

Reducing exposure to particulate matter in the indoor environment through filter application

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Lisa Corneillie, 30 May 2024

“This masterthesis is a part of an examination. Any comments made by the examination committee during the oral presentation of the master’s thesis have not been incorporated into this text.”

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Keywords

Particulate Matter – Health effects – Filter performance – Indoor air quality – DALYs

Abstract

In this dissertation the effectivity of a filter as an indoor air quality improvement strategy against particulate matter is investigated using simulations in Dymola. Therefore, a new division for particulate matter particles is developed based on the toxicity of different particles sizes and derived from literature. This new division aims to replace the deficient existing size divisions that generalize the size and source-dependent toxicity of particulate matter. After developing a filter component and adapting an existing multi-zone model, the harmfulness of the particulate matter concentration that reaches the indoor environment and the ability of filters to prevent this, were investigated in different scenarios. The influence on the overall indoor PM concentration was observed for different outdoor environments and different filters. Furthermore, the influence of PM emitted by cooking, the difference between 100% ventilation and demand-based ventilation and the implementation of open windows are also simulated. Additionally, the influence of filter efficiency variation, lifetime of the filter and subsequent energy use are investigated. The results show that a HEPA filter can sufficiently lower the indoor PM concentration for all harmful particle size fractions, while a coarse and fine filter cannot. However, cooking and window opening undermine the filter effectiveness to such an extent that a healthy indoor environment can no longer be guaranteed. A preliminary analysis of the energy use and the filter lifetime shows a significant extra energy cost when filters are used and a lifetime that requires adaptation of the range proposed by manufacturers. Analysis of the current filter testing standard shows that it is not sufficient to ensure a good comparison between different filters. The number of DALYs for PM_{2.5} exposure was at least four times smaller when a HEPA filter was present in all scenarios. However, PM was still the most damaging pollutant, before nitrogen dioxide, ozone, and formaldehyde.

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Keywords Particulate matter, Health effects, filter performance, indoor air quality, DALYs

I. INTRODUCTION

Yearly, almost 2 million deaths can be attributed to particulate matter inhalation and its subsequent health effects [1]. These effects can range from respiratory problems to heart deficiencies, cancer, and miscarriages. Since people spend about 90% of their time indoors, the largest share of

these PM-related deaths are caused by the indoor particulate matter concentration [2].

Different health effects can be linked to different size fractions of particulate matter. The smaller the particle, the deeper it can penetrate the body and the more damage it can inflict. Smaller particles are also more susceptible to reactions, due to their larger surface per unit mass ratio [3]. The chemical content of PM has a part in the overall health risk as well. Especially PAHs, mainly present in ultrafine PM, have proven to be very damaging [4]. In general, natural PM has a bigger size diameter than anthropogenic PM [5].

The existing particulate matter divisions only contain PM_{2.5} and PM₁₀ (and rarely PM₁ or PM_{0.1}). These divisions generalize many size and composition related health effects and are not able to accurately ascribe the possible health effects a specific size distribution can cause.

Furthermore, the European Air Quality Standards are not sufficiently focused on the health of a certain particle concentration, as their limit values do not align with the guidelines of the World Health Organization [6][7].

To reduce exposure to harmful indoor PM concentrations, filters can be used. Especially HEPA filters could be suitable for this purpose since they are able to remove about 99.97 % of the particles with a size diameter greater than or equal to 0.3 μm [8]. However, some questions about the effectivity of these filters in a real-life environment can be posed. In the laboratories used to test these filters, various influential parameters that influence indoor PM concentration are neglected. This could result in an undue sense of security in a real-life environment when these other factors come into play. The clogging of the filter could also result in a significant increase in energy cost and subsequent decrease in a filter lifespan. This influences the willingness of people to install a filter in their ventilation system.

Therefore, the objective of this dissertation is to research whether or not filters are able to sufficiently decrease indoor particulate matter concentration in a realistic indoor environment. Firstly, a new size division for particulate matter was developed based on the toxicity of different particles, filter efficiency curves and air tightness of the building. Secondly, a filter component was developed and an existing multi-zone model was adapted to encompass particulate matter. Lastly, a variety of simulations under different circumstances were performed to observe the influence of

various parameters. The investigated parameters include a varying outdoor PM concentration originating from different environments, different filters, the influence of cooking-related PM, demand-based ventilation, the opening of windows and the lifetime of a filter with subsequent energy use. Additionally, brief research concerning the number of DALYs or Dynamic Disability-Adjusted life years per 100 000 people resulting from the PM_{2.5} in different scenarios, was conducted. This is then compared to the number of DALYs originating from other indoor pollutants. This results in the determination of the most harmful pollutants when PM is not considered.

II. BIN SIZES

A. Based on toxicity

The size of a particle determines how far the particles can travel into the body and the specific organs or systems that can be influenced. Therefore, different boundaries were placed. PM between 10 and 5 μm is deposited in the lungs. Particles between 5 and 1 μm can enter the smallest parts of the lungs. A particle smaller than 1 μm behaves like a gas and can consequently reach the bloodstream and cause cell and DNA damage. This size fraction can also influence the hormonal systems. If a particle is smaller than 0.5 μm , the heart and coronary vessels can be reached. With a size diameter below 0.25 μm , the placental barrier can be penetrated. This can result in deficits of the baby, miscarriage or early delivery. Particles smaller than 0.2 μm can reach the brain and nervous system and damage these systems [9].

B. Based on filter efficiency

Every filter has a filter efficiency curve. Not all particle sizes are equally as effectively filtered by a filter. The MPPS or most penetrating particle size is different for all filters. A coarse or medium filter has a MPPS of about 0.2-0.3 μm . For a high efficiency filter this is 0.12-0.25 μm . Those size fractions will be filtered least efficiently [10][11].

C. Based on building shell

Particles with size 0.2-0.3 μm can penetrate the building envelop more easily than others [11].

Table 1 Resulting bin sizes based on IIA, IIB and IIC.

Nr.	Bin size	Size diameter
1	PM0.01-0.1	0.01 till 0.1 μm
2	PM0.1-0.2	0.1 till 0.2 μm
3	PM0.2-0.25	0.2 till 0.25 μm
4	PM0.25-0.3	0.25 till 0.3 μm
5	PM0.3-0.4	0.3 till 0.4 μm
6	PM0.4-0.5	0.4 till 0.5 μm
7	PM0.5-1	0.5 till 1 μm
8	PM1-2.5	1 till 2.5 μm
9	PM2.5-5	2.5 till 5 μm
10	PM5-10	5 till 10 μm

III. SIMULATION SETUP

A. Filter component

Next, a filter component was modelled in Dymola, using the Modelica language. This filter model was based on the current testing standard for high efficiency filters: NBN EN 1822-1 (2019) [8]. This testing method provides an efficiency curve and initial pressure drop. The tested velocity is determined by the manufacturer. As the filter efficiency is very velocity dependent, comparing different filters with different testing velocities is very difficult. The humidity condition is also quite broad. Since the humidity influences the settling rate, this could also result in differences. The filter is equipped with the determined bin sizes and corresponding filter efficiencies as derived from the efficiency curve of the testing standard.

B. Multi-zone model

In a following step, an existing multi-zone model developed by dr. ir. architect Klaas De Jonge is adapted to encompass particulate matter [12]. To do so, the outdoor PM concentrations were made variable and decay rates were added. This model consists of a 3-bedroom apartment with about 120 m^3 of living space. This simulation environment can be modified to investigate different case studies.



Figure 1 Floorplan of the multi-zone model [12].

C. Base scenario

As a base scenario, the apartment is equipped with a continuous D-ventilation system. The windows are closed and no occupant is present. The air tightness is constant. The results of this scenario will serve as a baseline for the different case studies. The outdoor PM concentration originates from Vilvoorde. All results are compared with the guidelines of the WHO for average annual concentration and maximum daily average concentration.

IV. SIMULATIONS

A. Varying outdoor concentration

The influence of different environments in Belgium is investigated based on data from irCELLine.be. Next to the base scenario, three different sites in Belgium were selected. These sites represent different environments that could entail different size distributions of PM. The investigated sites were just outside the ring road of Brussels (Vilvoorde), in the city center of Ghent, on the Antwerp ring road and in a more rural area in West-Flanders (Moerkerke). Based on this data, a worst-case scenario was assumed for each bin size, since exact size distributions are very place dependent.

B. Based on filter efficiency

Secondly, three filter types with different efficiencies were tested. The efficiencies for a coarse 60% filter and ePM1 55% filter for each bin size were derived by data from the Out2In research by Buildwise [13]. The efficiency for a H13 filter was derived via the mathematical approach of Chen, Ji and Zhao [14].

These efficiencies, as can be observed in table 2, were used as input for the filter component. This component is then placed in the overall multi-zone model.

Table 2 Filter efficiencies for different bin sizes.

	Coarse 60 % filter [%]	ePM1 55% filter [%]	H13 filter [%]
PM0.01-0.1	42.32	53.58	99.947
PM0.1-0.2	42.23	52.33	99.902
PM0.2-0.25	42.08	50.46	99.973
PM0.25-0.3	41.98	49.20	99.999
PM0.3-0.4	41.84	47.33	99.999
PM0.4-0.5	41.64	44.83	99.999
PM0.5-1	41.06	37.33	99.999
PM1-2.5	45.62	36.17	99.999
PM2.5-5	52.89	44.13	99.999
PM5-10	59.55	52.74	99.999

C. Indoor PM source

In the next case study, a family of four moves in the apartment. Cooking is assumed to be the most general and highest indoor PM source. Therefore, a cooking schedule with 3 basic meals, is developed as a worst-case scenario based on the occupancy of this family [15]. However, cooking is not equally distributed over all bin sizes. To determine the size distribution, a lognormal distribution was used with a mode diameter of 0.04 μm and standard deviation of 0.005 μm . This showed that close to all cooking-related PM can be attributed to the smallest size fraction of PM. A cooking hood with 70% efficiency was also present.

Table 3 Emission rate and duration of the selected meals [15].

Meal	Average emission rate [$\mu\text{g}/\text{m}^3$]	Duration [minutes]
Toast	220	3.85
Pasta Bolognese	1900	28
Chicken and vegetables stir-fry	3200	17

D. Demand based ventilation

As opposed to a ventilation system that works 100% of the time, an occupant-based system can also be investigated. In the selected system, the flow rate is adjusted to the needs of the wet rooms. However, a higher humidity accounts for the faster settling of particulate matter. Hence, this might prove to have a negative effect on the indoor particulate matter concentration.

E. Window opening schedule

The opening of a window lets in additional PM. A ventilation system with filter doesn't influence this concentration. Hence, all the particles enter the room directly. To investigate this influence, a window opening schedule was

constructed and a window component was modelled based on the research by Silke Verbruggen [16].

F. Lifetime of a filter

Lastly, a filter model with varying efficiency and pressure drop was developed. Due to the clogging of the filter, less particles penetrate the filter medium and the efficiency increases with time. However, the clogging of the filter also increases the pressure drop and the subsequent energy that is necessary to maintain a certain air velocity. To investigate this phenomenon, efficiency curves were constructed in Dymola based on real life tested data (outdoor concentration and efficiency increase) of the Out2In project [17].

The resulted mass in the filter was then linked to the corresponding pressure drop increase.¹

For the high efficiency filter, that was not part of that same research, a similar efficiency curve and pressure drop evolution was assumed.

G. DALYs

The number of DALYs for PM2.5 in the kitchen is investigated for some key scenarios. The total number of DALYs can be calculated using Eq. (1) and Eq.(2) [18].

$$\text{DALYs}(t) = \int_{t_0}^t \frac{\text{DALYs}}{\text{time}}(t) dt \quad (1)$$

$$\frac{\text{DALYs}}{\text{time}}(t) = \text{EF}_{\text{marginal}} * \text{IR}(t) \quad (2)$$

Where IR(t) is the intake rate [kg/s] and EF is the effect factor [DALY/kg]. To determine the intake rate Eq.(3) was used.

$$\text{IR}(t) = V_{\text{br}}(t) * C_{\text{exp}}(t) \quad (3)$$

where V_br is the breathing rate of time, which is assumed to be 16m³/day and C_exp is the exposure concentration profile.

H. Other pollutants

The multi-zone model already entailed various other pollutants like VOCs, nitrogen dioxide and ozone. Based on a DALYs assessment, a brief observation can be made as to which of those is most abundantly present in the indoor environment when PM is excluded.

V. RESULTS AND DISCUSSION

Simulating the indoor PM concentration for different outdoor concentrations, showed that the PM variation in Belgium is rather small. The overall average annual concentration for each bin size exceeds the guidance value regardless of the location of the outdoor PM data. The daily maximum concentration is exceeded for 90-100 days on average. Subsequently, without filter, the indoor air in Belgium is per definition unhealthy. Only the concentration of the highest bin size PM5-10 is consequently below the target value. However, there is a difference in size distribution based on location. More traffic results in more fine PM and a more

¹ Data derived from the research project Out2In conducted by Buildwise with the financial support of Innoviris Brussels

rural environment in more coarse PM. Since the smallest particles are considered the most damaging, different environments from most to least harmful are Antwerp, Vilvoorde, Ghent, Moerkerke. Or otherwise, the ring road is more damaging than the city center and the city center is more damaging than a rural environment.

When different filters are tested, a healthy environment was only obtained with the H13 filter, as can be observed in figure 2. In case of a coarse filter, the PM concentration decreases significantly, but not enough to reach healthy concentrations. Installing an ePM1 filter that is specialized in smaller particle ranges could be a good solution to lower the concentration of the most harmful particle sizes to just above the target values.

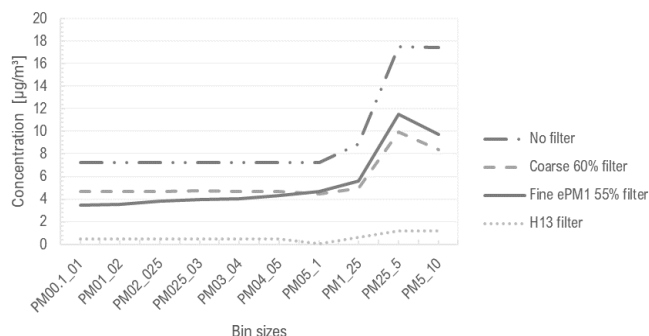


Figure 2 Comparison of the average annual concentration for each bin size in the kitchen.

