mu\text{g/m}^2/\text{h}$ )
Limonene	138-86-3	7225
alpha-Pinene	80-56-8	397
Camphene	79-92-5	595
beta-Pinene	127-91-3	43
alpha-Phellandrene	99-83-2	170
alpha-Terpinene	99-86-5	630
d-Limonene	5989-27-5	7225
gamma-Terpinene	99-85-4	815
Terpinolene	586-62-9	7296
1-Terpineol	586-82-3	1275
beta-Terpineol	138-87-4	602
4-Terpineol	562-74-3	630
alpha-Terpineol	98-55-5	5242
gamma-Terpineol	586-81-2	560
4-Isopropyltoluene	99-87-6	1155
Eucalyptol	470-82-6	2125

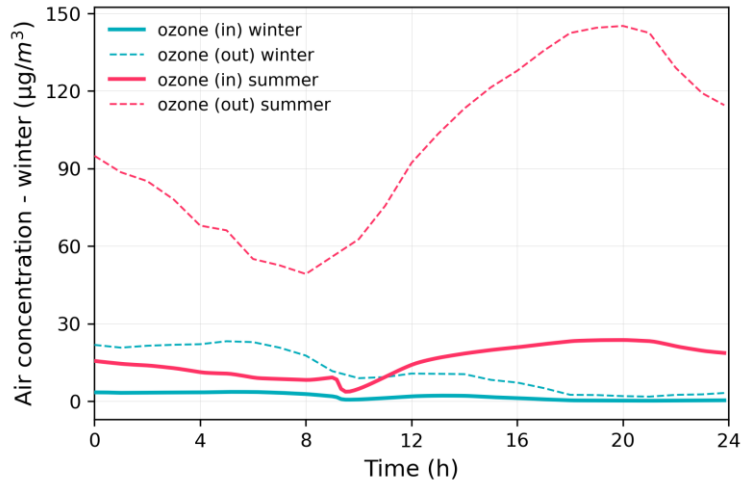
*Annex Table 8: VOCs emitted by an all-purpose floor cleaner and their emission rates (Singer et al. 2006)*

Substance	CAS number	Emission rate ( $\mu\text{g/m}^2/\text{h}$ )
Limonene	138-86-3	1133
alpha-Pinene	80-56-8	64
Camphene	79-92-5	99
beta-Pinene	127-91-3	7
alpha-Phellandrene	99-83-2	28
alpha-Terpinene	99-86-5	113
d-Limonene	5989-27-5	1133
gamma-Terpinene	99-85-4	120
Terpinolene	586-62-9	1275
1-Terpineol	586-82-3	262
beta-Terpineol	138-87-4	127
4-Terpineol	562-74-3	106
alpha-Terpineol	98-55-5	1629
gamma-Terpineol	586-81-2	177
4-Isopropyltoluene	99-87-6	177
Eucalyptol	470-82-6	319

## A2.2. Indoor and outdoor ozone concentrations

Ozone concentrations indoors and outdoors in summer and winter are presented in Annex Figure

6.

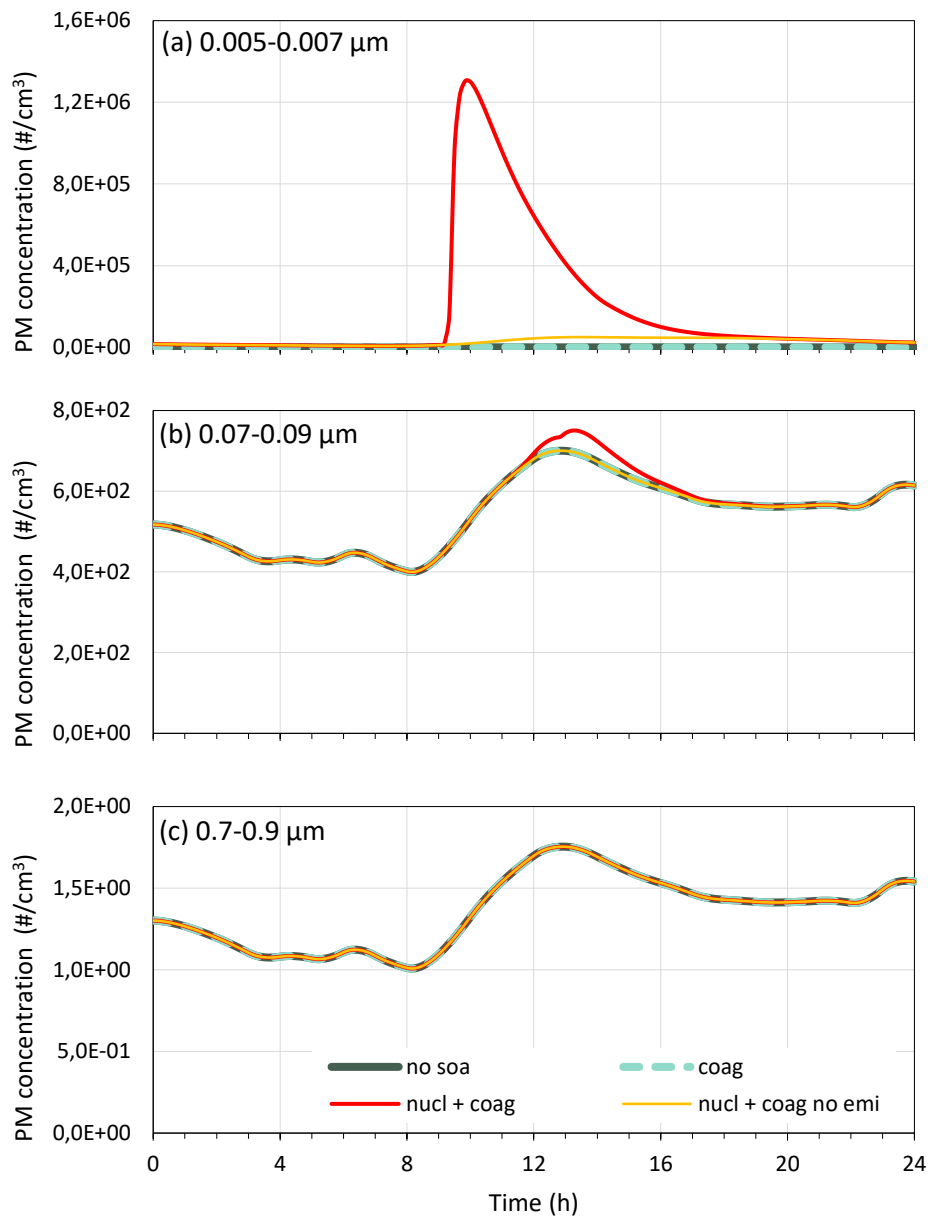


*Annex Figure 6: Indoor and outdoor ozone concentrations in winter and summer*

We note that indoor ozone concentrations are higher in summer than in winter, since outdoor concentrations are also higher.

## A2.3. Particle number with SOA formation

In INCA-Indoor, particles are separated into 32 categories according to their diameter, ranging from 0.005 µm to 50 µm. For each particle category, the mass and number of particles can be obtained. Annex Figure 7 illustrates the particle number in the following categories: (a) 0.005-0.007 µm, (b) 0.07-0.09 µm and (c) 0.7-0.9 µm. We note that the scales are different in each plot: the particle number concentration goes up to over  $10^6$  #/cm<sup>3</sup> for the smallest particles in category (a), 750 #/cm<sup>3</sup> for category (b) and less than 2 #/cm<sup>3</sup> for category (c).



*Annex Figure 7: Particle number concentration with and without SOA (coagulation, coag and nucleation, nucl), with and without emissions for particles in three categories: (a) 0.005-0.007  $\mu\text{m}$ , (b) 0.07-0.09  $\mu\text{m}$  and (c) 0.7-0.9  $\mu\text{m}$*

No increase in particle number concentration is observed for particles having a larger diameter than 0.9  $\mu\text{m}$ .



### A3.1. Integrated exposure-response model

The mortality attributed to  $PM_{2.5}$ ,  $M_{PM_{2.5}}$ , is expressed as an attribution factor multiplied by the total number of deaths.

$$M_{PM_{2.5}} = ARF \times M \quad (A-12)$$

Where  $M$  is the total number of deaths for the specific disease. The  $ARF$  is expressed as a function of  $RR$  (Nasari et al. 2016; Fantke et al. 2019):

$$ARF = (RR - 1)/RR \quad (A-13)$$

The relative risk ( $RR$ ) as expressed below (Burnett et al. 2014):

$$RR(C) = \begin{cases} 1 + \alpha \times (1 - e^{-\beta \times (C - C_0)^\delta} & \text{for } C \geq C_0 \\ 1 & \text{for } C < C_0 \end{cases} \quad (A-14)$$

$C$  represents the exposure concentration,  $C_0$  is the concentration below which there is no additional risk,  $1 + \alpha$  is the maximum relative risk,  $\beta$  is the ratio of relative risk at low-to-high exposures and  $\delta$  is the power of  $PM_{2.5}$  exposure concentration.  $C_0$ ,  $\alpha$ ,  $\beta$  and  $\delta$  have been computed from the mean of 1000 equally-likely combinations.

### A3.2. Resuspension

Resuspension rates linked to different activities are given in Annex Table 9.

*Annex Table 9: Resuspension rates of PM<sub>2.5</sub> for different indoor activities and the data sources*

<b>Activity</b>	<b>PM<sub>2.5</sub> emission rate (mg.min<sup>-1</sup>)</b>	<b>Reference</b>
1 person walk (low)	0.005	B. Wang et al. (2021)
1 person walk (medium)	0.008	B. Wang et al. (2021)
Dance on wood	0.03	Ferro et al. (2004)
Fold blankets	0.03	Ferro et al. (2004)
Sweeping (low)	0.05	He et al. (2004)
Heater - electric fan	0.05	He et al. (2004)
Vacuuming (low)	0.07	He et al. (2004)
Dusting	0.09	He et al. (2004)
1 person walk (high)	0.12	B. Wang et al. (2021)
Washing machine	0.12	He et al. (2004)
Make bed/fold clothes	0.15	Ferro et al. (2004)
Vacuum (medium)	0.17	Ferro et al. (2004)
Dance on rug	0.18	Ferro et al. (2004)
Dry dust	0.33	Ferro et al. (2004)
Vacuum (high)	0.46	Ferro et al. (2004)
Sweeping (high)	1.51	Aquilina and Camilleri (2022)

### **A3.3. Occupant emissions**

Occupant emissions were measured by Licina et al. (2017) for different activity levels: seated with moderate activity, seated with intensive activity and walking at 80 steps/min. The emission rates for different particle diameters are presented in Annex Table 10.