The addition of an indoor PM source (cooking) results in a significant increase for the lowest and most harmful bin size. This can be observed in figure 3. The HEPA filter is not able to compensate for these peaks in concentration. Additional source control is necessary to obtain a healthy indoor environment in this case.

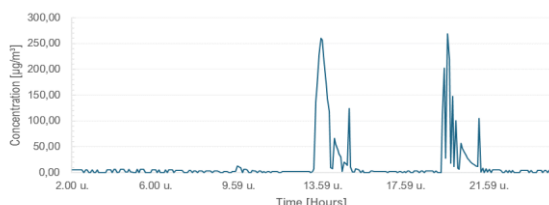


Figure 3 PM concentration for the smallest bin size in the kitchen on a day when 3 meals were cooked.

As could be expected, a demand-based ventilation system based on the occupant causes a significant increase in the indoor PM concentration. Especially in the wet rooms, PM concentration increases significantly. The continuous ventilation system allows for the faster decrease of peak concentrations.

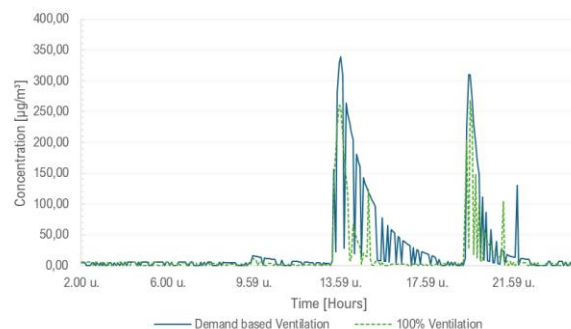


Figure 4 Average concentration of PM for the smallest bin size in the kitchen with different ventilation systems.

Introducing air exchange via open windows also has a significant influence on the indoor air quality. However, when no filter is present, the opening of windows has no influence or even a positive effect.

However, when a filter is present, the negative influence of the filter increases with increasing efficiency of that filter. The opening of a window does act as source control for cooking related PM. But when opened too long, the effect of the open windows overpowers the source control effect. This influence can be observed in figure 5 in the kitchen. Here the window is continuously open in the summer.

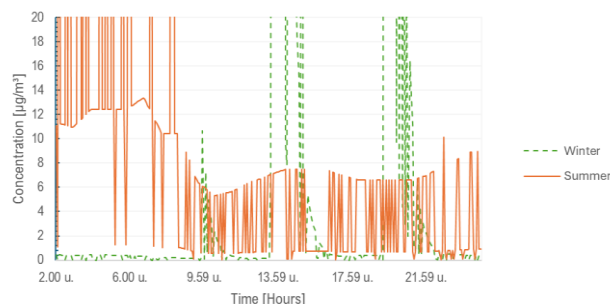


Figure 5 PM concentration for the smallest bin size in the kitchen in both summer and winter.

The annual average concentrations for the smallest bin size in the kitchen, that summarize the above can be found in table 4. This shows that the values were only below the guidelines when no windows were opened and no cooking-related PM was present. The total number of exceedance days for all rooms in different scenarios can be found in figure 6.



Figure 6 Total number of exceedance days in all zones in different scenarios.

Table 4 Average annual concentration & number of exceedance days for PM_{0.01-0.1} in the kitchen for different scenarios.

	Average annual concentration [µg/m ³]	Number of exceedance days (max 3)
TARGET VALUE	3	3
Vilvoorde	7.24	105
Coarse 60% filter	4.66	57
ePM1	3.51	34
H13 filter	0.52	0
Cooking (no filter)	13.25	206
H13 - Cooking	6.55	49
Occupant based ventilation	28.1	338
Window opening (no filter)	7.10	97
Coarse filter – Window Opening	5.74	78
H13 - Window opening	3.19	37
H13 - Window opening - Cooking	6.99	88

With the variable efficiency filter component, the lifetime of the filter can be investigated. For a coarse filter, the maximum pressure drop of 150 Pa, is reached at 84 days, which is a lot faster than was expected. This could be due to the simplifications that were made in the development of this model. It was assumed that all particles participate equally in the clogging process and that no compaction occurs. Furthermore, the model is developed with a constant outdoor PM concentration. After the addition of the mass in the filter at a certain time, the mass number was equally distributed over the ten bin sizes to construct filter efficiency curves.²

The peaks of the variable outdoor concentration could explain why results from this model deviate from the expected result based on the Out2In project [13]. Because, when the concentration for one bin size increases, they all do. The real-life variable particle concentration in the simulation is so high that filter efficiencies reach 100% in just a couple of months for a coarse filter. This improves the overall PM concentration significantly. However, since those results were not expected, more research is necessary to determine the validity of this model for a coarse filter.

For a H13 filter, efficiencies reach 100 % a lot later. The very high efficiency reduces the influence of the different and variable particle concentrations significantly. Again, the overall PM concentration improves due to the increasing efficiency. The final pressure drop is exceeded at 238 days or almost 8 months. This could mean that a filter is more likely to reach its final pressure drop closer to 6 months than to a year, as manufacturers often describe.

If the filter is not replaced when this final pressure drop is reached, the energy use of the system with H13 filter, increases till 6812.5 kWh. This is opposed to 2080.84 kWh if no filter is present. If the filter is replaced at 238 days, the energy use at that point is 3685 kWh. Thus, the timely

replacement of a filter could significantly decrease the overall energy cost.

The DALYs calculation for different scenarios confirms that any air filter can significantly decrease the number of DALYs after a year of exposure. Even when cooking and/or window opening is present, the improvements a HEPA filter causes are very significant from a health perspective in all cases. A comparison can be seen in figure 7.

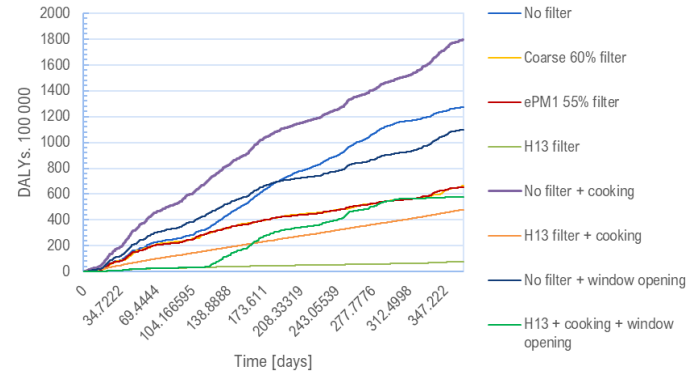


Figure 7 Number of DALYs for PM_{2.5} in the kitchen for different scenarios.

The DALYs calculation for the other pollutants shows that when cooking and window opening is present, the number of DALYs for PM_{2.5} is still far exceeding the number of DALYs for other pollutants in the same scenario. However, the pollutants that result in the highest level of healthy years lost are nitrogen dioxide, ozone and formaldehyde, as can be seen in figure 8. Furthermore, since the deposition rates for formaldehyde and nitrogen dioxide are even lower than PM, these particles could be expected to be very dangerous [19].

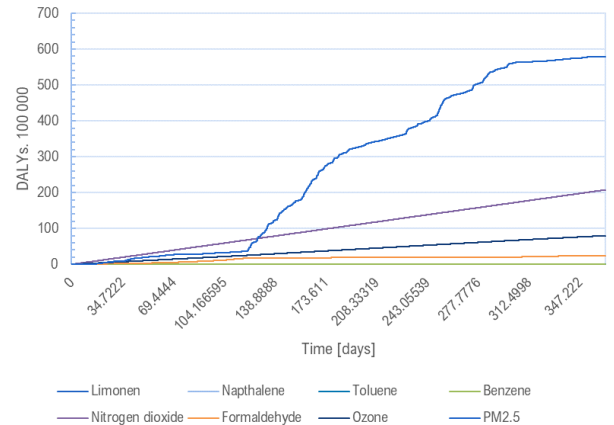


Figure 8 Number of DALYs for various pollutants for different scenarios.

VI. CONCLUSIONS

Particulate matter has proven to be a dominant factor in various health effects. To get a better idea of which size fractions result in which health effects, a new size division was developed. This division shows that particles with a size diameter below 1 µm can damage the most systems. However, larger particle sizes can already considerably damage the lungs. Therefore, all particulate matter is dangerous, but fine PM should get the most attention.

² Data derived from the research project Out2In conducted by Buildwise with the financial support of Innoviris Brussels

Furthermore, the outdoor PM concentration in Belgium only shows a very slight variation. Due to the more abundantly present smaller particles, an environment with a lot of traffic should be avoided. But overall, all air is dangerous to the general health of the public.

Additionally, only a H13 filter could sufficiently lower indoor PM concentration. In order to not undermine its effectivity, windows should stay closed as much as possible. The local peaks in PM due to cooking cannot be compensated by the high efficiency filter. To maintain a healthy environment when people cook meals, a form of local source control like an air purifier is necessary.

The ventilation system should also work 100% of the time, if a healthy environment needs to be maintained.

Figure 9 summarizes previous findings.

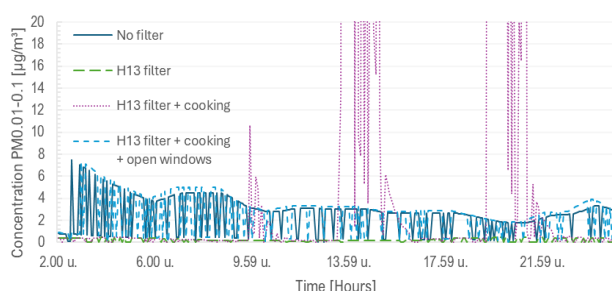


Figure 9 Average concentration PM0.01-0.1 kitchen for one day.

The lifetime of a filter remains questionable. Due to the assumptions made, the results can be called into question. More research is necessary to determine the validity of the variable efficiency component. However, these preliminary results suggest that filters reach their final pressure drop closer to 6 months than a year after installation. In any case, the replacement guidelines of a filter should be adapted. As, when a filter is replaced too early, the advantage of the increasing efficiency is negated. And when a filter is used too long, the energy use increases significantly.

The influence of filter use on the overall energy cost is also very significant. However, in the long run, many health effects and subsequent cost can be avoided. Therefore, a better indoor air quality still prevails.

Finally, the number of DALYs caused by PM in a real-life environment is still significant. Other pollutants are not nearly as dangerous to human health. The most dangerous pollutants, however, seem to be nitrogen dioxide, ozone and formaldehyde.

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Verminderen van de blootstelling aan fijn stof in de binnenomgeving door gebruik van filters

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Abstract In deze scriptie wordt de effectiviteit van een filter als strategie voor het verbeteren van de binnenluchtkwaliteit op vlak van fijnstof onderzocht met behulp van simulaties in Dymola. Hiervoor is een nieuwe indeling voor fijnstofdeeltjes ontwikkeld op basis van de toxiciteit van verschillende deeltjesgroottes, afgeleid uit de literatuur. Deze nieuwe indeling is bedoeld om de gebrekkige bestaande indelingen te vervangen, die de grootte- en bronafhankelijke toxiciteit van fijnstof veralgemenen. Na het ontwikkelen van een filtercomponent en het aanpassen van een bestaand multi-zone model, werd onderzocht hoe schadelijk de fijnstofconcentratie is die de binnenomgeving bereikt en hoe goed filters dit kunnen voorkomen in verschillende scenario's. De invloed op de totale binnenlucht PM-concentratie werd geobserveerd voor verschillende buitenomgevingen en verschillende filters. Verder zijn ook de invloed van PM die vrijkomt bij het koken, het verschil tussen 100% ventilatie en vraaggestuurde ventilatie, en de implementatie van open ramen gesimuleerd. Daarnaast werden de invloed van filterefficiëntievariatie, de levensduur van de filter en het bijbehorende energieverbruik onderzocht. De resultaten tonen aan dat een HEPA-filter de binnenlucht PM-concentratie voldoende kan verlagen voor alle schadelijke deeltjesgroottes, terwijl een grove en fijne filter dit niet kunnen. Echter, koken en het openen van ramen ondermijnen de effectiviteit van de filter dusdanig dat een gezonde binnenomgeving niet langer kan worden gegarandeerd. Een voorlopige analyse van het energieverbruik en de levensduur van het filter laat zien dat er aanzienlijke extra energiekosten zijn bij gebruik van filters en dat de levensduur een aanpassing van het door fabrikanten voorgestelde bereik, vereist. Analyse van de huidige filterteststandaard toont aan dat deze niet voldoende is om een goede vergelijking tussen verschillende filters te garanderen. Het aantal DALY's (Disability-Adjusted Life Years) voor blootstelling aan PM_{2.5} was in alle scenario's minstens vier keer kleiner wanneer een HEPA-filter aanwezig was. Echter, PM bleef de meest schadelijke stof voor de gezondheid, vóór stikstofdioxide, ozon en formaldehyde.

Sleutelwoorden Fijn stof, Gezondheidseffecten, Filter performantie, binnenluchtkwaliteit, DALYs

I. INTRODUCTIE

Jaarlijks kunnen bijna 2 miljoen sterfgevallen worden toegeschreven aan de inademing van fijnstof en de daaropvolgende gezondheidseffecten [1]. Deze effecten

kunnen variëren van ademhalingsproblemen tot hartfalen, kanker en miskramen. Aangezien mensen ongeveer 90% van de tijd binnen doorbrengen, wordt het grootste deel van deze aan PM-gerelateerde sterfgevallen veroorzaakt door de binnenluchtconcentratie van fijnstof [2]

Verschillende gezondheidseffecten kunnen worden gekoppeld aan verschillende deeltjesgroottes van fijnstof. Hoe kleiner het deeltje, hoe dieper het in het lichaam kan doordringen en hoe meer schade het kan aanrichten. Kleinere deeltjes zijn ook vatbaarder voor reacties vanwege hun grotere oppervlakte per massa-eenheid [3]. De chemische samenstelling van PM speelt ook een rol in het algehele gezondheidsrisico. Vooral PAK's (polycyclische aromatische koolwaterstoffen), die voornamelijk in ultrafijn PM aanwezig zijn, blijken zeer schadelijk te zijn [4]. Over het algemeen heeft natuurlijke PM een grotere diameter dan door de mens veroorzaakte PM [5].