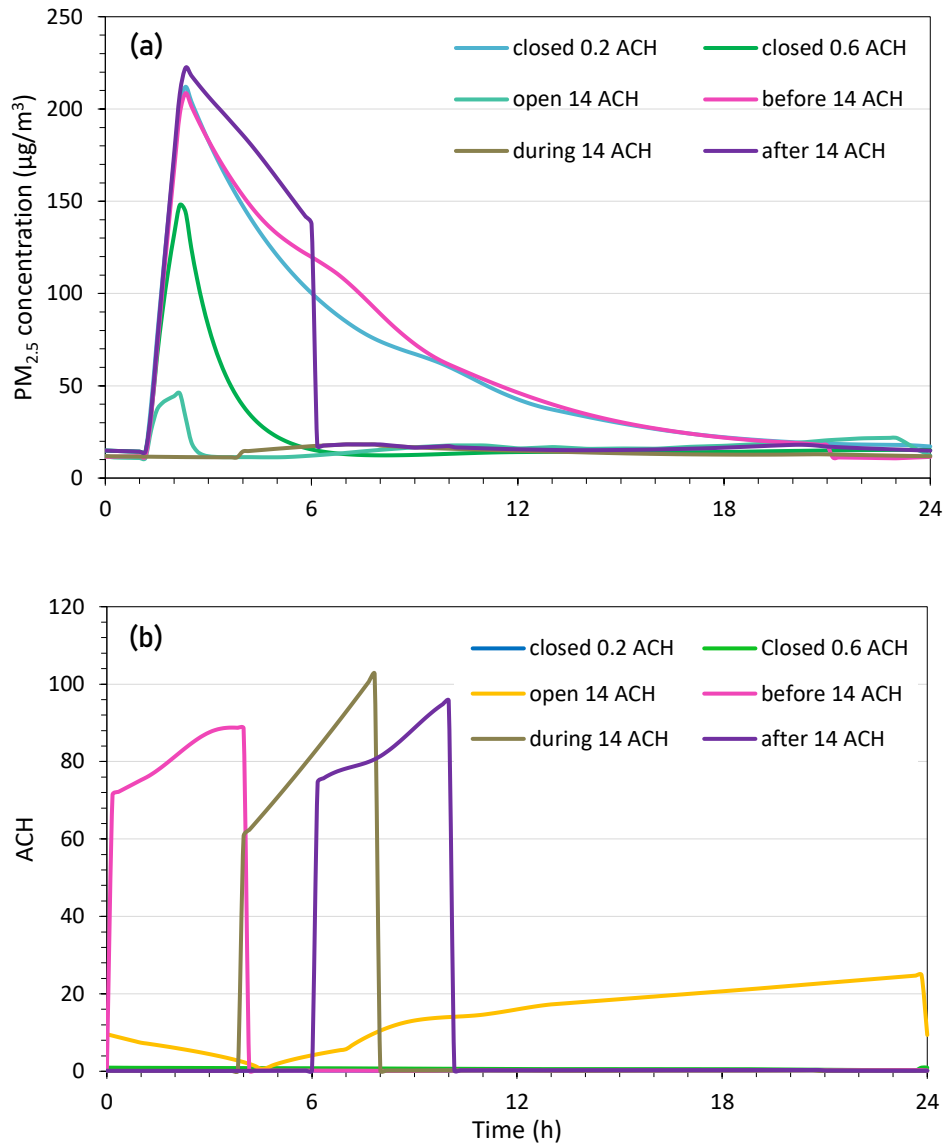
*Annex Table 10: Emission rates for particles of different diameters obtained from Licina et al. (2017)*

<b>Activity</b>	<b>PM size (<math>\mu\text{m}</math>)</b>	<b>Emission (<math>\mu\text{g/h}</math>)</b>
Seated with moderate activity	0.3-0.5	1.2
	0.5-1	2.6
	1-2	5.5
	2-3	24.7
	3-4	6.5
	4-5	12.2
	5-7	35.1
	7-10	88.6
Seated with intensive activity	0.3-0.5	2.2
	0.5-1	6.1
	1-2	6.9
	2-3	35.3
	3-4	11.9
	4-5	25.6
	5-7	62.9
	7-10	107.7
Walking	0.3-0.5	2.0
	0.5-1	5.1
	1-2	11.1
	2-3	60.8
	3-4	29.4
	4-5	46.6
	5-7	93.2
	7-10	218.2

Green-shaded areas represent diameters considered, though they also include diameters between 2.5  $\mu\text{m}$  and 3  $\mu\text{m}$ . In order to account for only  $\text{PM}_{2.5}$ , this category was divided into two (2-2.5  $\mu\text{m}$  and 2.5-3  $\mu\text{m}$ ) and an equal distribution was considered (which is a reasonable approximation given the step-shaped emission v/s diameter curve).

### A3.4. PM<sub>2.5</sub> concentrations according to ventilation scenario

Concentration curves due to toasting/cooking on an electric stove for an hour are presented in Annex Figure 8 for window-opening scenarios: always closed with 0.2 ACH and 3 ACH for the other scenarios, namely open before, open during and open after.



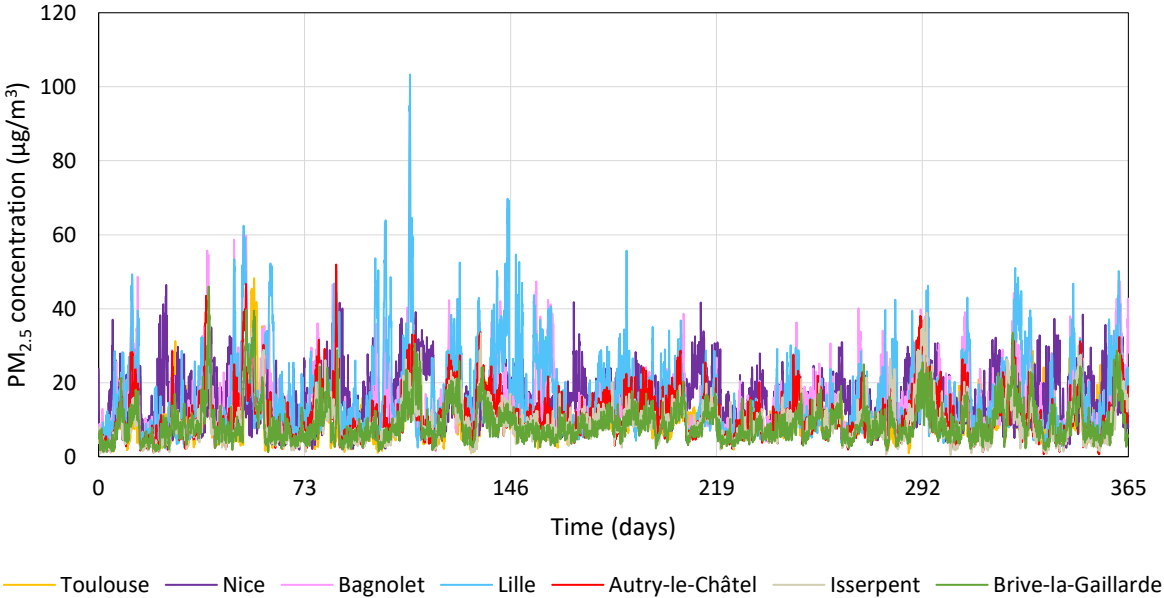
*Annex Figure 8: (a) Indoor air PM<sub>2.5</sub> concentrations due to toasting/cooking on an electric stove for one hour and (b) air change rates for 5 ventilation scenarios: windows always closed with 24-h average ACH of 0.2 ACH, windows always open, windows open before, during and after activity with 24-h average ACH of 3 ACH for the last four scenarios*

When windows are open before, there is no effect on particles emitted afterwards by the activity and hence concentrations match the “always closed” scenario. For “open after” scenario, a sharp decrease in concentrations is observed when the window is opened. Indoor concentrations are lower for “open during” as compared to “always open” because, since the 24-average is the same in both scenarios,

the ventilation rate is much higher during the activity and low before and after the activity in the “open during” scenario. We note, from Annex Figure 8 (a), that besides having a lower peak, decay rates are more important at higher ventilation rates, hence causing a decrease in exposure.