De bestaande indelingen voor fijnstof bevatten alleen PM_{2.5} en PM₁₀ (en af en toe PM₁ of PM_{0.1}). Deze indelingen veralgemenen veel grootte- en samenstellingsgerelateerde gezondheidseffecten en verhinderen de nauwkeurige beschrijving van de mogelijke gezondheidseffecten die een specifieke grootteorde kan veroorzaken. Bovendien zijn de Europese normen voor luchtkwaliteit niet voldoende gericht op de gezondheid bij een bepaalde deeltjesconcentratie, aangezien hun limietwaarden niet overeenkomen met de richtlijnen van de Wereldgezondheidsorganisatie [6] [7].

Om de blootstelling aan schadelijke PM-concentraties te verminderen, kunnen filters worden gebruikt. Vooral HEPA-filters zouden geschikt kunnen zijn voor dit doel, aangezien ze ongeveer 99,97% van de deeltjes met een diameter groter dan of gelijk aan 0,3 μm kunnen verwijderen [8]. Echter, er kunnen vragen worden gesteld over de effectiviteit van deze filters in een echte omgeving. In de laboratoria waar deze filters worden getest, worden verschillende invloedrijke parameters die de binnenlucht PM-concentratie beïnvloeden, genegeerd. Dit kan leiden tot een onterecht gevoel van veiligheid in een echte omgeving, waar deze andere factoren wel een rol spelen. Het verstoppert van de filter kan ook leiden tot een significante toename van de energiekosten en een daaropvolgende verkorting van de levensduur van de filter. Dit beïnvloedt de bereidheid van mensen om een filter in hun ventilatiesysteem te installeren.

Daarom is het doel van deze scriptie om te onderzoeken of filters in staat zijn om de concentratie van fijnstof in een realistische binnenomgeving voldoende te verlagen. Ten eerste werd een nieuwe indeling voor fijnstof ontwikkeld op basis van de toxiciteit van verschillende deeltjes, filterefficiëntiecurves en de luchtdichtheid van het gebouw. Ten tweede werd een filtercomponent ontwikkeld en een bestaand multi-zone model aangepast om fijnstof te omvatten. Ten slotte werden diverse simulaties onder verschillende omstandigheden uitgevoerd om de invloed van verschillende parameters te observeren. De onderzochte parameters omvatten een variërende buitenlucht afkomstig uit verschillende omgevingen, verschillende filters, de invloed van kook-gerelateerde PM, vraaggestuurde ventilatie, het openen van ramen en de levensduur van een filter met het daaropvolgende energieverbruik. Daarnaast werd een kort onderzoek gedaan naar het aantal DALY's (Disability-Adjusted Life Years) per 100.000 mensen als gevolg van PM_{2.5} in verschillende scenario's. Dit werd vervolgens vergeleken met het aantal DALY's afkomstig van andere vervuulende stoffen. Dit resulteerde in de bepaling van de meest schadelijke verontreinigende stoffen wanneer PM buiten beschouwing wordt gelaten.

II. DEELTJESKLASSEN

A. Gebaseerd op toxiciteit

De grootte van een deeltje bepaalt hoe ver de deeltjes het lichaam kunnen binnendringen en welke specifieke organen of systemen kunnen worden beïnvloed. Daarom werden verschillende grenzen bepaald. PM tussen 10 en 5 μm blijft hangen in de longen. Deeltjes tussen 5 en 1 μm kunnen de kleinste delen van de longen bereiken. Een deeltje kleiner dan 1 μm gedraagt zich als een gas en kan daardoor in de bloedbaan terechtkomen en cels- en DNA-schade veroorzaken. Deze groottefractie kan ook invloed hebben op de hormonen. Als een deeltje kleiner is dan 0,5 μm , kunnen het hart en de kransslagaders worden bereikt. Bij een diameter kleiner dan 0,25 μm kan de placenta worden doordrongen. Dit kan leiden tot afwijkingen bij de baby, miskramen of vroeggeboorte. Deeltjes kleiner dan 0,2 μm kunnen de hersenen en het zenuwstelsel bereiken en deze systemen beschadigen [9].

B. Gebaseerd op filterefficiëntie

Elke filter heeft een filterefficiëntiecurve. Niet alle deeltjesgroottes worden even effectief gefilterd door een filter. De MPPS, of de deeltjesgrootte die het meest doordringend is, verschilt voor alle filters. Een grove of middelmatige filter heeft een MPPS van ongeveer 0,2-0,3 μm . Voor een hoge efficiëntiefilter ligt dit tussen 0,12-0,25 μm . Deze groottefracties worden het minst efficiënt gefilterd [10] [11].

C. Gebaseerd op de gebouwschil

Deeltjes met een grootte van 0,2-0,3 μm kunnen gemakkelijker door de gebouwschil dringen dan andere [11].

Tabel 1 Resulterende deeltjesklassen gebaseerd op IIA, IIB & IIC.

Nr.	Deeltjesklassen	Grote diameter
1	PM _{0.01-0.1}	0.01 tot 0.1 μm
2	PM _{0.1-0.2}	0.1 tot 0.2 μm
3	PM _{0.2-0.25}	0.2 tot 0.25 μm
4	PM _{0.25-0.3}	0.25 tot 0.3 μm
5	PM _{0.3-0.4}	0.3 tot 0.4 μm
6	PM _{0.4-0.5}	0.4 tot 0.5 μm
7	PM _{0.5-1}	0.5 tot 1 μm
8	PM _{1-2.5}	1 tot 2.5 μm
9	PM _{2.5-5}	2.5 tot 5 μm
10	PM ₅₋₁₀	5 tot 10 μm

III. SIMULATIE OPSTELLING

A. Filter component

Vervolgens werd een filtercomponent gemodelleerd in Dymola, met gebruik van de Modelica-taal. Dit filtermodel is gebaseerd op de huidige standaard voor hoge efficiëntiefilters: NBN EN 1822-1 (2019) [8]. Deze testmethode biedt een efficiëntiecurve en initiële drukval. De geteste snelheid wordt bepaald door de fabrikant. Aangezien de filterefficiëntie sterk afhankelijk is van de snelheid, is het vergelijken van verschillende filters met verschillende testsnelheden zeer moeilijk. De vochtigheidsvereisten zijn ook vrij breed. Omdat de vochtigheid de bezinkingssnelheid beïnvloedt, kan dit ook tot verschillen leiden. De filter is uitgerust met de bepaalde deeltjesklassen en bijbehorende efficiënties zoals afgeleid uit de efficiëntiecurve van de teststandaard.

B. Multi-zone model

In een volgende stap werd een bestaand multi-zone model, ontwikkeld door dr. ir. architect Klaas De Jonge, aangepast om fijnstof te bevatten [12]. Om dit te realiseren werden de buitenconcentraties aan PM variabel gemaakt en werden vervalsnelheid toegevoegd. Dit model bestaat uit een appartement met 3 slaapkamers en is ongeveer 120 m³ groot. Deze simulatieomgeving kan worden aangepast om verschillende casestudies te onderzoeken.



Figuur 1 Grondplan van het multi-zone model [12].

C. Basisscenario

Als basisscenario is het appartement uitgerust met een D-ventilatiesysteem dat continue werkt. De ramen zijn gesloten en er zijn geen bewoners. De luchtdichtheid is constant. De resultaten van dit scenario zullen dienen als richtlijn voor de verschillende casestudies. De buitenconcentratie aan PM is afkomstig uit Vilvoorde. Alle resultaten worden vergeleken

met de richtlijnen van de WHO voor gemiddelde jaarlijkse concentratie en maximale dagelijkse gemiddelde concentratie.

IV. SIMULATIES

A. Variable buitenconcentratie

De invloed van verschillende omgevingen in België wordt onderzocht op basis van gegevens van irCELLine.be. Naast het basisscenario werden drie verschillende locaties in België geselecteerd. Deze locaties vertegenwoordigen verschillende omgevingen die verschillende grootteverdelingen van fijnstof kunnen bevatten. De onderzochte locaties waren net buiten de ring van Brussel (Vilvoorde), in het stadscentrum van Gent, op de Antwerpse ring en in een meer landelijk gebied in West-Vlaanderen (Moerkerke). Op basis van deze gegevens werd voor elke deeltjesklasse een worstcasescenario aangenomen, aangezien exacte grootteverdelingen sterk plaatsafhankelijk zijn.

B. Gebaseerd op filterefficiëntie

Ten tweede werden drie filtertypen met verschillende efficiënties getest. De efficiënties voor een grof 60% filter en een ePM1 55% filter voor elke deeltjesklasse werden afgeleid van gegevens uit het Out2In-onderzoek van Buildwise [13]. De efficiëntie voor een H13-filter werd afgeleid via de wiskundige benadering van Chen, Ji en Zhao [14]. Deze efficiënties, zoals te zien in tabel 2, werden gebruikt als input voor de filtercomponent. Deze component werd vervolgens in het multi-zone model geplaatst.

Tabel 2 Filterefficiënties voor verschillende deeltjesklassen

	Coarse 60 % filter [%]	ePM1 55% filter [%]	H13 filter [%]
PM0.01-0.1	42.32	53.58	99.947
PM0.1-0.2	42.23	52.33	99.902
PM0.2-0.25	42.08	50.46	99.973
PM0.25-0.3	41.98	49.20	99.999
PM0.3-0.4	41.84	47.33	99.999
PM0.4-0.5	41.64	44.83	99.999
PM0.5-1	41.06	37.33	99.999
PM1-2.5	45.62	36.17	99.999
PM2.5-5	52.89	44.13	99.999
PM5-10	59.55	52.74	99.999

C. Binnenbronnen van fijn stof

In de volgende casestudy verhuist een gezin van vier naar het appartement. Koken wordt verondersteld de meest algemene en hoogste bron van fijnstof te zijn in een binnenomgeving. Daarom is een kookschema ontwikkeld met 3 basismaaltijden als worst case scenario, gebaseerd op de bezetting van dit gezin [15]. Echter, koken is niet gelijk verdeeld over alle deeltjesgroottes. Om de grootteverdeling te bepalen, werd een lognormale verdeling gebruikt met een modusdiameter van 0,04 μm en een standaardafwijking van 0,005 μm . Hieruit bleek dat bijna al het aan koken gerelateerde fijnstof kan worden toegeschreven aan de kleinste fractie van fijnstof. Een dampkamp met een efficiëntie van 70% was ook aanwezig.

Tabel 3 Emissieniveau en duur van de geselecteerde maaltijden [15].

Maaltijd	Gemiddeld emissieniveau [$\mu\text{g}/\text{m}^3$]	Duur [minuten]
Toast	220	3.85
Pasta Bolognaise	1900	28
Kip en groenten- stoofpot	3200	17

D. Vraaggestuurde ventilatie

In tegenstelling tot een ventilatiesysteem dat continu werkt, kan ook een op vraaggestuurd systeem worden onderzocht. In het geselecteerde systeem wordt de luchtstroom aangepast aan de behoeften van de natte ruimtes. Echter, een hogere luchtvochtigheid zorgt voor een snellere bezinking van fijnstof. Dit kan dus een negatief effect hebben op de binnenconcentratie van fijnstof.

E. Raamopeningsschema

Het openen van een raam laat extra fijnstof binnen. Een ventilatiesysteem met filter beïnvloedt deze concentratie niet. Daarom komen alle deeltjes direct de kamer binnen. Om deze invloed te onderzoeken, werd een raamopeningsschema geconstrueerd en werd een raamcomponent gemodelleerd op basis van het onderzoek van Silke Verbruggen [16]

F. Levensduur van de filter

Ten slotte werd een filtermodel ontwikkeld met variabele efficiëntie en drukval. Door de verstopping van de filter dringen er minder deeltjes door het filtermedium en neemt de efficiëntie in de loop van de tijd toe. Echter, de verstopping van de filter verhoogt ook de drukval en de daaropvolgende energie die nodig is om een bepaalde luchtsnelheid te handhaven. Om dit fenomeen te onderzoeken, werden efficiëntiecurves geconstrueerd in Dymola op basis van gegevens (buitenshuisconcentratie en efficiëntietoename) van het Out2In-project [17]. De resulterende massa in de filter werd vervolgens gekoppeld aan de overeenkomstige toename van de drukval.¹ Voor de hoge efficiëntiefilters, die geen onderdeel waren van datzelfde onderzoek, werd aangenomen dat een soortgelijke efficiëntiecurve en evolutie van de drukval van toepassing waren.

G. DALYs

Het aantal DALYs voor PM2.5 in de keuken is onderzocht voor een aantal sleutelscenario's. Het totale aantal DALYs wordt berekend met Eq. (1) en Eq.(2) [18].

$$\text{DALYs}(t) = \int_{t_0}^t \frac{\text{DALYs}}{\text{time}}(t) dt \quad (1)$$

$$\frac{\text{DALYs}}{\text{time}}(t) = \text{EF}_{\text{marginal}} * \text{IR}(t) \quad (2)$$

Met $\text{IR}(t)$ = de opnamesnelheid [kg/s] en EF = de effect factor [DALY/kg]. Om $\text{IR}(t)$ te bepalen wordt Eq.(3) gebruikt.

$$\text{IR}(t) = V_{\text{br}}(t) * C_{\text{exp}}(t) \quad (3)$$

¹ Data afgeleid van het onderzoeksproject Out2In uitgevoerd door Buildwise met de financiële steun van Innoviris Brussel

Met V_{br} = ademhalingssnelheid, waarvoor $16\text{m}^3/\text{dag}$ wordt aangenomen en C_{exp} = bloedstellingsprofiel.