### A3.5. Outdoor PM<sub>2.5</sub> concentrations in France

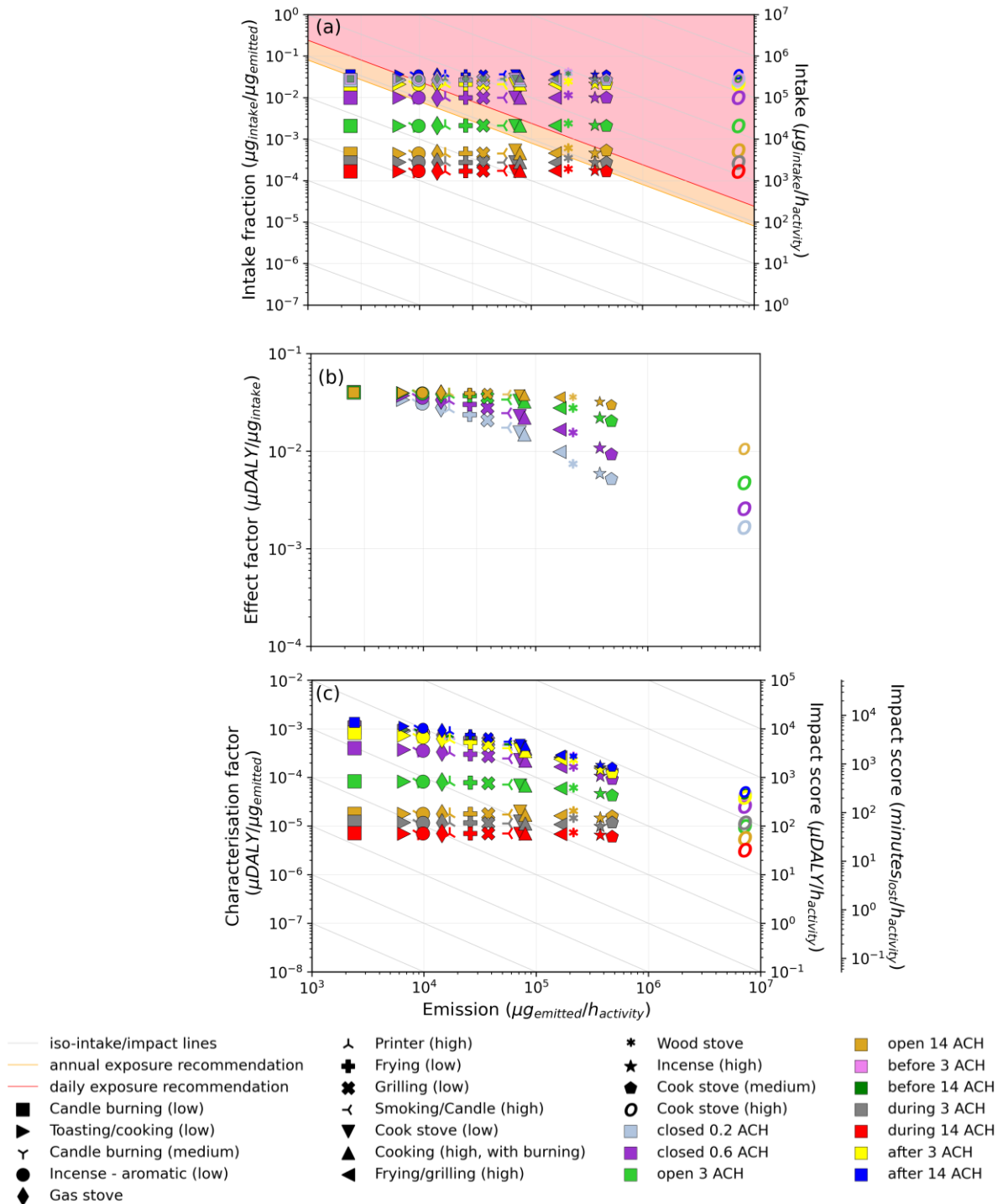
Outdoor PM<sub>2.5</sub> concentrations in 2018 for seven French towns are shown in Annex Figure 9.



Annex Figure 9: Outdoor PM<sub>2.5</sub> concentrations in seven French towns (Airparif)

## A3.6. Effect factors, characterisation factors and impact scores

### A3.6.1. Results for the larger room



Annex Figure 10: (a) Intake fractions ( $\mu\text{g}_{\text{intake}}/\mu\text{g}_{\text{emitted}}$ ) for all activities and ventilation scenarios on the primary y-axis and total intake ( $\mu\text{g}_{\text{intake}}/h_{\text{activity}}$ ) on the secondary y-axis, with iso-intake diagonal lines in grey, and intake higher than annual and daily recommendations represented by yellow and red shaded areas, (b) effect factors ( $\mu\text{DALY}/\mu\text{g}_{\text{intake}}$ ) for different activities and four standard ventilation scenarios, and (c) characterisation factors ( $\mu\text{DALY}/\mu\text{g}_{\text{emitted}}$ ) on the primary y-

axis and health damages ( $\mu\text{DALY}/h_{\text{activity}}$  and  $\text{minuteslost}/h_{\text{activity}}$ ) on the two secondary y-axes (left and right) with iso-impact diagonal lines in grey

### A3.6.2. Summarised results for all scenarios

Effect factors, characterisation factors and impact scores for all activities and scenarios, and for small ( $30\text{ m}^3$ ) and large ( $67\text{ m}^3$ ) rooms are presented in Annex Table 11, Annex Table 12, Annex Table 13, Annex Table 14, and Annex Table 15.

*Annex Table 11: Effect factors for  $\text{PM}_{2.5}$  emission from 19 activities for 10 ventilation scenarios in a small room ( $30\text{ m}^3$ )*

	Effect factor ( $\mu\text{DALY}/\mu\text{g}_{\text{intake}}$ )									
	AC 0.2ACH	AC 0.6ACH	OA 3ACH	AO 14ACH	OB 3ACH	OB 14ACH	OD 3ACH	OD 14ACH	OA 3ACH	OA 14ACH
Candle burning (low)	31.70	39.48	40.24	39.75	30.92	30.83	43.64	44.19	33.63	32.85
Toasting/cooking electric (low)	23.07	33.79	38.70	38.64	22.49	22.68	40.72	42.67	25.94	25.35
Candle burning (medium)	20.90	31.89	38.02	38.09	20.27	20.35	40.71	42.69	23.71	23.22
Incense (low)	20.11	31.22	37.76	37.84	19.47	19.55	40.71	42.71	22.95	22.50
Gas stove	17.10	28.48	36.53	36.71	16.44	16.51	40.68	42.75	19.96	19.62
Printer (high)	15.92	27.36	35.90	36.58	15.25	15.27	40.68	42.69	18.72	18.40
Frying (low)	12.73	24.08	34.03	34.48	12.11	12.15	40.62	42.77	15.45	15.21
Grilling (low)	10.25	21.23	32.08	32.61	9.69	9.73	40.57	42.78	12.75	12.57
Smoking/Candle with eucalyptus oil diffusion (high)	7.83	17.88	29.49	30.03	7.39	7.42	40.47	42.80	9.90	9.77
Cookstove (low)	6.69	16.27	28.06	30.02	6.24	6.24	43.12	44.14	8.55	8.47
Cooking (high, with burning)	6.35	15.28	27.23	27.65	6.02	6.04	40.36	42.81	8.00	7.91
Frying/grilling (high)	4.41	10.19	22.00	22.31	4.25	4.25	39.95	42.82	5.24	5.19
Wood heater	3.30	8.73	20.11	22.18	3.04	3.22	39.64	42.45	4.45	4.56
Incense (high)	3.22	6.08	15.73	15.49	3.11	3.11	39.02	42.83	3.69	3.67
Cookstove (average)	2.95	5.40	14.12	16.52	2.85	2.85	38.61	42.68	3.38	3.36
Cookstove (high)	0.62	1.72	3.40	3.82	0.57	0.57	26.19	42.19	0.85	0.84

*Annex Table 12: Effect factors for PM<sub>2.5</sub> emission from 19 activities for 10 ventilation scenarios in a large room (67 m<sup>3</sup>)*

	Effect factor (μDALY/μgintake)									
	AC 0.2ACH	AC 0.6ACH	OA 3ACH	AO 14ACH	OB 3ACH	OB 14ACH	OD 3ACH	OD 14ACH	OA 3ACH	OA 14ACH
Candle burning (low)	40.31	40.35	40.30	40.09	39.38	39.16	44.51	43.14	39.67	37.46
Toasting/cooking electric (low)	33.59	37.25	39.58	39.97	33.06	33.27	42.55	41.98	34.27	30.85
Candle burning (medium)	31.64	36.07	39.27	39.91	31.08	31.22	42.49	41.92	32.52	28.86
Incense (low)	30.95	35.63	39.15	39.90	30.38	30.51	42.47	41.90	31.91	28.17
Gas stove	28.14	33.72	38.53	39.77	27.55	27.65	42.34	41.79	29.35	25.39
Printer (high)	26.96	32.82	38.14	39.62	26.35	26.42	42.31	41.83	28.22	24.19
Frying (low)	23.68	30.24	37.10	39.40	23.06	23.12	42.05	41.59	25.13	20.95
Grilling (low)	20.81	27.79	35.87	39.05	20.19	20.24	41.76	41.40	22.36	18.13
Smoking/Candle with eucalyptus oil diffusion (high)	17.46	24.74	34.05	38.46	16.85	16.89	41.28	41.08	19.07	14.89
Cookstove (low)	15.60	23.05	32.96	37.57	14.92	14.92	44.55	42.05	17.33	13.21
Cooking (high, with burning)	14.87	22.22	32.34	37.82	14.28	14.31	40.76	40.73	16.48	12.44
Frying/grilling (high)	9.87	16.72	27.94	35.79	9.39	9.41	39.04	39.60	11.25	8.01
Wood heater	7.01	14.72	26.23	34.10	6.58	6.91	42.73	39.21	9.56	6.68
Incense (high)	5.92	10.80	22.06	32.11	5.66	5.67	35.79	37.25	6.73	4.98
Cookstove (average)	5.21	9.31	20.42	29.90	5.01	5.01	42.78	36.65	5.84	4.46
Cookstove (high)	1.66	2.58	4.75	10.58	1.59	1.59	42.73	19.06	1.84	1.38



*Annex Table 13: Characterisation factors for PM<sub>2.5</sub> emission from 19 activities for 10 ventilation scenarios in a small room (30 m<sup>3</sup>)*

	Characterisation factor (μDALY/μg <sub>emitted</sub> )									
	AC 0.2ACH	AC 0.6ACH	OA 3ACH	AO 14ACH	OB 3ACH	OB 14ACH	OD 3ACH	OD 14ACH	OA 3ACH	OA 14ACH
Candle burning (low)	3.43E-3	9.62E-4	1.94E-4	1.35E-4	3.68E-3	3.67E-3	2.73E-6	3.42E-8	2.52E-3	2.50E-3
Toasting/cooking electric (low)	2.55E-3	8.27E-4	1.87E-4	1.38E-4	2.75E-3	2.72E-3	2.56E-6	5.44E-8	1.95E-3	1.93E-3
Candle burning (medium)	2.28E-3	7.82E-4	1.84E-4	1.39E-4	2.43E-3	2.43E-3	2.55E-6	4.09E-8	1.78E-3	1.77E-3
Incense (low)	2.20E-3	7.68E-4	1.84E-4	1.41E-4	2.34E-3	2.33E-3	2.55E-6	4.39E-8	1.72E-3	1.71E-3
Gas stove	1.87E-3	7.04E-4	1.78E-4	1.45E-4	1.97E-3	1.97E-3	2.55E-6	4.64E-8	1.50E-3	1.49E-3
Printer (high)	1.73E-3	6.68E-4	1.73E-4	1.27E-4	1.82E-3	1.82E-3	2.54E-6	3.46E-8	1.41E-3	1.40E-3
Frying (low)	1.39E-3	5.96E-4	1.66E-4	1.40E-4	1.45E-3	1.45E-3	2.54E-6	4.42E-8	1.16E-3	1.16E-3
Grilling (low)	1.12E-3	5.27E-4	1.57E-4	1.36E-4	1.16E-3	1.16E-3	2.54E-6	4.68E-8	9.58E-4	9.57E-4
Smoking/Candle with eucalyptus oil diffusion (high)	8.53E-4	4.45E-4	1.45E-4	1.30E-4	8.84E-4	8.83E-4	2.53E-6	4.78E-8	7.44E-4	7.44E-4
Cookstove (low)	7.26E-4	3.96E-4	1.35E-4	1.02E-4	7.79E-4	7.78E-4	2.70E-6	3.37E-8	6.44E-4	6.45E-4
Cooking (high, with burning)	6.92E-4	3.81E-4	1.34E-4	1.24E-4	7.20E-4	7.19E-4	2.53E-6	4.98E-8	6.01E-4	6.02E-4
Frying/grilling (high)	4.80E-4	2.55E-4	1.08E-4	1.04E-4	5.06E-4	5.05E-4	2.50E-6	4.86E-8	3.93E-4	3.95E-4
Wood heater	2.86E-4	2.14E-4	9.75E-5	8.08E-5	3.80E-4	4.00E-4	2.67E-6	3.89E-8	3.63E-4	3.51E-4
Incense (high)	3.51E-4	1.53E-4	7.78E-5	7.87E-5	3.71E-4	3.70E-4	2.44E-6	5.27E-8	2.77E-4	2.80E-4
Cookstove (average)	3.20E-4	1.32E-4	6.82E-5	5.75E-5	3.35E-4	3.35E-4	2.41E-6	3.45E-8	2.53E-4	2.56E-4
Cookstove (high)	6.72E-5	4.20E-5	1.65E-5	1.36E-5	6.69E-5	6.68E-5	1.64E-6	3.43E-8	6.36E-5	6.38E-5

*Annex Table 14: Characterisation factors for PM<sub>2.5</sub> emission from 19 activities for 10 ventilation scenarios in a large room (67 m<sup>3</sup>)*

	Characterisation factor (μDALY/μg <sub>emitted</sub> )									
	AC 0.2ACH	AC 0.6ACH	OA 3ACH	AO 14ACH	OB 3ACH	OB 14ACH	OD 3ACH	OD 14ACH	OA 3ACH	OA 14ACH
Candle burning (low)	1.06E-3	4.01E-4	8.38E-5	1.78E-5	1.12E-3	1.12E-3	1.23E-5	7.13E-6	8.40E-4	1.37E-3
Toasting/cooking electric (low)	8.95E-4	3.70E-4	8.27E-5	1.78E-5	9.52E-4	9.50E-4	1.18E-5	6.99E-6	7.26E-4	1.13E-3
Candle burning (medium)	8.36E-4	3.58E-4	8.22E-5	1.78E-5	8.89E-4	8.90E-4	1.17E-5	7.01E-6	6.89E-4	1.05E-3
Incense (low)	8.18E-4	3.54E-4	8.21E-5	1.79E-5	8.69E-4	8.70E-4	1.17E-5	7.04E-6	6.76E-4	1.03E-3
Gas stove	7.44E-4	3.35E-4	8.11E-5	1.79E-5	7.88E-4	7.89E-4	1.17E-5	7.08E-6	6.22E-4	9.27E-4
Printer (high)	7.10E-4	3.26E-4	7.95E-5	1.76E-5	7.52E-4	7.53E-4	1.17E-5	6.94E-6	5.98E-4	8.83E-4
Frying (low)	6.25E-4	3.00E-4	7.83E-5	1.78E-5	6.58E-4	6.59E-4	1.16E-5	7.08E-6	5.32E-4	7.65E-4
Grilling (low)	5.49E-4	2.76E-4	7.58E-5	1.76E-5	5.77E-4	5.77E-4	1.15E-5	7.08E-6	4.74E-4	6.62E-4
Smoking/Candle with eucalyptus oil diffusion (high)	4.61E-4	2.46E-4	7.21E-5	1.74E-5	4.81E-4	4.81E-4	1.14E-5	7.05E-6	4.04E-4	5.44E-4
Cookstove (low)	4.18E-4	2.29E-4	6.86E-5	1.98E-5	4.35E-4	4.35E-4	1.23E-5	6.95E-6	3.67E-4	4.83E-4
Cooking (high, with burning)	3.93E-4	2.21E-4	6.86E-5	1.72E-5	4.08E-4	4.08E-4	1.12E-5	7.02E-6	3.49E-4	4.54E-4
Frying/grilling (high)	2.60E-4	1.66E-4	5.94E-5	1.63E-5	2.68E-4	2.68E-4	1.08E-5	6.86E-6	2.38E-4	2.92E-4
Wood heater	2.43E-4	1.46E-4	5.48E-5	1.81E-5	2.53E-4	2.45E-4	1.30E-5	6.56E-6	2.02E-4	2.50E-4
Incense (high)	1.56E-4	1.07E-4	4.71E-5	1.47E-5	1.61E-4	1.61E-4	9.87E-6	6.51E-6	1.43E-4	1.82E-4
Cookstove (average)	1.37E-4	9.25E-5	4.26E-5	1.58E-5	1.42E-4	1.42E-4	1.18E-5	6.08E-6	1.24E-4	1.63E-4
Cookstove (high)	4.34E-5	2.56E-5	9.91E-6	5.60E-6	4.50E-5	4.50E-5	1.18E-5	3.18E-6	3.90E-5	5.04E-5

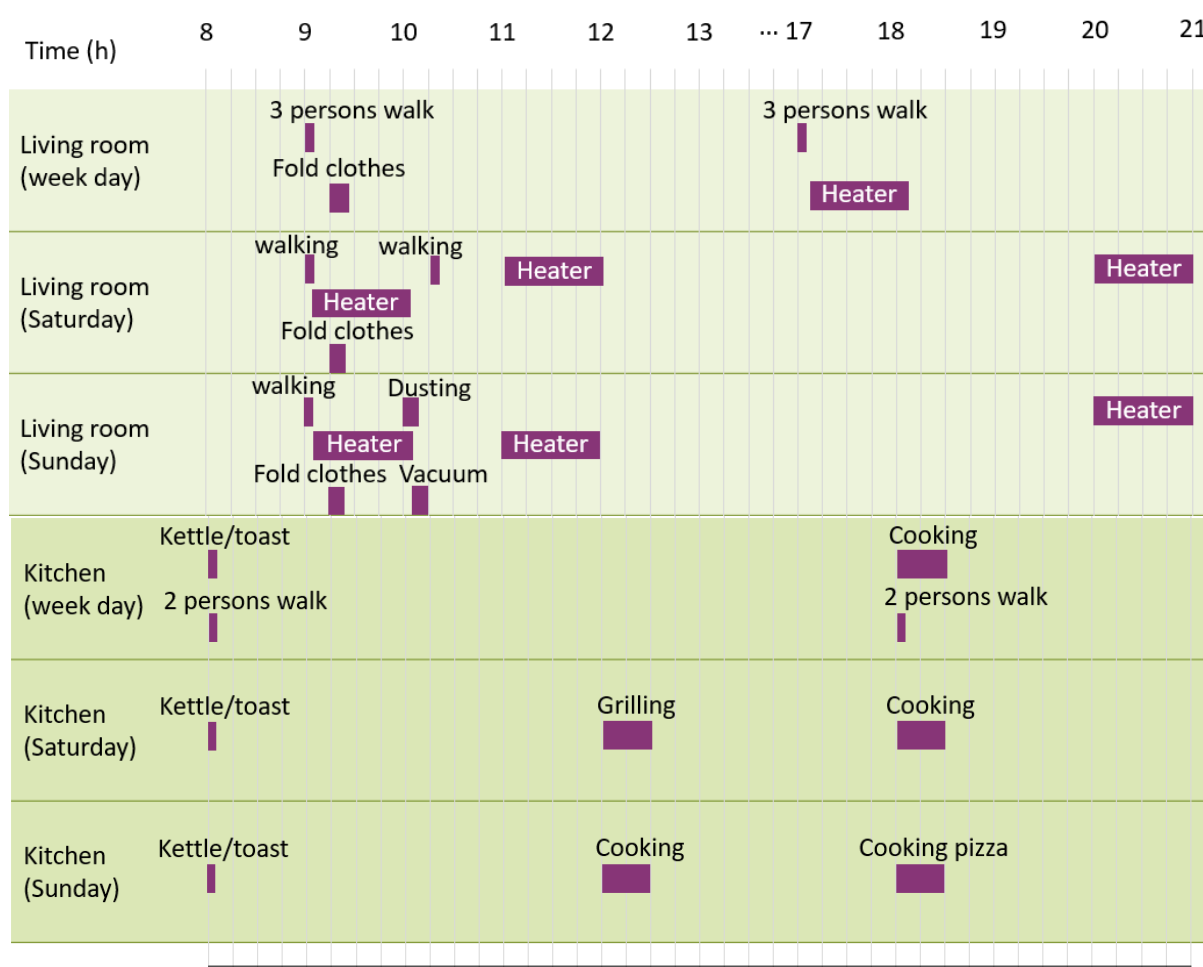
*Annex Table 15: Impact scores for PM<sub>2.5</sub> emission from 19 activities for 10 ventilation scenarios in a large room (67 m<sup>3</sup>)*

	Impact score (μDALY/FU)									
	AC 0.2ACH	AC 0.6ACH	OA 3ACH	AO 14ACH	OB 3ACH	OB 14ACH	OD 3ACH	OD 14ACH	OA 3ACH	OA 14ACH
Candle burning (low)	2.55	0.96	0.20	0.04	2.70	2.68	0.03	0.02	2.02	3.28
Toasting/cooking electric (low)	5.91	2.44	0.55	0.12	6.28	6.27	0.08	0.05	4.79	7.44
Candle burning (medium)	7.43	3.18	0.73	0.16	7.89	7.91	0.10	0.06	6.12	9.36
Incense (low)	8.02	3.47	0.80	0.18	8.52	8.53	0.11	0.07	6.62	10.08
Gas stove	10.72	4.82	1.17	0.26	11.35	11.35	0.17	0.10	8.95	13.35
Printer (high)	12.02	5.51	1.34	0.30	12.72	12.74	0.20	0.12	10.11	14.94
Frying (low)	16.15	7.76	2.02	0.46	17.01	17.03	0.30	0.18	13.76	19.77
Grilling (low)	20.45	10.27	2.82	0.66	21.46	21.47	0.43	0.26	17.63	24.65
Smoking/Candle with eucalyptus oil diffusion (high)	26.27	14.00	4.11	0.99	27.42	27.42	0.65	0.40	23.02	30.98
Cookstove (low)	30.09	16.48	4.94	1.42	31.34	31.29	0.89	0.50	26.42	34.76
Cooking (high, with burning)	31.33	17.60	5.48	1.37	32.54	32.54	0.90	0.56	27.84	36.23
Frying/grilling (high)	42.61	27.19	9.73	2.67	43.87	43.87	1.76	1.12	39.02	47.88
Wood heater	51.90	31.20	11.71	3.86	54.00	52.38	2.79	1.40	43.25	53.33
Incense (high)	58.13	39.96	17.55	5.48	60.16	60.13	3.68	2.43	53.11	67.72
Cookstove (average)	64.80	43.82	20.18	7.48	67.38	67.37	5.58	2.88	58.62	77.21
Cookstove (high)	312.57	184.34	71.35	40.30	324.31	324.27	84.70	22.87	281.04	362.77

### A4.1. Emissions and resuspension of PM<sub>2.5</sub>: residential

PM<sub>2.5</sub>-emitting activities in the residential living room and kitchen are presented in Annex Table 16 for week days.