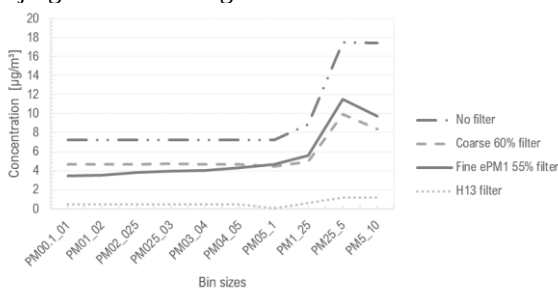
H. Andere vervuilende stoffen

Het multi-zone model omvatte al verschillende andere verontreinigende stoffen zoals VOS (Vluchtige Organische Stoffen), stikstofdioxide en ozon. Op basis van een DALY's berekening kan een beknopte observatie worden gedaan over welke van deze het meest overvloedig aanwezig is in de binnenomgeving wanneer PM wordt uitgesloten.

V. RESULTATEN EN DISCUSSIE

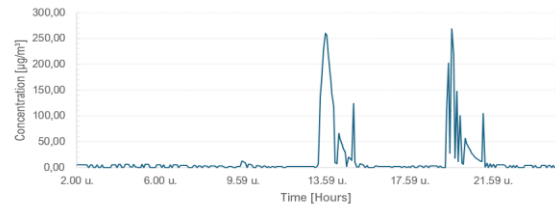
Het simuleren van de binnenconcentratie aan fijnstof voor verschillende buitenconcentraties toonde aan dat de variatie in PM in België vrij klein is. De algehele gemiddelde jaarlijkse concentratie voor elke deeltjesklasse overschrijdt de richtwaarde ongeacht de locatie van de buitenconcentratie gegevens. De dagelijkse maximale concentratie wordt gemiddeld gedurende 90-100 dagen overschreden. Bijgevolg is de binnenlucht in België zonder filter per definitie ongezond. Alleen de concentratie aan PM₅₋₁₀ blijft onder de richtwaarde. Er is echter een verschil in grootteverdeling op basis van locatie. Meer verkeer resulteert in meer fijn PM en een meer landelijke omgeving in meer grof PM. Aangezien de kleinste deeltjes het schadelijkst zijn, kunnen de omgevingen van meest naar minst schadelijk als volgt worden gerangschikt: Antwerpen, Vilvoorde, Gent, Moerkerke. Of anders gezegd, de ringweg is schadelijker dan het stadscentrum en het stadscentrum is schadelijker dan een landelijke omgeving.

Bij het testen van verschillende filters werd een gezonde omgeving alleen bereikt met het H13-filter, zoals te zien is in figuur 2. Bij een grove filter neemt de PM-concentratie aanzienlijk af, maar niet genoeg om gezonde concentraties te bereiken. Het installeren van een ePM1-filter die gespecialiseerd is in kleinere deeltjes zou een goede oplossing kunnen zijn om de concentratie van de meest schadelijke deeltjesgroottes te verlagen tot net boven de doelwaarden.



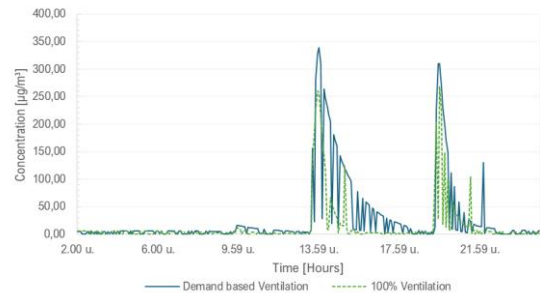
Figuur 2 Vergelijking van de gemiddelde concentratie per jaar voor elke deeltjesklasse in de keuken

De toevoeging van een binnenbron van fijnstof (koken) resulteert in een aanzienlijke toename van de laagste en meest schadelijke deeltjesklasse. Dit is te zien in figuur 3. De HEPA-filter is niet in staat om deze pieken in concentratie te compenseren. Aanvullende bronbeheersing is noodzakelijk om in dit geval een gezonde binnenomgeving te verkrijgen.



Figuur 3 PM concentratie voor de kleinste deeltjesklasse

Zoals verwacht leidt een vraaggestuurd ventilatiesysteem tot een duidelijke toename van de binnenconcentratie van fijnstof. Vooral in de natte ruimtes neemt de PM-concentratie aanzienlijk toe. Het continu werkende ventilatiesysteem zorgt voor een snellere afname van de piekconcentraties.



Figuur 4 Gemiddelde concentratie van PM voor de kleinste deeltjesklasse in de keuken met verschillende ventilatiesystemen

Het introduceren van open ramen heeft ook een aanzienlijke invloed op de binnenluchtkwaliteit. Echter, wanneer er geen filter aanwezig is, heeft het openen van ramen geen invloed of zelfs een positief effect. Maar wanneer er wel een filter aanwezig is, neemt de negatieve invloed van de filter toe naarmate de efficiëntie van de filter toeneemt. Het openen van een raam werkt wel als bronbeheersing voor fijnstof als gevolg van koken. Maar wanneer het te lang geopend blijft, overtreft het effect van de open ramen het effect van bronbeheersing. Deze invloed is te zien in figuur 5 in de keuken, waar het raam continu open staat in de zomer.

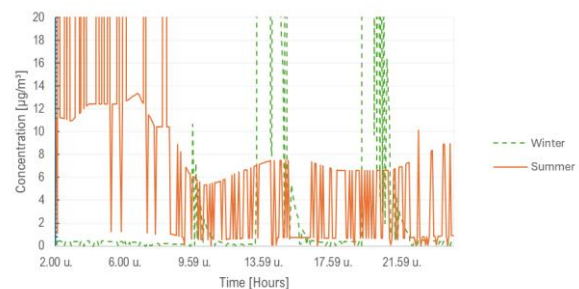
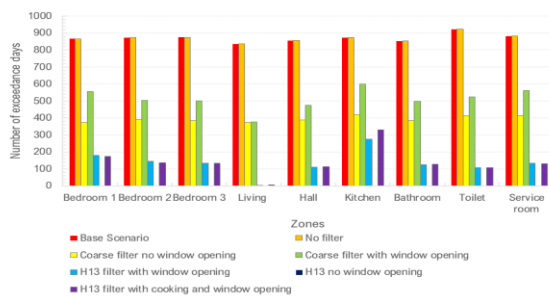


Figure 5 PM concentratie voor de kleinste deeltjesklasse in de keuken tijdens de zomer en winter

De jaarlijkse gemiddelde concentraties voor de kleinste deeltjesklassen in de keuken, die het bovenstaande samenvatten, zijn te vinden in tabel 4. Dit laat zien dat de waarden alleen onder de richtlijnen lagen wanneer er geen ramen geopend waren en er geen fijnstof door koken aanwezig was. Het totale aantal overschrijdingsdagen voor alle kamers in verschillende scenario's is te vinden in figuur 6.



Figuur 6 Totaal aantal dagen boven de norm in verschillende scenario's

Tabel 4 Gemiddelde concentratie per jaar en aantal dagen boven de norm voor PM0.01-0.1 in de keuken volgens verschillende scenario's

	Gemiddelde concentratie per jaar [$\mu\text{g}/\text{m}^3$]	Aantal dagen boven de norm
RICHTWAARDE	3	3
Vilvoorde	7.24	105
Coarse 60% filter	4.66	57
ePM1	3.51	34
H13 filter	0.52	0
Cooking (no filter)	13.25	206
H13 - Cooking	6.55	49
Occupant based ventilation	28.1	338
Window opening (no filter)	7.10	97
Coarse filter – Window Opening	5.74	78
H13 - Window opening	3.19	37
H13 - Window opening - Cooking	6.99	88

Met de variabele efficiëntie filtercomponent kan de levensduur van de filter worden onderzocht. Voor een grove filter wordt de maximale drukval van 150 Pa bereikt na 84 dagen, wat sneller is dan verwacht. Dit kan te wijten zijn aan de vereenvoudigingen die zijn gemaakt bij de ontwikkeling van dit model. Bovendien is het model ontwikkeld met een constante buitenconcentratie aan PM. Na de toevoeging van de massa in de filter op een bepaald tijdstip werd de massaconcentratie gelijkmatig verdeeld over de tien deeltjesklassen om filterefficiëntiecurves te construeren.²

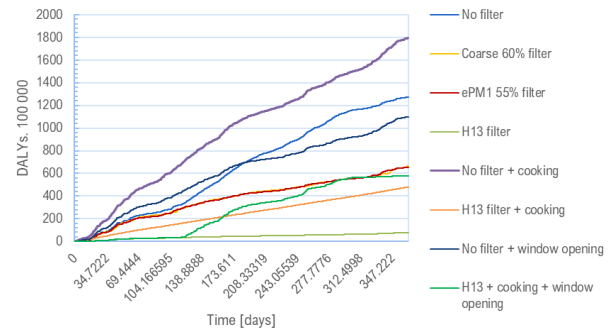
De pieken van de variabele buitenconcentratie zouden kunnen verklaren waarom de resultaten van dit model afwijken van het verwachte resultaat op basis van het Out2In-project [13]. Immers, wanneer de concentratie voor één deeltjesklasse toeneemt, is dit voor alle deeltjesklassen het geval. De werkelijke variabele deeltjesconcentratie in de simulatie is zo hoog dat de filterefficiënties binnen een paar maanden tot 100% stijgen voor een grove filter. Dit verbetert de algehele PM-concentratie aanzienlijk. Echter, aangezien die resultaten niet overeenkomen met de verwachtingen, is er extra onderzoek nodig om de geldigheid van dit model te bepalen.

² Data afgeleid van het onderzoeksproject Out2In uitgevoerd door Buildwise met de financiële steun van Innoviris Brussel

Voor een H13-filter bereiken de efficiënties pas veel later 100%. De zeer hoge efficiëntie vermindert de invloed van de verschillende en variabele deeltjesconcentraties aanzienlijk. Opnieuw verbetert de algehele PM-concentratie door de toenemende efficiëntie. De uiteindelijke drukval wordt overschreden na 238 dagen, of bijna 8 maanden. Dit zou kunnen betekenen dat een filter waarschijnlijk zijn uiteindelijke ,dichter bij 6 maanden dan bij een jaar bereikt, zoals fabrikanten vaak beschrijven.

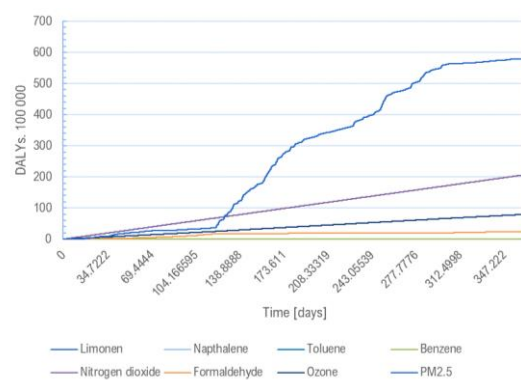
Als de filter niet wordt vervangen wanneer de uiteindelijke drukval is bereikt, neemt het energieverbruik van het systeem met een H13-filter toe tot 6812,5 kWh. Dit staat tegenover 2080,84 kWh als er geen filter aanwezig is. Als de filter wordt vervangen na 238 dagen, is het energieverbruik op dat moment 3685 kWh. Daarom is het tijdig vervangen van de filter essentieel om de energie-efficiëntie van het systeem te behouden. De vervanging van een filter kan aanzienlijk bijdragen aan de algehele verlaging van de energiekosten.

De berekening van DALY's voor verschillende scenario's bevestigt dat elke filter het aantal DALY's aanzienlijk kan verlagen. Zelfs wanneer koken en/of raamopening aanwezig zijn, zijn de verbeteringen die een HEPA-filter veroorzaakt in alle gevallen zeer significant vanuit gezondheidsperspectief. Een vergelijking is te zien in figuur 7.



Figuur 7 Aantal DALY's voor PM2.5 in de keuken.

Het aantal DALY's voor de andere verontreinigende stoffen toont aan dat wanneer koken en open ramen aanwezig zijn, het aantal DALY's voor PM2.5 nog steeds ver boven het aantal DALY's voor andere vervuilde stoffen in hetzelfde scenario uitstijgt. Echter, de verontreinigende stoffen die resulteren in het hoogste niveau van verloren gezonde jaren zijn stikstofdioxide, ozon en formaldehyde, zoals te zien is in figuur 8. Aangezien de neerslagssnelheden voor formaldehyde en stikstofdioxide zelfs lager zijn dan die voor PM, zouden deze deeltjes bovendien zeer gevaarlijk kunnen worden geacht [19].



Figuur 8 Aantal DALY's voor verschillende vervuilde stoffen.

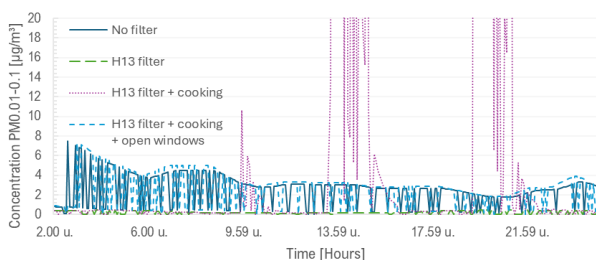
VI. CONCLUSIES

Fijnstof is een bewezen dominante factor bij verschillende gezondheidseffecten. Om een beter idee te krijgen van welke deelgroottes tot welke gezondheidseffecten leiden, werd een nieuwe grootteverdeling ontwikkeld. Deze verdeling toont aan dat deeltjes met een grootte-diameter onder de 1 µm de meeste systemen kunnen beschadigen. Echter, grotere deeltjesgroottes kunnen al aanzienlijke schade aan de longen toebrengen. Daarom is alle fijnstof gevaarlijk, maar fijn PM verdient de meeste aandacht.

Bovendien vertoont de buitenconcentratie aan PM in België slechts een zeer lichte variatie. Vanwege het groter aantal kleinere deeltjes moet een omgeving met veel verkeer vermeden worden. Maar over het algemeen is alle lucht gevaarlijk voor de algemene gezondheid van de bevolking.

Daarnaast kon alleen een H13-filter de binnenconcentratie van PM voldoende verlagen. Om de effectiviteit niet te ondermijnen, moeten ramen zo veel mogelijk gesloten blijven. De lokale pieken in PM als gevolg van koken kunnen niet worden gecompenseerd door de hoge efficiëntiefilter. Om een gezonde omgeving te behouden wanneer mensen maaltijden bereiden, is een vorm van lokale bronbeheersing noodzakelijk.

Het ventilatiesysteem moet ook 100% van de tijd werken om een gezonde omgeving te handhaven. Figuur 9 vat de eerdere bevindingen samen.