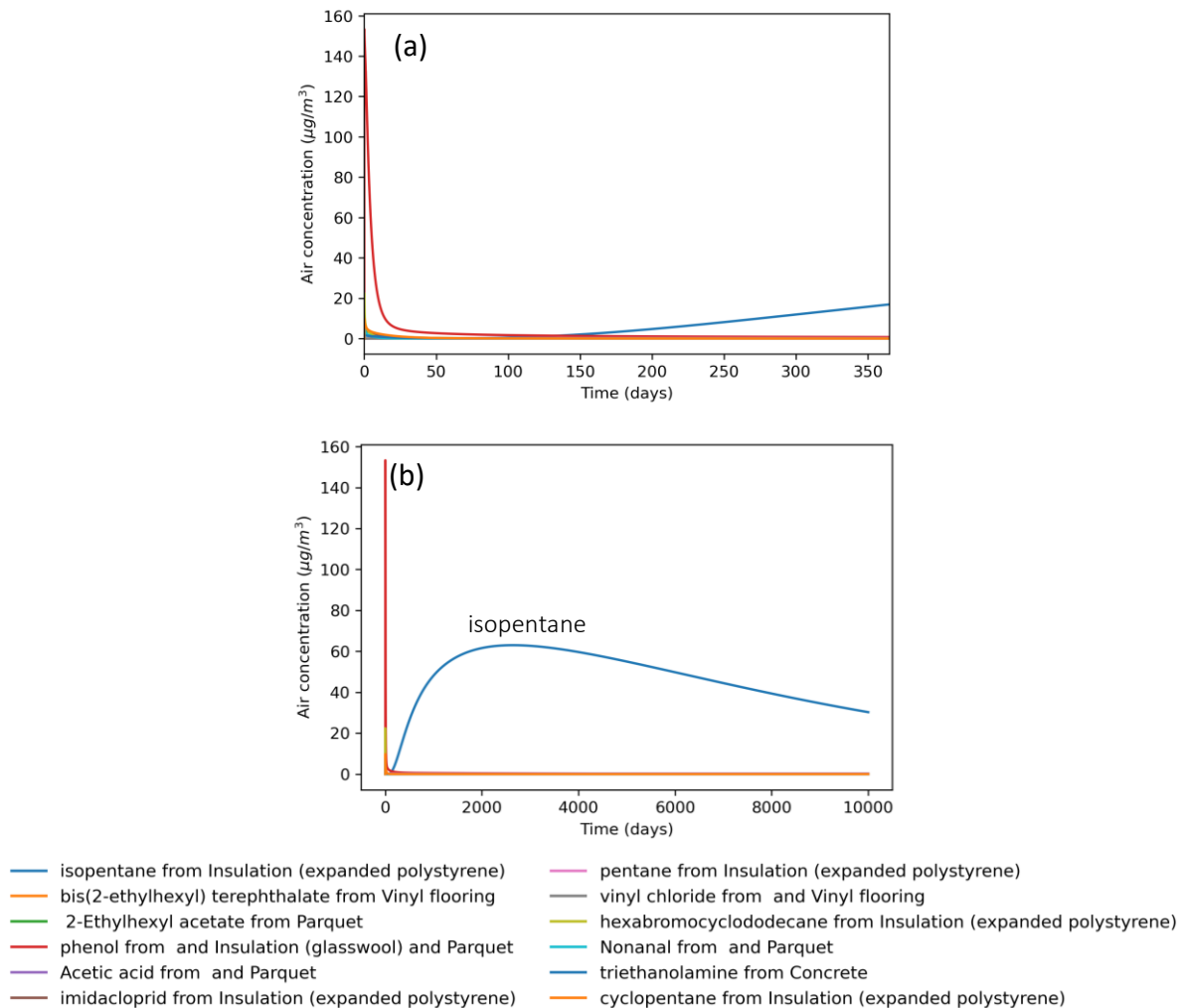
*Annex Table 16: Activity scenario in the rooms of the residential building (living room and kitchen) during week days and weekends*



Electrical stoves are used for cooking, grilling and making pizza. We consider three types of heating in the living room: gas, electrical fan and heating stove using coal fuel. Fugitive PM<sub>2.5</sub> emissions from the heating stove were calculated as described in Chapter 4, using the average heating power (W) over the heating season calculated by Pleiades STD, and presented in Annex Table 31. We note that the heating type (radiant heat stove with coal fuel) represents certain rural uses specifically, and is not representative of current best practices, for which PM<sub>2.5</sub> emission rates were not available (e.g. the Flamme Verte label in France). It is considered that burning while cooking occurs once a week, increasing emission rates.

## A4.2. Material VOC concentration curves (multilayered model)

Concentration curves of 19 substances emitted by the flooring composition in the meeting room are presented in Annex Figure 11, with a zoom on a) 0-365 days and for b) 0-10000 days.



*Annex Figure 11: Concentration curves for different VOCs emitted by the meeting room flooring, composed of concrete, expanded polystyrene insulation and bamboo flooring (parquet) for a) 0 to 365 days and b) 0 to 10000 days*

## A4.3. Materials and chemicals

Material properties are given in Annex Table 17 and the list of released chemicals for each substance are given in Annex Table 18 – Annex Table 25.

*Annex Table 17: Material properties*

Material	Thickness (m)	$\tau$ (K)	$b$ (-)	$\beta$ (-)	Density (kg/m <sup>3</sup> )
Paint (latex)	0.0003	0	-2.26	2.92	1540
Gypsum	0.0125	1676	-5.77	1.28	850
Insulation (glasswool)	0.1	1676	-7.35	1.06	12

Concrete	0.2	0	0.33	2.2	2300
Insulation (PUF)	0.1	1676	-7.35	1.06	70
Bamboo flooring	0.013	1676	-5.61	1.36	757
Insulation (XPS)	0.1	1676	-7.35	1.06	25
Vinyl flooring	0.00325	1676	-6.77	2.26	1200
Carpet	0.0055	1676	-1.23	1.97	226

*Annex Table 18: VOC content of carpet (Pharos, Healthy Building Network 2000)*

Substance	CAS Number	Mass fraction
styrene	100-42-5	4.58E-05
caprolactam	105-60-2	8.35E-04
1,3-butadiene	106-99-0	4.58E-05
propylene	115-07-1	9.70E-05
perfluorohexanoic acid	307-24-4	1.74E-04

*Annex Table 19: VOC content of concrete (Pharos, Healthy Building Network 2000)*

Substance	CAS Number	Mass fraction
triethanolamine	102-71-6	2.50E-05

*Annex Table 20: VOC content of gypsum ceiling (Pharos, Healthy Building Network 2000)*

Substance	CAS Number	Mass fraction
ethylene glycol	107-21-1	5.50E-03
2-methyl-2,4-pentanediol	107-41-5	5.50E-03
methyl isobutyl ketone	108-10-1	5.50E-03
phenol	108-95-2	3.15E-05
diethylene glycol	111-46-6	5.50E-03
4-hydroxy-4-methyl-2-penta0	123-42-2	5.50E-03
1,3-pentanediol, 2,2,4-trimethyl-, monoisobutyrate	25265-77-4	5.50E-03
formaldehyde	50-00-0	3.15E-05
urea	57-13-6	3.15E-05
vinyl chloride	75-01-4	1.65E-07
methyl ethyl ketone	78-93-3	5.50E-03

*Annex Table 21: VOC content of expanded polystyrene insulation (Pharos, Healthy Building Network 2000)*

<b>Substance</b>	<b>CAS Number</b>	<b>Mass fraction</b>
pentane	109-66-0	6.60E-03
imidacloprid	138261-41-3	2.00E-04
cyclopentane	287-92-3	6.60E-03
hexabromocyclododecane	3194-55-6	9.90E-03
isopentane	78-78-4	6.60E-03

*Annex Table 22: VOC content of glass wool insulation (Pharos, Healthy Building Network 2000)*

<b>Substance</b>	<b>CAS Number</b>	<b>Mass fraction</b>
hexamethylenetetramine	100-97-0	5.40E-04
caprolactam	105-60-2	6.50E-04
phenol	108-95-2	5.40E-04
vinyl alcohol	557-75-5	1.60E-05
citric acid	77-92-9	1.50E-02
(3-aminopropyl)triethoxysilane	919-30-2	3.40E-04
bis(2-hydroxyethyl) terephthalate	959-26-2	7.25E-05

*Annex Table 23: VOC content of polyurethane foam insulation (Pharos, Healthy Building Network 2000)*

<b>Substance</b>	<b>CAS Number</b>	<b>Mass fraction</b>
4-nonylphenol	104-40-5	5.40E-04
2-butyne-1,4-diol	110-65-6	6.50E-04
diethanolamine	111-42-2	5.40E-04
c.i. pigment red 3	2425-85-6	
diphenylmethane diisocyanate (mdi) - non isomer specific	26447-40-5	
propylene glycol	57-55-6	1.60E-05
2,4'-diphenylmethanediisocyanate	5873-54-1	1.50E-02
diethyltoluenediamine	68479-98-1	3.40E-04
(dimethylamino)cyclohexane	98-94-2	7.25E-05

*Annex Table 24: VOC content of low VOC acrylic eggshell paint (average from Pharos, Healthy Building Network 2000)*

<b>Substance</b>	<b>CAS Number</b>	<b>Mass fraction</b>
4-nonylphenol	104-40-5	1.62E-05
ethylene glycol	107-21-1	1.62E-05
ethyl acrylate	140-88-5	3.26E-04
methylchloroisothiazolinone (cit, cmit)	26172-55-4	1.10E-03
methylisothiazolinone (mit)	2682-20-4	2.90E-03
acrylic acid	79-10-7	4.89E-05
methacrylic acid	79-41-4	3.26E-04
methyl methacrylate	80-62-6	3.26E-04

*Annex Table 25: VOC content of vinyl flooring (average from Pharos, Healthy Building Network 2000)*

<b>Substance</b>	<b>CAS Number</b>	<b>Mass fraction</b>
vinyl acetate	108-05-4	1.44E-05
bis(2-ethylhexyl) terephthalate	6422-86-2	3.96E-02
vinyl chloride	75-01-4	1.44E-05

#### A4.4. Inventory of materials considered in LCA

Construction materials considered in the building's LCA, together with their quantities, are given in Annex Table 26.