Figuur 9 Gemiddelde concentratie PM_{0.01-0.1} in de keuken voor 1 dag.

De levensduur van een filter blijft twijfelachtig. Door de aannames die zijn gedaan, kunnen de resultaten in twijfel worden getrokken. Meer onderzoek is nodig om de geldigheid van de variabele efficiëntiecomponent te bepalen. Niettemin suggereren deze voorlopige resultaten dat filters hun uiteindelijke drukval dicht bij 6 maanden dan bij een jaar na installatie bereiken. In elk geval moeten de vervangingsrichtlijnen van een filter worden aangepast. Als een filter te vroeg wordt vervangen, wordt het voordeel van de toenemende efficiëntie tenietgedaan. En wanneer een filter te lang wordt gebruikt, neemt het energieverbruik aanzienlijk toe.

De invloed van het gebruik van filters op de totale energiekosten is ook zeer significant. Echter, op de lange termijn kunnen vele gezondheidseffecten en de bijbehorende kosten worden vermeden. Daarom verdient een betere binnenluchtkwaliteit toch de voorkeur.

Ten slotte is het aantal DALY's veroorzaakt door PM in een echte omgeving nog steeds significant. Andere verontreinigende stoffen zijn lang niet zo gevaarlijk voor de menselijke gezondheid. De meest gevaarlijke verontreinigende stoffen lijken echter stikstofdioxide, ozon en formaldehyde te zijn.

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Frequently used Abbreviations or symbols

PM = particulate matter

PM_{2.5} = particulate matter with a size diameter $\leq 2.5 \mu\text{m}$

PM_{0.1} = particulate matter with a size diameter $\leq 0.1 \mu\text{m}$

PM₁₀ = particulate matter with a size diameter $\leq 10 \mu\text{m}$

Fine PM = particulate matter with a size diameter between 0.1 and 2.5 μm

Ultrafine PM = particulate matter with a size diameter between 0 and 0.1 μm

Coarse PM = particulate matter with a size diameter between 2.5 and 10 μm

MPPS = most penetrating particle size; particle diameter that is able to penetrate the filter medium most effectively

MERV = minimum efficiency reporting value

PAH's = polyaromatic hydrocarbons

OCP's = organochlorine pesticides

CUP = colloidal unimolecular polymer

DDT = dichloordifenyltrichloorethaan ; a organic chemical insecticide

WHO = world health association

ROS = reactive oxygen species

ER = Estrogen receptor

AR = Androgen receptor

Ahr = Aryl hydrocarbon receptor

1.

Introduction

Each year, exposure to particulate matter is responsible for more than 2 million deaths globally. In addition, more than eighty millions of disability adjusted life-years can be attributed to particulate matter smaller than $2\mu\text{m}$. This shows that not all particulate matter possesses the same level of toxicity. Fine particulate matter has been linked to various health effects that make these particles one of the biggest threats to the general public health of the worldwide population (Lu et al., 2023).

Since people spend more than 90 % of their time indoors, this concentration will have a very important contribution to the overall concentration of PM people are subjected to (M. Kim et al., 2023). The current legislation, however, only entails a limited division of PM. The question could be posed whether or not the current division is sufficient to result in a health-specific approach for particulate matter control.

Indoor particulate matter exposure could possibly be reduced by the use of filters in ventilation systems. This could be a way to decrease the connected health effects and subsequent overall load on the health department. Nevertheless, the use of filter comes with questions regarding their influence in a real-life environment with indoor sources, window opening, different outdoor concentrations and life-span.

This dissertation aims to combine the already existing knowledge with regards to particulate matter related health effects and filters-use in order to form a conclusion on the effectiveness of filters in the indoor space with all the challenges and difficulties this contains.

2.

About particulate matter

2.1. General characteristics

The term particulate matter is used to describe all solid and liquid particles suspended in air with a diameter smaller than $10\text{ }\mu\text{m}$ (Lowther et al., 2020). Particulate matter contains particles that vary in size, chemical and physical characteristics (Kim et al., 2015). PM is currently classified into several categories according to the aerodynamic diameter. PM₁₀ consists of all PM particles smaller than $10\text{ }\mu\text{m}$. PM_{2.5} includes all particulate matter smaller than $2.5\text{ }\mu\text{m}$. And lastly, PM_{0.1} encompasses particulate matter smaller than $0.1\text{ }\mu\text{m}$. The coarse fraction is composed of all particles between 2.5 and $10\text{ }\mu\text{m}$. Everything between 2.5 and $0.1\text{ }\mu\text{m}$ belongs to the fine fraction. Particulate matter particles with a size diameter smaller than $0.1\text{ }\mu\text{m}$ are considered ultrafine particles (Scapellato & Lotti, 2007).

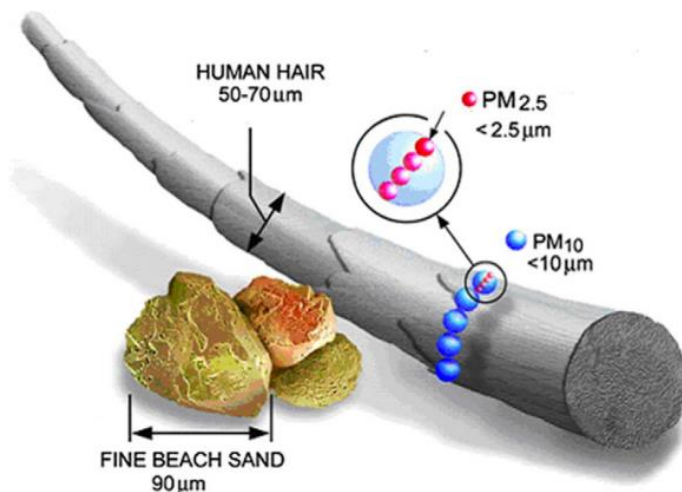


Figure 1: Size comparison of PM_{2.5} and PM₁₀ (Source: Guaita et al., 2011).

PM can consist of various different constituents, dependent on the emission source (Sondreal et al., 2004). Possible constituents include for example sulfates, nitrates and carbon compounds. Furthermore, crustal materials and various metals can be included in PM as well (Tolbert, 2007). The particulate matter concentration also varies significantly over space and time depending on their sources and the weather. The heterogeneity of particulate matter is one of the main reasons it is so difficult to regulate and

describe. When inhaled, particulate matter can be deposited in the lungs and other organs. This can result in various health defects. (Kyung & Jeong, 2020). Toxicity related exposure rates of particulate matter can be categorized by particle size, the chemical composition of PM, the chemicals that are absorbed by the particles and other gaseous pollutants (Scapellato & Lotti, 2007). Varying compositions of PM can therefore account for another level of toxicity and related health effects (Tolbert, 2007).

2.2. Outdoor sources

PM can be formed in two ways. Particles that are directly emitted are called primary PM. Particles that are formed in the atmosphere by reason of chemical reactions belong to the category of secondary PM (Sondreal et al., 2004). Primary particulate matter particles can have a large variety of different sources. In general, they can be divided into two subcategories: Natural and Anthropogenic sources (Mukherjee & Agrawal, 2017).

Natural PM can originate from various natural phenomena like wildfires, wind erosion, volcanoes, and sea spray. Desert sand can also be a large contributor to natural PM (Mukherjee & Agrawal, 2017). Furthermore, bioaerosols can be a part of particulate matter as well. For example, fungi were found in samples near a traffic junction. And bacteria were found at waste treatment facilities (Manojkumar & Srimuruganandam, 2021). Various studies show that on a global scale, particulate matter originating from natural sources exceeds the emission of PM by anthropogenic sources (Mukherjee & Agrawal, 2017).

Anthropogenic sources can be differentiated into particles deliberately produced, for example for certain scratch-resistant coatings and self-cleaning surfaces, or unintentionally produced by-products (Todea et al., 2020). Unintentional emission sources of PM are mainly emissions from traffic or industrial processes, coal combustion, soil resuspension, crustal components, diesel exhaust, mineral matter, secondary inorganic aerosols, biomass burning, organic matter and elemental carbon. Most of these sources are either related to traffic, combustion or mechanical processes. In developing countries other sources can also be observed like domestic cook stoves, power plants and open waste burning (Mukherjee & Agrawal, 2017).

In the case of combustion, we can also distinguish another type of PM. Next to the filterable PM fuel, combustion also emits condensable PM. Filterable PM is filtered just after the combustion process, but condensable PM exist as a gaseous material at that temperature and can therefore escape through the filter to the outside environment. These particles go from gaseous to liquid or solid due to dilution and colling in the plume after emission. Condensable PM accounts for about half of the PM emission caused

by natural gas-fired boilers. These particles belong to the fraction of PM_{2.5}, stay airborne for a long time and travel large distances before deposition. Therefore, they are worth mentioning in this investigation. But since most research surrounding these particles focusses on coal-fired combustion, these particles will not be taken into account in this dissertation. (Feng, Li, & Cui, 2018).

2.3. Indoor sources

2.3.1. Cooking

PM emitted by cooking is the primary source of PM in non-smoking households (O'Leary et al., 2019). Cooking can cause a tremendous increase in PM depending on the kind of activity, heat source and temperature (Vu et al., 2015).

Installing a cooking hood can be very effective in this case, especially when kept operational for the cooking time and an additional 10 minutes. The airflow of the cooking hood can be adapted so it can remove most of the pollutants before mixing with the kitchen air. The capture efficiency of a hood is a function of the airflow rate, height of the installation, the capture volume, and the fraction of coverage it gives the stovetop. Different cooker hoods can cover a wide variety of capture efficiencies (O'Leary et al., 2019).

2.3.2. Printers

Another indoor source for PM is printer equipment, more specifically laser printers. In the study by Akif Ari (2020), an office room where intense printing activities occurred, was tested. When the equipment was working, instant peaks in particulate matter (PM_{2.5}) occurred. After the printing stopped, the concentration rapidly decreased. This indicates the dependence of PM emission on the printers (Ari, 2020).

2.3.3. Others

Furthermore, indoor smoking is also considered to be an important source (Manojkumar & Srimuruganandam, 2021). Other possible indoor sources can be: pet presence, taking a shower, deodorant, carpet walking, candles, air fresheners, vacuum cleaner and household product (Kim et al., 2015).

Evidently, the indoor PM concentration is also dependent on the outdoor concentration that enters the indoor environment via window ventilation or leaks in the building envelop. This is why ground floors usually experience a higher mass and number concentration of PM compared to rooms on a level (Manojkumar & Srimuruganandam, 2021). Hence, because of the endless variation in factors that influence the PM concentration, the indoor concentration varies from room to room.

2.4. Physical characterization

Each source emits a different size distribution and composition of particles. Therefore each particle of PM can have different physical characteristics, like different mass, number and morphology. (Manojkumar & Srimuruganandam, 2021).

Due to their light weight, particles with a smaller size stay in the air longer and take longer to settle (Kim et al., 2015). Ultrafine particles are thus longer present in the air. They also have a much larger surface area per unit mass, which allows them to react with other substances. That's why these particles are often considered the most dangerous (Scapellato & Lotti, 2007). On the other hand, coarse PM will settle a lot faster and have less time to react. In terms of morphology, particles from an anthropogenic source are often rounder and more perfect in shape than naturally emitted PM.

Since the size of the particle is important for the removal efficiency and the correlating health effects of a specific fraction, it is important to know which sources produce the most problematic particle sizes. By analyzing this, removal via source control can be made more efficient (Mukherjee & Agrawal, 2017).

Overall, anthropogenic sources like combustion seem to mainly emit fine particles, while mechanical processes are responsible for larger particles (Mukherjee & Agrawal, 2017). Furthermore, natural dust and salt are usually more present in coarse particles (Karagulian et al., 2015). PM particles that are unintentionally emitted often belong to the ultrafine particle range (Todea et al., 2020). They are usually introduced into the air by secondary aerosols and emissions originating from vehicles and industrial processes (Lowther et al., 2020). Likewise, Biomass burning mainly contributes to PM₁ (Manojkumar & Srimuruganandam, 2021). Many viruses also contribute to the ultrafine particle range. For example, the SARS-CoV 19 virus is on the edge of the ultrafine range and fine range with a size varying from 60 nm to 140 nm (Todea et al., 2020).

2.5. Chemical characterization

The chemical characteristics of PM are dictated by the different kinds and quantities of ions, elements, organic carbon, elemental carbon and polyaromatic hydrocarbons, that are present in the particulate matter particles (Manojkumar & Srimuruganandam, 2021). The chemical content of PM is strongly correlated with the size and surface area of the particulate matter as well. Most of the particulate matter consists of carbon, nitrate, sulphate, ammonium and crustal components.

Coarse particles can contain minerals, metals, organic carbon (originating from soil), road and/or industrial dust, coal, and oil combustion products. Bio-aerosols like pollen, fungi and bacteria can also be present in coarse PM, as well as aqueous droplets and multiple gases, such as ammonia, carbon monoxide and ozone (Scapellato & Lotti, 2007). In the investigation by Manojkumar and Srimuruganandam (2021) the size-segregation of particulate matter in urban and rural areas in India is investigated thoroughly. This showed that particulate matter consisted of elements such as copper, iron, aluminum, lithium, and titanium as well. These elements were mainly present in the larger regions of coarse PM ($PM > 7 \mu m$). And were mainly contributed to the earth crust. Elements represented about 25 % of coarse PM mass, while ions represented 23 % of the concentration (Manojkumar & Srimuruganandam, 2021). Metallic ions were best represented in this size fraction. These ions are produced by traffic emissions, the burning of petrochemicals or the blow up of soil dust (Yang et al., 2023).