*Annex Table 26: Inventory of materials implemented in the building envelope and equipment*

<b>Material</b>	<b>Quantity</b>	<b>Unit</b>
10 kW gas heater	1	-
3-core cable	480	m
Certified wood board	225	kg
Concrete	175380	kg
Copper	34	kg
Domestic PVC floor	1222	kg
Double-glazed aluminium windows	28	m <sup>2</sup>
Expanded polystyrene	86	kg
Exterior wooden door	3	m <sup>2</sup>
Extruded polystyrene	6	kg
Glass wool	216	kg
Gypsum and cardboard	2097	kg
Gypsum board	1470	kg
Interior wooden door	10	m <sup>2</sup>
Low-alloyed steel	109	kg
Polyethylene	50	kg
Polyurethane foam	243	kg
PVC	21	kg
Rockwool	123	kg
Solvent-based paint	215	m <sup>2</sup>
Tiles	192	kg
Vapour barrier (polyamide)	478	kg
Water-based paint	509	m <sup>2</sup>
Waterproof polyethylene	50	kg

#### A4.5. Material waste treatment

A summary of the waste treatment of different materials is given in Annex Table 27.

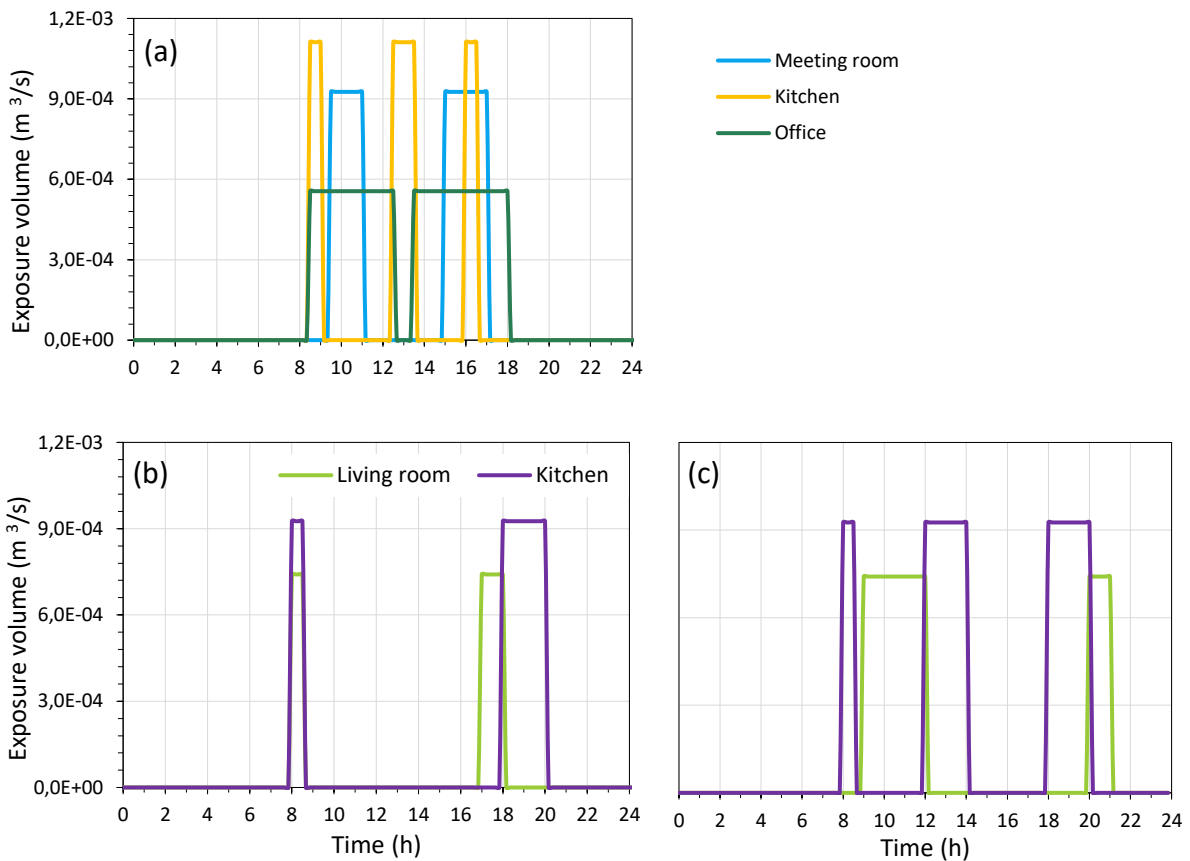


Annex Table 27: Waste treatment of different materials

Material	Waste treatment
Copper, aluminium, galvanised steel, plasterboard	Recycled
Concrete	Crushed and reused as gravels
Plastics (polystyrene, PVC, etc.) and wood	Incinerated
Others (ceramic, glass wool, rubber and rockwool)	Landfilled

#### A4.6. Occupancy

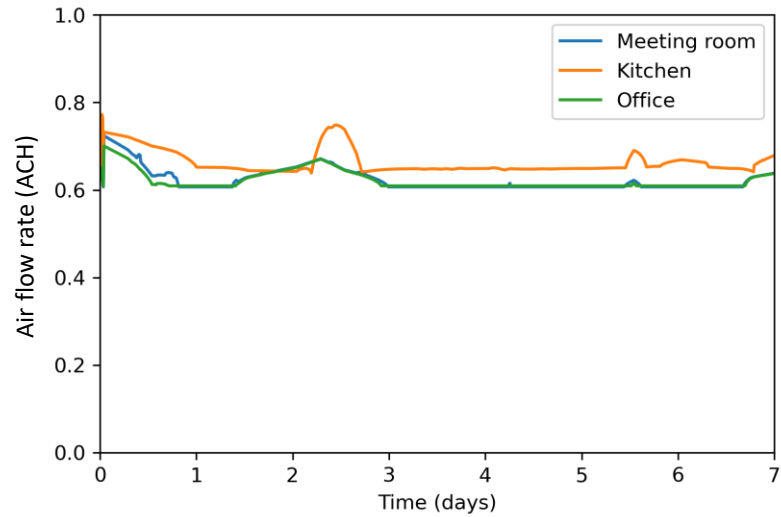
The exposure of occupants ( $m^3/s$ ), product of breathing rate and number of occupants, in the different rooms are presented in Annex Figure 12, indicating their presence in the rooms, during a typical week day in the tertiary building and during a typical day of the week and weekend in the residential building. We consider the presence of 5 persons at a time in the meeting room, 6 in the kitchen, 3 in the office and 5 in the residential rooms.



Annex Figure 12: Exposure ( $m^3/s$ ) and presence of occupants in a) tertiary rooms for a week day, b) residential rooms on a week day and c) residential rooms on a day of the weekend

### A4.7. Ventilation rates

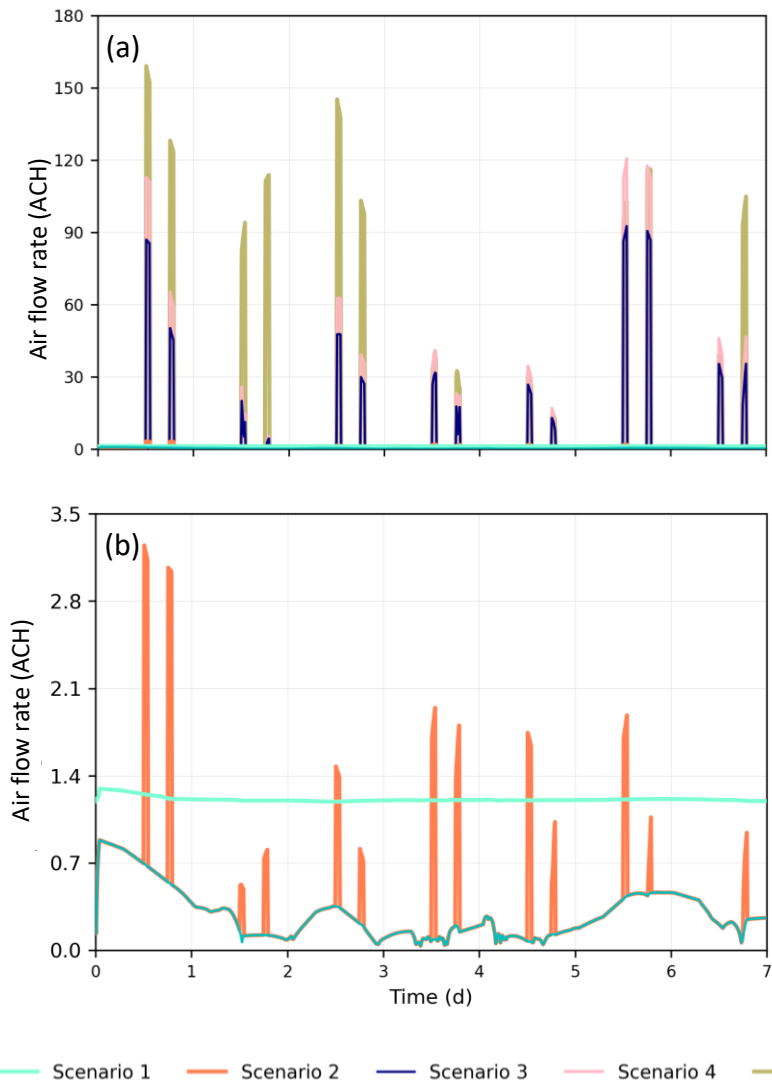
Ventilation rates in the three rooms of the office building are presented in Annex Figure 13 for a mechanical ventilation rate of 0.6 ACH.



*Annex Figure 13: Ventilation rates in the meeting room (blue), kitchen (orange) and office (green)*

We note that the ventilation rates are not constant: there is a fixed rate of 0.6 ACH from the mechanical ventilation but fluctuations due to infiltration.

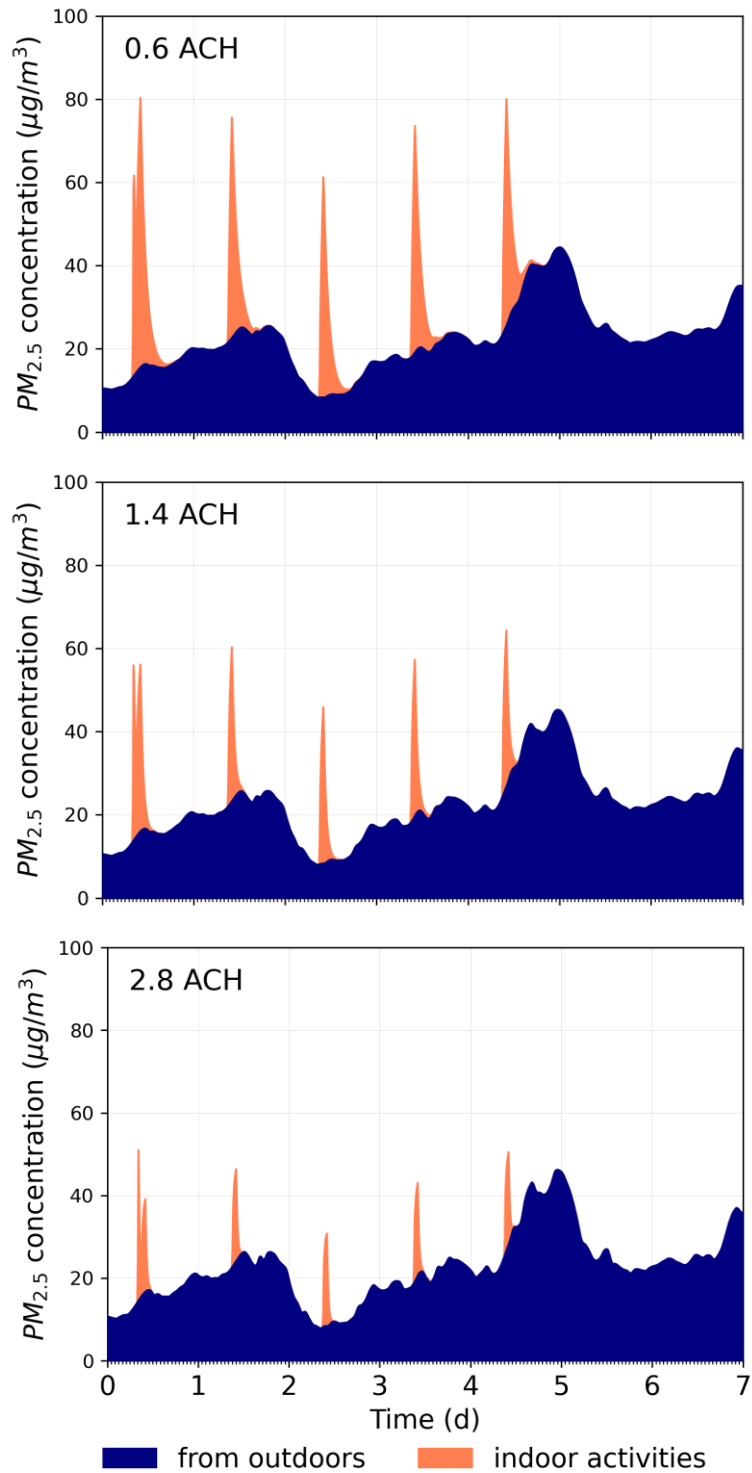
Ventilation rates in the kitchen for the different scenarios are presented in Annex Figure 14, with different scales in (a) and (b) to represent highest and lowest ventilation rates respectively.



*Annex Figure 14: Air flow rate (ACH) for the kitchen window scenarios a) for highest ventilation rates and b) for lowest ventilation rates*

Highest ventilation rates occur for cross-ventilation and lowest rates occur for the base scenario with no extraction or open windows.

In Annex Figure 15, indoor  $PM_{2.5}$  concentrations for the same activities in the meeting room, are illustrated for different ventilation rates: 0.6 ACH, 1.4 ACH and 2.8 ACH.



*Annex Figure 15: Indoor (orange) and outdoor (navy blue)  $PM_{2.5}$  concentrations in the meeting room for three ventilation rates: 0.6 ACH, 1.4 ACH and 2.8 ACH*

We note that the concentrations are different: 1) indoor  $PM_{2.5}$  concentrations due to activities (peaks) are lower for higher ventilation rates and 2) the difference between indoor and outdoor  $PM_{2.5}$  concentrations (navy blue area) are lower for higher ventilation rates. This indicates that, at higher ventilation rates without filters, more particles penetrate in the room.

#### A4.8. Heating needs

The heating need (kWh/year) and ventilation fan electricity consumption (kWh/year) for each room and each ventilation rate are given in Annex Table 28 – Annex Table 30.

*Annex Table 28: Heating needs (kWh) and ventilation fan electricity consumption (kWh) for each ventilation rate (ACH) in the meeting room*

Ventilation rate (ACH)	Heating needs (kWh)	Fan electricity consumption (kWh/year)
0.2	496	7
0.6	609	21
0.7	626	21
0.8	649	24
0.9	675	27
1.2	759	43
1.4	794	47
1.8	909	63
2.3	1024	78
2.7	1138	94
5	1710	156
15	4228	468

*Annex Table 29: Heating needs (kWh) and ventilation fan electricity consumption (kWh) for each ventilation rate (ACH) in the kitchen*

Ventilation rate (ACH)	Heating needs (kWh)	Fan electricity consumption (kWh/year)
0.2	486	6
0.6	563	16
0.7	555	17
0.8	570	19
0.9	587	21
1.60	733	33
1.8	766	37
2.3	878	49
2.9	991	61
3.5	1104	73
5	1395	122
5.8	1421	168
7.3	1668	200
8.8	1923	233
10.2	2174	265
11.7	2426	297
13.2	2685	330
15	3345	364

*Annex Table 30: Heating needs (kWh) and ventilation fan electricity consumption (kWh) for each ventilation rate (ACH) in the office*

Ventilation rate (ACH)	Heating needs (kWh)	Fan electricity consumption (kWh/year)
0.2	188	7
0.6	262	20
0.7	257	20
0.8	272	23
0.9	286	26
1.3	384	40
1.4	413	45
1.9	512	60
2.4	617	74
2.9	723	89
5	1213	148
15	3582	444

The average heating power required in the living room (W), used to calculate PM<sub>2.5</sub> emission rates (µg/h), are given in Annex Table 31.

*Annex Table 31: Average heating power (W) required for each ventilation rate (ACH) in the living room and consequent PM<sub>2.5</sub> fugitive emission rates (µg/h)*

Ventilation rate (ACH)	Heating power (W)	PM <sub>2.5</sub> emission (µg/h)
0.20	141	1.85E+03
0.60	271	3556.45
0.70	277	3.63E+03
0.80	283	3.71E+03
0.90	292	3.83E+03
1.23	317	4.16E+03
1.37	328	4.30E+03
1.82	364	4.77E+03
2.28	401	5.25E+03
2.73	438	5.75E+03
5.00	632	8.29E+03
5.24	660	8.65E+03
5.46	679	8.90E+03
5.69	699	9.17E+03
7.00	798	1.05E+04
15.00	1475	1.93E+04

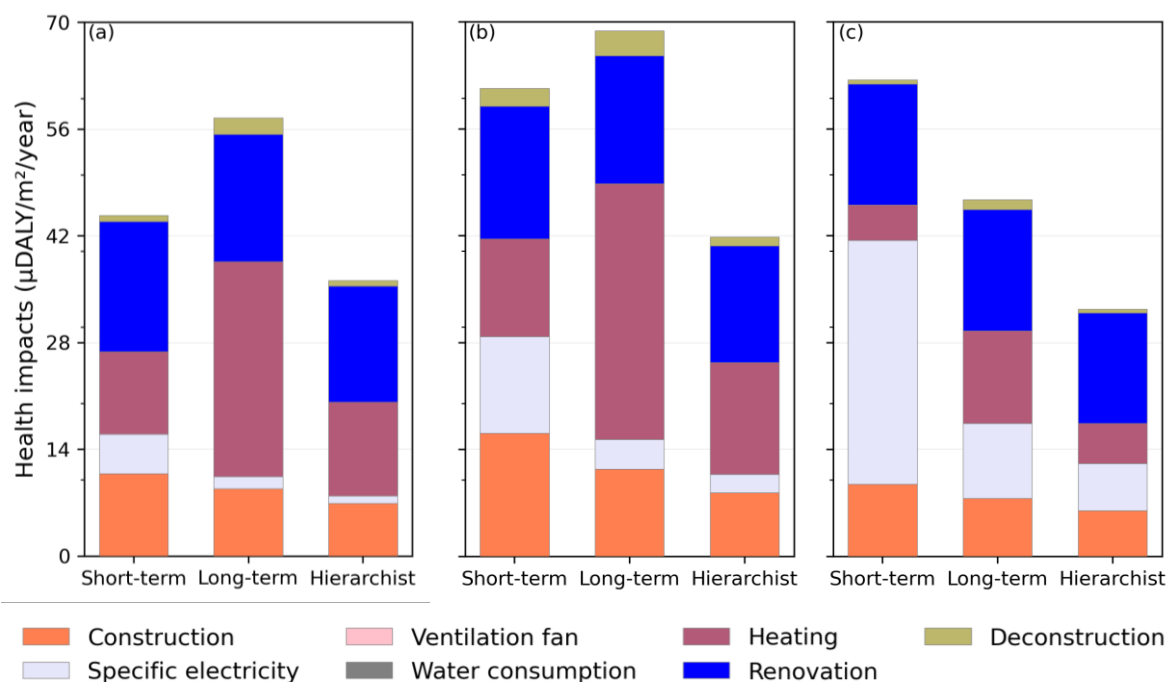
Heating needs for the kitchen (kWh) are given in Annex Table 32.