Substances that can be found in fine particulate matter are minerals, metals and carbonaceous materials derived from the soil, road dust or combustion processes. Natural fine PM due to volcanoes and oceans can result in nitrates and sulfates. Other components of fine PM can be: bio-aerosols and gases such as formaldehyde, ozone and carbon monoxide (Scapellato & Lotti, 2007). Elements like arsenic, cadmium, chromium, lead, and tin were mainly found in $PM < 1.1 \mu m$ in the research by Manojkumar and Srimuruganandam (2021). They are mostly attributed to anthropogenic sources (crustal, industrial, vehicular, and mixed sources). These different elements were responsible for about 20% of fine PM. Ions also contributed largely to the total fine PM mass (12-57%). Another large contributor to fine particulate matter is polyaromatic hydrocarbon. Benzo[a]pyrene was the largest contributor (23%) to the overall mass of these polyaromatic hydrocarbons in this case. PAHs were mainly concentrated around traffic sites and biomass burning sites (Manojkumar & Srimuruganandam, 2021). Water-soluble ions also contributed more to this fine fraction. These were attributed to traffic, burning of chemicals and dust blow up (Yang et al., 2023). Furthermore, heavy metals are mainly present in fine PM. Cadmium, Copper, Zinc and Lead indicate the emission of particles by the tires of a vehicle. The wearing of the brakes leads to emission of Copper, Lead and Zinc. PM from these sources usually belongs to the larger size regions of fine PM. The third and main cause for heavy metals in particulate matter is vehicle exhaust. These particles are part of the smaller size fractions of fine PM mainly consisting of Chrome, Manganese, Cobalt and Nickel (Zhang, Mao, Zhang, & Wu, 2021).

Ultrafine PM contains carbon, metals and organic components originating from the condensation of volatile emissions from plants, microbes or fuel combustion (Scapellato & Lotti, 2007). The organic component and elemental carbon concentration is higher in comparison to other PM sizes. The

concentration of polycyclic aromatic hydrocarbon is also significantly higher in UFPs. However, the metal content of ultrafine particles is lower than that of coarse or fine PM particles. Heavy metals can also be present in this size fraction (Zhang, Mao, Zhang, & Wu, 2021). Ion concentration was 17% (Li et al., 2003). The ions found were mainly chloride, fluoride, nitrate, sulfate, ammonium, calcium, potassium, and magnesium. From those, ammonium, nitrate, and sulfate (secondary inorganic ions) were formed through chemical reactions. Those three also included 10-20% of the total particulate matter mass. The concentration of ions seemed to be the highest in the winter months. Main sources for ions are traffic, crustal material, biomass and coal burning, secondary reactions and long-range transport (Manojkumar & Srimuruganandam, 2021).

The chemical composition of outdoor and indoor PM is also very different. Figure 2 shows the average ratio of chemical components developed by M. Kim et al. (2023). The biggest difference is that the organic carbon content is higher indoors and nitrate concentration is higher outdoors. In the investigation by Manojkumar and Srimuruganandam (2021), the indoor environment was also evaluated. Usually the ions present are sulphate, sodium, nitrate, and magnesium. Furthermore, the elements that can be found contribute to both the coarse (calcium, chromium, copper and iron) and fine (aluminum, nickel, lead and zinc) fraction of PM. The fine fraction seemed to mainly belong to anthropogenic sources. While calcium seemed to be the overall dominating element in all size fractions. Bioaerosol concentrations were usually higher indoors than outdoors. Additionally, fungal aerosols were of a smaller size in the interior space. Normally, bioaerosols could be contributed to the coarse fraction of PM (Manojkumar & Srimuruganandam, 2021).

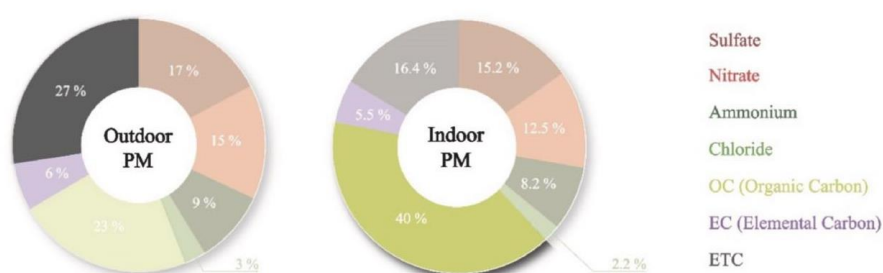


Figure 2 : Average chemical composition of indoor and outdoor PM (Source: M. Kim et al., 2023).

2.6. Guidelines

Following 19th century industrialization, particulate matter came forward as a significant area of interest regarding human health. Nevertheless, the first boundary values were only established by the USA in 1987. The EU followed some years later in 1999. Those first limit values were only subjected to PM₁₀.

The notion of fine PM was only developed later (1997 for the USA and 2008 for the EU) (Manigrasso et al., 2022).

For the last couple of decades, an advised maximum PM concentration on a daily and yearly basis has been issued by the World Health Organization (WHO, 2021). These guidelines have been adapted numerous times over the years, as the health effects of PM became clearer. The evolution of these values can be seen in figure 3. Today the guidelines for PM_{2.5} and PM₁₀ are set on respectively maximum 15 µg/m³ and 45 µg/m³ daily concentration (Manigrasso et al., 2022). Which is a significant decrease, especially for PM_{2.5}.

Pollutant	Averaging Time	2005 Guidelines [µg/m ³]	2021 Guidelines [µg/m ³]
PM_{2.5}	Annual	10	5
	24-hour	25	15
PM₁₀	Annual	20	15
	24-hour	50	45

Table 1: Guidelines for particulate matter according to the World Health Organization (Source: WHO, 2021).

These guidelines are unfortunately not mandatory. Countries that are part of the European Union are however, obliged to follow a European standard, which can be seen in figure 4. These limit values are generally higher than the values described by the WHO, but they do impose a certain boundary.

Pollutant	Averaging period	Concentration [µg/m ³]	Year of legalisation
		µg/m ³	
PM_{2.5}	Annual	20	2020
PM₁₀	Annual	40	2005
	24-hours	50	2005

Table 2: European air quality standards (Source: Directive 2008/50/EC)

This reflects the increasing awareness regarding the health impact of particulate matter on various systems in the body. However, the WHO emphasizes that there does not seem to be a safe threshold for PM concentrations. It is consequently recommended to keep the PM concentration as low as possible. Note that no separate guidelines are imposed for PM₁ or PM_{0.1}. These smaller sizes are included in the larger sizes but seem to provoke more serious health effects. A separate category would therefore be more useful.

These regulations and the increased global awareness have resulted in a significant reduction of the global PM concentrations in the atmosphere over the last couple of decades. In 2017 a 22% decrease in the global concentration was measured in comparison to 20 years before. This reduction can be mainly attributed to developed countries and their regulations and improvements in roads and vehicle maintenance (Mukherjee & Agrawal, 2017). However, in 2020 it was estimated that 90% of the worldwide population still breaths in air that exceeds the limit values of the WHO guidelines (Lowther et al., 2020).

2.7. DALYs

The long-term health effects that can be associated with particulate matter exposure, or exposure to other harmful pollutants, can be quantified by using Dynamic Disability-Adjusted Life Years or DALYs. One DALY presents one healthy life year lost due to either premature death or disability. This allows for the comparison of different health effects with different burdens of disease. The total number of years of life lost due to premature death (YLLs) and the years of healthy life lost due to disability (YLDs) presents the total number of DALYs lost. Usually, the number of DALYs is presented per 100 000 people. In order to calculate the harmfulness of a PM concentration, the number of DALYs caused by every particle fraction needs to be combined. DALYs can be calculated using different methods (Fantke et al., 2019).

In this dissertation, the method proposed by the Energy in Buildings and Communities Program or EBC is used. Here an effect factor is used to quantify the possible health effects of a pollutant and the consequent loss of health. This effect factor is multiplied by a general intake rate that is based on the exposure to the pollutant and the breathing rate that results in intake of that pollutant (Guyot et al., 2024).

$$DALYs(t) = \int_{t_0}^t \frac{DALYs}{time}(t) dt$$

$$\text{With } \frac{DALYs}{time}(t) = EF_{\text{marginal}} * IR(t)$$

$$IR(t) = V_{br}(t) * C_{exp}(t)$$

DALYs(t) as the cumulative DALYs function

IR(t) = intake rate [kg/s]

$$V_{br}(t) = \text{breathing rate over time} = 16 \frac{m^3}{day}$$

$C_{exp}(t)$ = exposure concentration profile

EF = effect factor [DALY/kg]

The high difference in the effect factors for different pollutants versus PM shows the dangerousness of particulate matter. Its effect factor is more than 6 times higher than any of the others.

	EF marginal (median)
	[DALY/kg]
PM2.5	118.6111038
Formaldehyde	18.0350555
Ozone	4.26384255
Nitrogen dioxide	15.7923668
Benzene	0.1268377
Toluene	0.001586365
Naphthalene	0.67175558
Limonene	0.017484409

Table 3: Effect factors used in this dissertation (Source: Fantke et al., 2019).

3.

Health effects

3.1. Overview

Yearly, over 2 million deaths are predicted to be attributable to particulate matter exposure. On average, exposure to PM_{2.5} reduces the life expectancy of a person by about 8.6 months. Globally, about 3% of deaths due to cardiopulmonary reasons and about 5% of deaths as a result of lung cancer can likely be contributed to PM exposure (Kim et al., 2015).

PM enters the body through the respiratory tract and travels to different parts of the lungs, infecting the respiratory system. But respiratory causes can only be held accountable for about 16% of PM related deaths. Heart disease, on the other hand, accounts for about 44% of these deaths. Consequently this is a way more important cause of mortality (Neas, 2000). The remaining 40% of PM related deaths have a various other causes. This chapter provides an overview of different possible health effects due to particulate matter exposure.

3.2. Respiratory diseases

Exposure to PM has been linked to various mild respiratory problems like shortness of breath, coughing and wheezing, chest discomfort and pain (Kim et al., 2015). According to different studies, there is also a strong correlation between an increasing PM concentration and more serious respiratory effects like the reduction of lung function, chronic obstructive pulmonary disease, bronchial asthma, lung cancer and several other respiratory diseases. PM has also shown to increase the mortality rate of pulmonary related diseases (Kyung & Jeong, 2020).

3.2.1. Pulmonary function

Lung function is an important parameter of respiratory health. This pulmonary function increases from birth and starts to decrease once someone reaches his/her mid-twenties. Exposure to PM has shown to accelerate this decrease in adults (Kyung & Jeong, 2020). In the days after a PM peak in an urban area, a worsening of lung function is measured, as well as an increased frequency of other respiratory related symptoms (Scapellato & Lotti, 2007).

3.2.2. Chronic obstructive pulmonary disease

Furthermore, exposure to high concentrations of PM is shown to lead to an increase in the hospitalization and mortality rates due to the lung illness COPD (chronic obstructive pulmonary disease). An increase of $10 \mu\text{g}/\text{m}^3$ in PM₁₀ concentration leads to an increase in the hospital admission rate of 2.7 % and an increase of 1 % mortality. Recent studies show that the PM_{2.5} concentration has a larger responsibility than PM₁₀ in this matter, specifically in the three months prior to the hospitalization (Kyung & Jeong, 2020). Since PM₁₀ and PM_{2.5} were the only divisions used it is difficult to pinpoint whether or not ultrafine particles might be responsible for these results.

3.2.3. Bronchial asthma

Other allergic illnesses like asthma and allergic rhinitis can also be seriously exacerbated due to an allergic inflammatory response. This can increase hyperresponsiveness and an exhaled nitric oxide fraction. This response could aggravate the existing inflammation of the lungs (Li et al., 2009). The lung function of an asthma patient also decreases, just like the lung diffusion capacity. All these symptoms evidently increase asthma related hospital admissions. Again, PM_{2.5} seems to be the biggest culprit. For an increase of $10 \mu\text{g}/\text{m}^3$ in concentration of PM_{2.5} hospital visits increase with 0.49 -0.67 %. This contributes significantly to the overall influence of asthma (Kyung & Jeong, 2020). Surrounding a PM peak in an urban area, an increased frequency of asthma related symptoms and use of asthmatic medications is measured as well (Scapellato & Lotti, 2007). This association is supported by studies that proved a link between the incidence of an allergic disorder and proximity to a large freeway (Li et al., 2009).

3.2.4. Idiopathic pulmonary fibrosis

An international study of 2014 revealed a correlation between PM and the exacerbation of idiopathic pulmonary fibrosis (IPF) in the period till six weeks after the exposure. The PM exposure is related to a decrease in forced vital capacity of the lungs and therefore progresses the illness further. A French study shows an increase in IPF mortality rate of 2-8 times the initial rate when PM concentration is increased with $10 \mu\text{g}/\text{m}^3$ (Kyung & Jeong, 2020).

3.2.5. Pneumonia

Next, exposure to PM can also lead to aggravated pneumonia. The pneumonia related mortality rate in both adults and children increases significantly when a higher PM exposure is registered. In children an increase of $10 \mu\text{g}/\text{m}^3$ for PM₁₀ and PM_{2.5} increases the risk of pneumonia with 1.5% and 1.8% (Kyung & Jeong, 2020).

3.2.6. Lung cancer

Long term exposure to PM is also associated with an increase in the development of lung cancer due to its carcinogenic properties. According to the European ESCAPE study, increasing PM₁₀ with 10 µg/m³ and PM_{2.5} with 5 µg/m³ results in a hazard ratio of 1.22 and 1.18 respectively. The risk for lung cancer incidence was strongly associated with exposure to PM_{2.5}. Only a weak correlation was observed for PM₁₀. This could account for the higher lung cancer incidence in smokers that are frequently exposed to high concentrations of PM_{2.5}. Smokers' lung cancer development also seems to be more effected by fine PM than healthy lungs. The mortality rate of lung cancer patients was significantly higher for both PM_{2.5} and PM₁₀ exposure (Kyung & Jeong, 2020).

3.3. Cardiovascular diseases

Even though respiratory related death is the main focus when looking at PM, cardiovascular diseases play an important role in PM related health issues as well. When PM reaches the bloodstream, the particles can also reach other organs including the heart. If particulate matter is deposited in the cardiac tissue, this can cause arrhythmia and subsequently death (Mukherjee & Agrawal, 2017). Similar to pulmonary diseases, a correlation was found for fine PM especially (Neas, 2000).

Subsequently, an increasing PM concentration also appears to have a strong correlation with coronary artery disease. A correlation with the reduction of the high-frequency component of the heart rate variability in adults and elderly people both with and without this disease was found as well (Scapellato & Lotti, 2007). Particularly short-term exposure to particulate matter smaller than 2.5 µm seems to be responsible. The same conditions can also be linked to an increased carotid-intima-media thickness (Kim et al., 2015).