Annex Table 32: Heating needs (kWh) for the different window scenarios in the residential kitchen

Scenario	Heating needs (kWh)
0	689
1	906
2	1168
3	2168
4	2290
5	2561

#### A4.9. LCA health impacts: different methods

Results for the different rooms and over the whole life cycle are shown for short-term and long-term from the Impact World+ method and for hierarchist from the ReCiPe 2016 method in Annex Figure 16, for a ventilation rate of 0.6 ACH and gas heating.



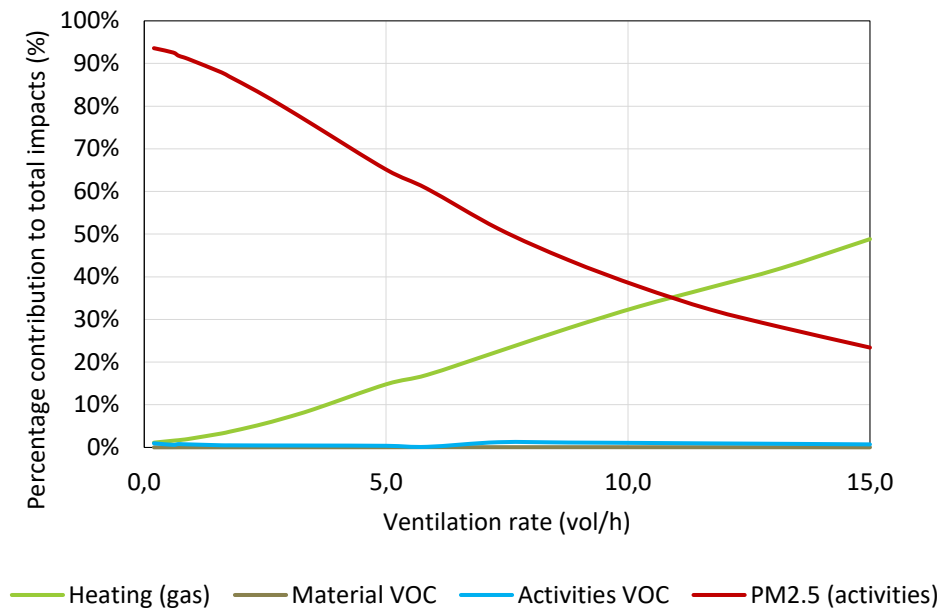
Annex Figure 16: Contributors to human health impacts for different LCA methods: Impact World+ short-term and long-term and ReCiPe 2016 Hierarchist in the a) meeting room, b) kitchen and c) office

Water use in France has a characterisation factor (CF) of  $9.49 \times 10^{-8} \text{ DALY}/\text{m}^3_{\text{consumed}}$  using ReCiPe 2016, while in Impact World+ it is  $0 \text{ DALY}/\text{m}^3_{\text{consumed}}$ . As a comparison, global water use has a CF of  $2.22 \times 10^{-6} \text{ DALY}/\text{m}^3_{\text{consumed}}$ .

#### A4.10. Contribution of $\text{PM}_{2.5}$ and heating to total impacts in the tertiary kitchen

The percentage contribution of heating (gas), VOCs from materials and activities, and  $\text{PM}_{2.5}$  from activities to impacts over the whole life cycle are shown in Annex Figure 17.  $\text{PM}_{2.5}$  has the highest

impacts at low ventilation rates, while heating has the highest impacts at the highest ventilation rates. VOC impacts are negligible.

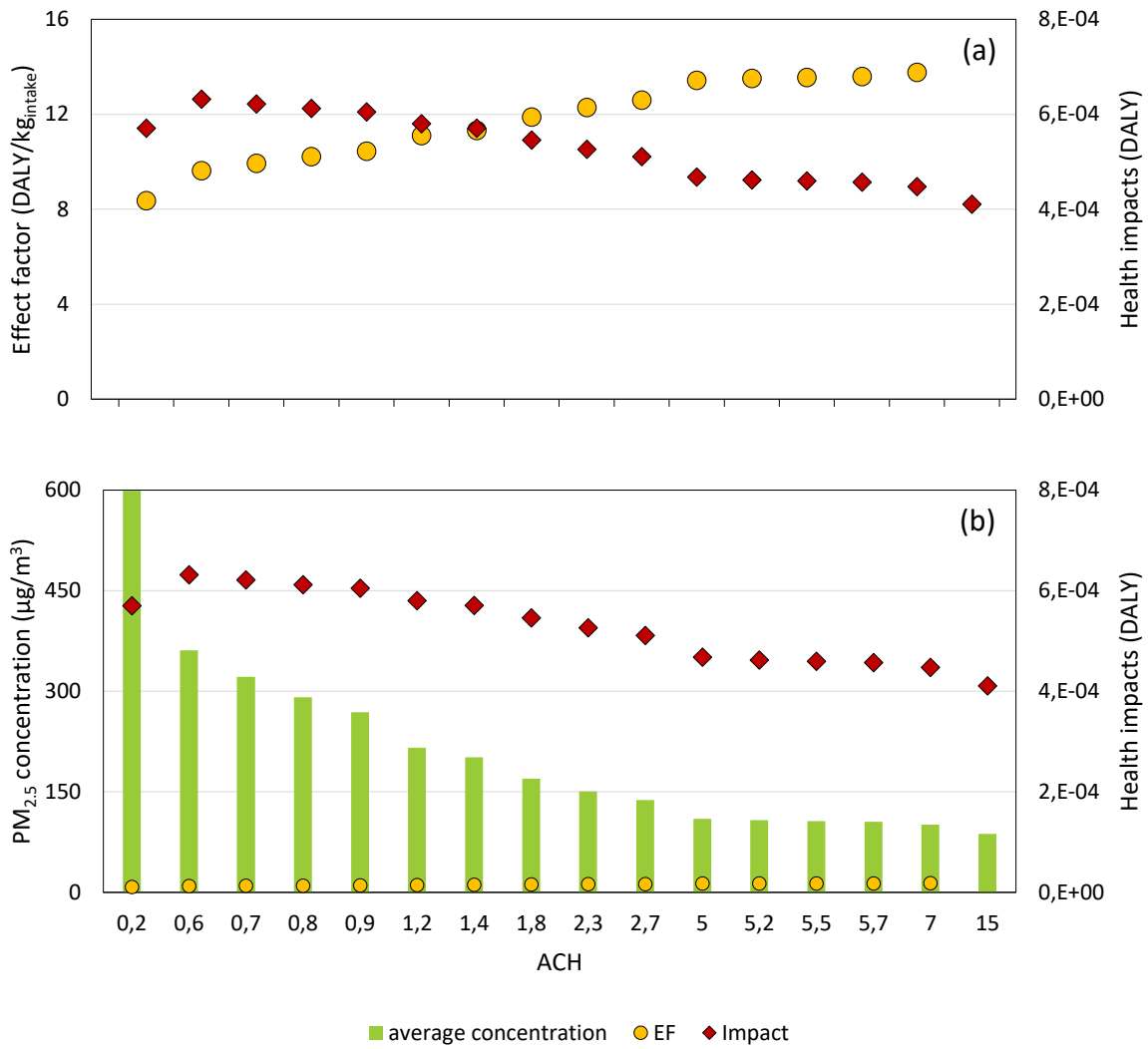


*Annex Figure 17: Percentage contribution of gas heating, material VOC emissions, activity VOC emissions and activity PM<sub>2.5</sub> emissions to total impacts (LCA+IAQ) inside the tertiary kitchen for different ventilation rates*

We note that the sum of heating and IAQ impacts do not reach 100%. The difference corresponds to contributions of other stages of the building’s life cycle to total impacts (construction, operation – excluding heating and IAQ, renovation and end-of-life).

The average indoor PM<sub>2.5</sub> concentrations (µg/m<sup>3</sup>), health impacts (DALY/week) and effect factors (DALY/kg<sub>intake</sub>) associated with coal heating in the living room for different ventilation rates are presented in Annex Figure 18.





Annex Figure 19: (a) Effect factor and health impacts and (b) average concentration and impacts according to ventilation rates associated with PM<sub>2.5</sub> emissions from a heating stove

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## RÉSUMÉ

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Les impacts de la qualité de l'air intérieur (QAI), une des principales causes de mortalité dans le monde, sont intégrés à l'analyse de cycle de vie (ACV) du bâtiment. Deux principaux polluants, dont les concentrations intérieures dépassent souvent les recommandations, sont considérés : les composés organiques volatils (COV) et les particules fines (PM<sub>2.5</sub>), provenant des matériaux de construction, de l'extérieur, des occupants et de leurs activités en intérieur. Les impacts sont calculés en nombre d'années de vie en bonne santé perdues (DALYs), unité commune à l'ACV. Les polluants sont évacués avec une bonne ventilation, mais les impacts de la consommation d'énergie pour le chauffage sont potentiellement augmentés. La prise en compte de la QAI dans l'ACV permet de trouver les débits optimaux de ventilation qui permettent de réduire les impacts totaux sur tout le cycle de vie du bâtiment.

## MOTS CLÉS

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Analyse de cycle de vie (ACV), qualité de l'air intérieur (QAI), composés organiques volatils (COV), particules fines (PM<sub>2.5</sub>), DALY, ventilation, santé

## ABSTRACT

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Human health impacts of indoor air quality (IAQ), one of the main causes of global death, are integrated to the life cycle assessment (LCA) of buildings. Two main pollutants, whose indoor concentrations often exceed recommendations, are considered: volatile organic compounds (VOCs) and fine particulate matter (PM<sub>2.5</sub>), emitted by construction materials, occupants and their activities, or penetrating from outdoors. IAQ impacts are calculated in the number of healthy life years lost (DALYs), a unit common to LCA. Pollutants can be evacuated by adequate ventilation, but impacts of heat consumption are potentially high. By integrating IAQ to LCA, optimal ventilation rates can be identified in order to reduce total impacts on the whole life cycle of the building.

## KEYWORDS

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Life cycle assessment (LCA), indoor air quality (IAQ), volatile organic compound (VOC), fine particulate matter (PM<sub>2.5</sub>), DALY, ventilation, health