Lastly, studies have shown that exposure to particulate matter comes with an increased risk for hospitalization due to myocardial infraction and aggravation of congestive heart failure for more vulnerable populations. Particulate matter is responsible for the initiation and promotion of different parameters leading to heart failure and other diseases like atherosclerotic progression (Kim et al., 2015).

3.4. Adult diabetes

By damaging the endocrine system, PM exposure can also interfere with the workings of the hypothalamus-pituitary-thyroid axis. This can result in obesity, insulin resistance syndrome and other metabolic diseases.

3.5. Decline in female fertility and risk of pregnancy loss

A correlation between a damaged endocrine system and a decline in female fertility has also been established (S.-R. Park et al., 2021). Especially during the early stages of the pregnancy (follicular phase) or the menstrual cycle, PM exposure can be very dangerous. Thus, the decline in female fertility is mainly caused by exposure to PM before pregnancy and not during pregnancy. This indicates that the damage is already done before a woman becomes pregnant. PM exposure can also lead to an increased risk for pregnancy loss (S.-R. Park et al., 2021).

3.6. Fetus defects

Long-term exposure of pregnant woman to high PM concentration can negatively influence the fetus in various ways. Exposure during the beginning of the pregnancy can result in changes on the cellular level of the fetus, causing DNA damage and heart and lung defects. The risk for miscarriages also increases. Exposure in the second half of the pregnancy showed a correlation to premature birth or low birth weight. Furthermore, exposure to PM results in an increased risk of respiratory mortality of the fetus and sudden infant death syndrome (Mukherjee & Agrawal, 2017).

3.7. Nerve damage

PM exposure has also been linked to an increased pulse rate. This can indicate problems with the autonomic nervous system (Neas, 2000).

3.8. Vulnerability and recovery

Not every person seems to be equally susceptible to PM related health effects. Since exposure to PM is associated with a longer recovery time and subsequent higher mortality rate, people with a weaker immune system will be more likely to die during recovery (Kyung & Jeong, 2020). A higher PM concentration, therefore, has a larger influence on older people than on younger people (Neas, 2000). Exacerbation of an already existing disease or defect due to particulate matter is also very common. Furthermore, some people are more vulnerable to certain conditions. Overall, children, older people, or people with preexisting conditions, are more susceptible to the risks that come with PM exposure (Kim et al., 2015).

4.

Causes of PM related health effects

The large variety of health effects resulting from particulate matter exposure may be explained by various characteristics of particulate particles. The different defining characteristics can be physical, chemical, catalytical or biological. Subsequently, those different characteristics determine which mechanisms take effect and ultimately which health effects one may experience after exposure (Neas, 2000).

4.1. Characteristics of PM related health effects

4.1.1. Physical characteristics

Physical characteristics of particulate matter that can play an important role in possible health effects include gravimetric mass, particle number, particle size and particle surface area. Those parameters determine how far into the body the particulate matter migrates and eventually the damage the particles can inflict (Neas, 2000).

Firstly, smaller, and lighter particles are able to penetrate deeper into the human body. The size will determine the organs the particulate matter has access to and can subsequently damage. Studies have shown that ultrafine PM frequently penetrates inside mitochondria, causing extensive disruption of the structure. Fine particles show a similar effect but to a significantly lesser extent (Li et al., 2009). Furthermore, due to their small size, ultrafine particles can penetrate into the gas exchange regions of the lungs. This allows for a more effective absorption of toxic chemicals (Novák et al., 2020). Subsequently, smaller particles are more available for reactions due to their larger surface area. This allows them to carry double the amount of organic chemical compounds. Therefore, they have a larger content of bioavailable compounds per mass than larger particles (Lowther et al., 2020).

Subsequently, due to their lower settling rate, the number of fine and ultrafine particles in the air is higher than that of coarse particles. The risk of inhalation is therefore higher (Kim et al., 2015).

4.1.2. Chemical characteristics

The health effects of the particulate matter on the respiratory tract could be attributable to the organic chemical fraction of the particulate matter (Li et al., 2009). The present chemical compounds determine

the reaction possibilities of the particulate matter with another available agent. Chemical reactions in which reactive metal ions form reactive oxygen or reactive nitrogen are called catalytic reactions (Neas, 2000). Tests show that this reaction can mostly be attributed to PAH (Li et al., 2009). Ultrafine PM is shown to contain a large concentration of organic carbon (like those PAHs and polychlorinated dibenzo-p-dioxins) that can carry the organic compounds (Novák et al., 2020).

Additionally, transition metals like nickel, vanadium and copper are suggested to be an important factor in the emergence of acute cardio and pulmonary toxicity. Chemical agents like sulfates, nitrates and organic carbon seem to foreshadow extra negative effects as well, especially on the respiratory tract (Neas, 2000). Subsequently, ozone and nitrogen dioxide were also connected to the exacerbation of idiopathic pulmonary fibrosis (Kyung & Jeong, 2020). Additionally, in case of heart conditions, the risk for cardiac arrhythmia seems to be higher when the particulate matter contains fine sulfate particles (Neas, 2000).

4.1.3. Biological characteristics

When an immunological reaction is caused as a reaction to a biological constituent in the particulate matter, this falls under the category of biological characteristics of PM. Examples can be bacterial endotoxins (respiratory problems), fungal spores, pollen and antigens induced by different insects (Neas, 2000).

4.2. Mechanisms of PM related health effects

The damage done by PM exposure is often much greater than the expected accumulated damage from the particles alone. This indicates that a larger process is initiated once particulate matter enters certain parts of the body. Several mechanisms have been investigated to explain these accumulated effects (Scapellato & Lotti, 2007).

4.2.1. Direct mechanisms

The toxicity of particulate matter is largely dependent on the size of PM. The smallest particles are able to penetrate through the smallest parts of the lungs into the blood. This is considered a direct mechanism. Once in the circulatory system, particulate matter can also affect other organs. Direct contact can cause irritation and affect the accessed organs directly (Scapellato & Lotti, 2007).

4.2.2. Immunological mechanisms

Furthermore, immunological mechanisms can be set in motion by various pulmonary cells when irritated by PM. Those cells then release inflammatory cytokines into the bloodstream. Among the possible released cytokines are platelet activating factor, interleukin-8 and tumor necrosis factor. The last two have been associated with cardiac thrombosis and atherosclerosis (Scapellato & Lotti, 2007). Particulate matter originating from anthropogenic sources seems to possess a stronger potential for the production of inflammatory cytokines. Therefore, anthropogenic PM seems to have a bigger impact on the respiratory and cardiovascular system (Kim et al., 2015). When these inflammatory cytokines lead to PM induced pulmonary inflammation they can also aggravate respiratory symptoms (Kyung & Jeong, 2020). Additionally, the bacterial content of PM can also be correlated to endotoxins that lead to inflammation or lung injury. Metals and elements could act as mediators for airway injuries and inflammation as well (Kim et al., 2015). As airway inflammation in the lungs increases, this results in the release of different toxic substances in the pulmonary system. These instances could lead to pneumonia (Kyung & Jeong, 2020).

The increase of proinflammatory effects can also be due to oxidative stress, generated by chemical compounds and metals present in PM (Li, Xia, & Nel, 2008). Oxidative stress is closely linked to ultrafine PM and their embedded PAHs. The oxidative potential varies over time and temperature, likely due to the variation of UFP and the partitioning of redox cycling PAHs onto the surfaces of the particles. PAHs can be converted to redox cycling quinones by the metabolic system of the lungs or as by-products in the process of diesel combustion (Li et al., 2009). These compounds can act as catalysts in the production of ROS or reactive oxygen species that later result in oxidative stress (Li et al., 2003). These ROS can change the redox equilibrium of certain cells in the immune system in a way that decreases their ability to connect proteins to other cells. This results in an immune reaction that decreases the number of helper cells. The allergic sensitization due to UFP can also lead to the aggravation of allergic inflammatory reactions in the respiratory system, like asthma (Li et al., 2009). Because ultrafine PM has a higher PAH content, the redox activity of ultrafine PM could therefore be between 8.6 and 21.7 times larger than that of larger particulate matter (Li, Xia, & Nel, 2008).

The resulting oxidative stress can generate health effects in the pulmonary alveolar macrophages and bronchial epithelial cells. Low levels of oxidative stress seem to activate antioxidant defense mechanisms, which can carefully prevent oxidative stress with antioxidant enzymes and proteins as well as low-molecular-weight scavengers.

However, higher levels can lead to proinflammatory effects. In this case, this initial defense mechanism may end up more damaging when higher levels of oxidative stress are present (Li, Xia, & Nel, 2008).

Transition metals are also considered to enlarge the production of reactive oxygen species in vivo (D'Evelyn et al., 2021).

4.2.3. Neural mechanisms

If the autonomic nervous system reacts to pulmonary irritation due to PM, this is considered a neural mechanism. When lung fibers detect irritation, the signal is transmitted to the respiratory centers. This can ultimately lead to decreased function of the nervous system which can be linked to a decrease in heart rate variability and increase in heart rate (Scapellato & Lotti, 2007).

4.2.4. Endocrine mechanisms

Organic compounds present in PM can cause different endocrine disrupting potentials. This disruption is usually implemented by pollutants that interfere with the signaling of nuclear receptors, like the estrogen receptor (ER), androgen receptor (AR) and aryl hydrocarbon receptor (AhR). By damaging those receptors, the particulate matter obstructs the transcription of the endocrine system, causing numerous adverse health effects. This can ultimately lead to problems with the nervous and immune system or even impede the reproduction system. Various pollutants have been shown to possess the potential to interact with the receptors, for example PAH's, OCP and CUPs. An increased concentration of those pollutants in PM (mainly in fine and ultrafine PM) points to a larger risk for such adverse health effects. The mechanisms that disrupt the endocrine system are called after the receptors whose signal they interfere with: acryl hydrocarbon mediated activity, estrogenicity, genotoxicity and anti-androgenicity (Novák et al., 2020).

PM₁ was responsible for more than 75% of the particle fractions presenting cases of AhR-mediated activity. The compounds responsible were PAHs and organic carbon. (Novák et al., 2014).

Estrogenicity, however, was mainly driven by PAHs during the winter and by pesticides during the summer (Novák et al., 2020). Estrogenicity has been proven to be mainly responsible for problems in the reproductive system and the formation of cancerous cells (Novák et al., 2014).

In case of anti-androgenicity a correlation to chemicals in PM was observed. Anti-androgenicity was caused by the largest and smallest particle size fractions. There also could be a link with oven emitted PM and diesel particles (Novák et al., 2014).

Genotoxicity can occur both in vivo and in vitro and is induced by oxidative stress from metal particles. It is usually induced by the smaller size fractions of particulate matter. This potential was correlated with PAHs, DDT and organic carbon (Novák et al., 2014). Transition metals are also linked to certain genotoxic effects (e.g. leading to pulmonary injury) (Kim et al., 2015).

4.2.5. Damaging stem cells

By damaging endometrial stem cells, various favorable functions of the endometrial system are prevented. The investigation of Park et al. (2021) showed that particulate matter (especially when containing diesel exhaust or heavy metals) attacks the SERPINB2 gene and by doing so, obstructs the self-renewal, transdifferentiation and migration of endometrial stem cells, both outside and inside the womb. The self-regenerative capacity of the endometrium is vital to each menstrual cycle, in which the endometrium is build, demolished, and rebuild every cycle. This is necessary for the reproductive system. The endometrial stem cells form the base layer of this endometrium and stay in place every month to regenerate the endometrium. The endometrium is the tissue in which the embryo will implant itself and grow during the pregnancy. By damaging the endometrial cells, their deficiency limits the cyclic regenerative capacity of the endometrium. This consequently results in a higher pregnancy failure rate. (Park et al., 2021)

5.

Air filters

5.1. Protective measures against particulate matter exposure

To prevent exposure to PM, protective masks can be used. This is the most common and simple way to protect yourself from particulate matter. In areas with a high particulate matter concentration, this is already recommended. Unfortunately, these masks can cause discomfort when they are worn for a longer period of time. Furthermore, wearing protective masks causes the accumulation of carbon dioxide on the inside of the mask and can also result in a decline in the inhalation of oxygen. This can complicate breathing for people with preexisting respiratory problems or low lung function (Kyung & Jeong, 2020). Wearing a face mask is therefore only a temporary solution. A more permanent solution is the decrease of both the indoor and outdoor concentration of PM.

To reduce the concentrations of PM in the indoor environment, indoor air filters can be used. Those filters can be installed as part of a mechanical ventilation system. The main goal of a ventilation system is to maintain oxygen levels in the air and restrict carbon dioxide levels. Furthermore, ventilation is used to maintain a certain degree of relative humidity and temperature. 'Secondary', a ventilation system can be equipped with a filter to prevent damaging outdoor particles from entering the indoor environment. These filters are not able to dilute PM and other pollutants originating from indoor sources. If, however, such an effect is required, an air purifier can be used. The same filters can be installed both in a mechanical ventilation system and an air purifier.

Rather than removing previously emitted particles, source control is an even better option. Preventing the particles from entering the air remains the most effective way of particulate matter control. Even though a filter can remove certain particle sizes very efficiently, the remaining concentrations could still be higher than the standards. (Dubey, Rohra, & Taneja, 2021).

5.2. Filter classification

5.2.1. European standards

In Europe, filters for ventilation systems have previously been classified according to 2 standards. The standard NBN EN 779:12 is responsible for the first classification of the filter efficiency classes. This includes coarse, medium and fine filters. Coarse and medium filters (G1 to G4, M5 and M6) are usually used as pre-filters before fine filters (F7-F8-F9) in HVAC systems of residential buildings or workplaces. The filters are ranked according to their filtration efficiency (how efficient certain particles are able to penetrate through the filter medium). In these standards, classes are defined based on the fractional filtration efficiency at a particle size of 0.4 μm . (Todea et al., 2020).

Group	Class	Final test pressure drop Pa	Average arrestance (A_m) of synthetic dust %	Average efficiency (E_m) of 0,4 μm particles %	Minimum Efficiency ^a of 0,4 μm particles %
Coarse	G1	250	$50 \leq A_m < 65$	-	-
	G2	250	$65 \leq A_m < 80$	-	-
	G3	250	$80 \leq A_m < 90$	-	-
	G4	250	$90 \leq A_m$	-	-
Medium	M5	450	-	$40 \leq E_m < 60$	-
	M6	450	-	$60 \leq E_m < 80$	-
Fine	F7	450	-	$80 \leq E_m < 90$	35
	F8	450	-	$90 \leq E_m < 95$	55
	F9	450	-	$95 \leq E_m$	70
^a Minimum efficiency is the lowest efficiency among the initial efficiency, discharged efficiency and the lowest efficiency throughout the loading procedure of the test.					

Figure 3: Classification of filters according to NBN EN 779 : 2012.

In the 1990s the forming of a new standard was motored by the development of the new ULPA filters and HEPA filters. Those technical more advanced and more effective filters could not be classified according to the old standards. Therefore, it was investigated whether or not other existing standards could be used to characterize these filters. Using the British Standard BS 3928 was initially the plan, since that standard was already widely adapted in other countries and in the Eurovent 4/10 standard (Wepfer, 1995).

The Eurovent (Europe's Industry Association for Indoor Climate, process cooling and food cold chain technologies) 4/10 2005 guideline, was compiled as a recommendation for minimizing the operational costs of a filter system. In this guideline, particles are counted from 0.2 μm till 1 μm , since in normal

ventilation air most of the particles present are smaller than 1 μm . The test procedure estimates the airflow and measures the temperature, relative humidity and pressure drop, based on the NBN EN 779 standard (Eurovent, 2005).

However, it was established that a whole new measuring method was necessary to achieve consistent and logical systems for both HEPA, ULPA filters and cleanrooms. A testing method was needed that could test different filters with different efficiencies and sensitivities. Testing very large efficiencies requires a sensitivity level that is much higher than the testing of average filters requires. The filter testing method needed to be able to handle both. This new method also needed to account for leaks in the different efficiencies and a better correlation between different test setups to avoid errors (Wepfer, 1995).

In this new standard, called NBN EN 1822 : 1998, 3 testing methods were introduced. The first is a test to evaluate the MPPS or most penetrating particle size of a flat filter medium at the nominal medium face velocity. The second test procedure measures the efficiency and pressure drop at a specific nominal volumetric air flow and at a specific MPPS for a finished filter medium. The third procedure consists of leakage testing. The standard evolves around minimal integral MPPS efficiencies (and maximal efficiencies for the higher performing filters) (Wepfer, 1995). This MPPS represents the particle size that corresponds to the lowest filtration efficiency for the tested filter. The standard introduces two extra groups of filters, H (HEPA) and U (ULPA) filters, with different subclasses. The efficiencies of those groups are higher than the efficiencies of coarse, medium and fine filters.

Filter class	Overall value		Local value ¹⁾²⁾	
	Efficiency (%)	Penetration (%)	Efficiency (%)	Penetration (%)
H 10	85	15	---	---
H 11	95	5	---	---
H 12	99,5	0,5	---	---
H 13	99,95	0,05	99,75	0,25
H 14	99,995	0,005	99,975	0,025
U 15	99,999 5	0,000 5	99,997 5	0,002 5
U 16	99,999 95	0,000 05	99,999 75	0,000 25
U 17	99,999 995	0,000 005	99,999 9	0,000 1
1) see 6.5.2 and prEN 1822-4				
2) local values lower than those given in the table may be agreed between supplier and purchaser				

Figure 4: Classification of HEPA and ULPA filters according to NBN EN 1822-1.

5.2.2. American standards

In the USA the ASHRAE 52.2 “*Method of Testing General Ventilation Air-Cleaning Devices for Removal Efficiency by Particle size*” norm is used. This standard, first developed in 1999, defines different MERV classes based on a wide range of particles efficiencies varying in size from 0.3 μm to 10 μm . Efficiency curves are presented for clean, slightly dusted and more heavily dusted filters. The MERV or minimum efficiency reporting value is determined by taking the smallest efficiency value in each of 12 tested size ranges. Additionally, it takes the average for small, medium and large sizes. Small particle (0.3 μm to 1 μm) removal efficiencies are given by E1, medium particles (1 till 3 μm) by E2 and large particles (3 till 10 μm) by E3. The combination of these 3 values determines the MERV for a filter. This table has been revised and updated over the years. Filters above MERV 16 are considered HEPA Filters and follow the performance evaluation by the current European standard. Recently, a particulate matter efficiency calculation was added to the MERV standards in the form of PM_{52.2} efficiency values. A flat distribution is used as the average of distributions of different kinds of air that can pass through a filter media. $PM_{1_{52.2}}$, $PM_{2.5_{52.2}}$ and $PM_{10_{52.2}}$ therefore correspond to PM1, PM2.5 and PM10 (Owen & Kerr, 2020).

5.2.3. Current standards

The older European standards rarely test the fractional deposition efficiency curve of full-scale filters in the case of nanoparticles. Additionally, both Europe and America used different standards. In order to create a common testing and classification system, a new standard was developed in 2017. This standard is used up until today. Coarse filters are classified according to NBN EN ISO 16890 and the high efficiency filters are classified via the NBN EN ISO 29463 standard.

In the coarse standard NBN EN ISO 16890 particle sizes are tested from 0.3 μm to 10 μm . An airflow between 0.25 m^3/s and 1.5 m^3/s is recommended. Furthermore, the data evaluation and the filter classification schemes underwent some changes. Coarse filters are now assigned to four categories depending on the filtration efficiencies for different groups of PM. The possible categories are coarse, ePM1, ePM2.5 and ePM10. Every filter that is not able to achieve a filtration efficiency of 50 % in case of PM10 was allocated to the coarse class. The other categories represent the filter efficiencies for a particle range between 0.3 μm and the respective upper limit and are reported as a percentage with 5% increments up till 95% (Todea et al., 2020).

Group designation	Requirement			Class reporting value
	ePM ₁ , min	ePM _{2,5} , min	ePM ₁₀	
ISO Coarse	—	—	<50 %	Initial grav. arrestance
ISO ePM ₁₀	—	—	≥50 %	ePM ₁₀
ISO ePM _{2,5}	—	≥50 %	—	ePM _{2,5}
ISO ePM ₁	≥50 %	—	—	ePM ₁

Figure 5: Filter groups according to EN ISO 16890-1 2017.

For the high efficiency filters the NBN EN ISO 29463 is used. However, at European level, the first part of this standard is replaced by the updated NBN EN-1822-1 standard from 2019. Here, the classification system was adjusted slightly. The filters were divided into 3 groups (instead of 2) : EPA filters, HEPA filters and ULPA filters.

Efficient particulate air filter or EPA filters are divided into three classes: E10, E11 and E12. These filters have the lowest efficiency regarding blocking both fine and ultra fine particles. HEPA filters or high efficiency particulate air filters are a lot more efficient. They are divided into 2 classes: H13 and H14. However, the most efficient filters for ultra fine particles are ULPA filters, or Ultra Low Penetration Air filters that consist of 3 different classes, again according to ascending efficiency. These are U15, U16 and U17.

Filter Group Filter Class	Integral value		Local value a b	
	Efficiency (%)	Penetration (%)	Efficiency (%)	Penetration (%)
E10	≥ 85	≤ 15	__c	__c
E11	≥ 95	≤ 5	__c	__c
E12	≥ 99,5	≤ 0,5	__c	__c
H13	≥ 99,95	≤ 0,05	≥ 99,75	≤ 0,25
H14	≥ 99,995	≤ 0,005	≥ 99,975	≤ 0,025
U15	≥ 99,999 5	≤ 0,000 5	≥ 99,997 5	≤ 0,002 5
U16	≥ 99,999 95	≤ 0,000 05	≥ 99,999 75	≤ 0,000 25
U17	≥ 99,999 995	≤ 0,000 005	≥ 99,999 9	≤ 0,000 1
^a See 7.5.2 and EN ISO 29463-4. ^b Local penetration values lower than those given in the table may be agreed between supplier and purchaser. ^c Group E filters (Classes E10, E11 and E12) cannot and shall not be leak tested for classification purposes.				

Figure 6: Classification of EPA, HEPA and ULPA filters according to EN 1822-1 2019.

The first part of NBN EN ISO 29463 includes a similar classification system for high efficiency air filters based on efficiency and penetration. Figure 7 contains a comparison between the two standards.

EN 1822-1			ISO 29463-1:2017		
Filter class and group	Overall value		Filter class and group	Overall value	
	Efficiency (%)	Penetration (%)		Efficiency (%)	Penetration (%)
E10	≥ 85	≤ 15			
E11	≥ 95	≤ 5			
E12	≥ 99,5	≤ 0,5			
H13	≥ 99,95	≤ 0,05			
H14	≥ 99,995	≤ 0,005			
U15	≥ 99,999 5	≤ 0,000 5			
U16	≥ 99,999 95	≤ 0,000 05			
U17	≥ 99,999 995	≤ 0,000 005			

^a For group H filters local penetration is given for reference MPPS particle scanning method. Alternate limits may be specified when photometer or oil thread leak testing is used.

Figure 7: EN 1822-1 versus EN ISO 29463-1:2017 (Source: NBN EN 1822-1 : 2019).

5.3. Filtration efficiency

The removal efficiency of a filter is influenced by a few different factors. The most important ones are particle size, charge, and shape. To remove pollutants, filters use 4 different mechanisms. To remove the smallest particles diffusion is used. Due to Brownian motion, very small particles move in random zigzag motions. Those particles will divert from the flow path and as a result, collide with the filter medium. For the removal of the largest particles, a filter relies on interception, inertial impaction and sieving (Lowther et al., 2020). Inertial impaction happens when particles are too heavy to follow the streamlines around a filter fiber. Their inertia makes them collide with the filter fiber instead of going around. This mechanism therefore has an increasing share in the removal efficiency for increasing particle sizes. Interception will occur when a particle with a certain diameter, is collected by the fiber because it does not obtain a minimal distance of half of the diameter between the flow and the fiber surface.

Due to the different mechanisms of a filter, there is a difference in the removal efficiency for particles with different sizes. Those mechanisms result in a typical V-shape a fractional filter efficiency graph like in figure 8. The particle size that is most difficultly filtrated is defined as the MPPS or the Most Penetrating

Particle Size. The position of this MPPS and efficiency for every particle size is different for every filter media and geometry (Todea et al., 2020).

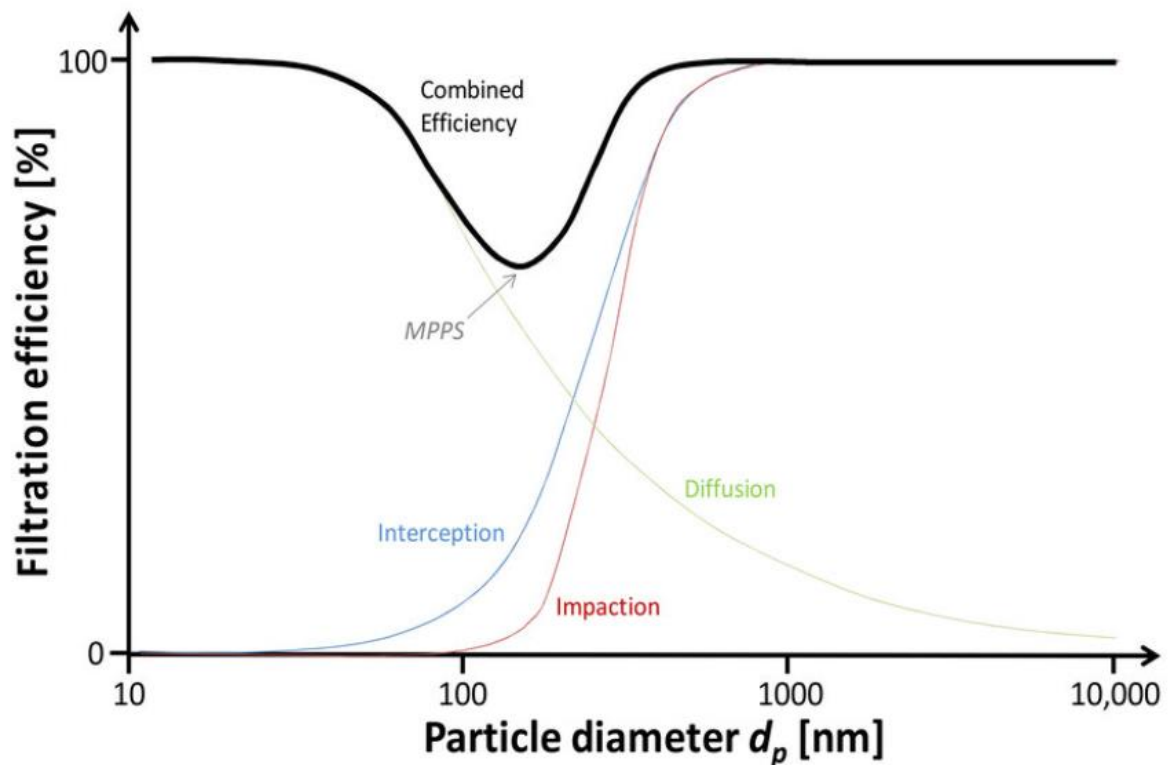


Figure 8: Fractional filter efficiency graph for a typical filter (Source: Todea et al., 2020).

5.3.1. Filter types

Filter structures exist in different types. Overall, filters can be divided into two large groups based on the air flow characteristics. When the airflow enters the filter on one end and goes out the other, a plate-type filter was used. Subcategories are the separator-plate filter, the pleat-plate filter, and the V-shaped filter. The first two have a similar shape, but the pleat density of the pleat-plate filter can be higher. The pleat-plate filter therefore has a higher filter area. The V-shape filter consists of different straight parts that are arranged in V forms in order to increase the filter area even more. The second type are cylinder type filters. Here, dusted air is sent through the filter from different directions. The clean air comes together enclosed between different parts of the filter medium. Depending on shape, the subcategories are cylinder filters and prism filters. The different types of filters can be distinguished in Figure 9. The shape of the filter can thus have a significant impact on the filtration area, airflow rate and subsequent efficiency (W. Zhang et al., 2022).

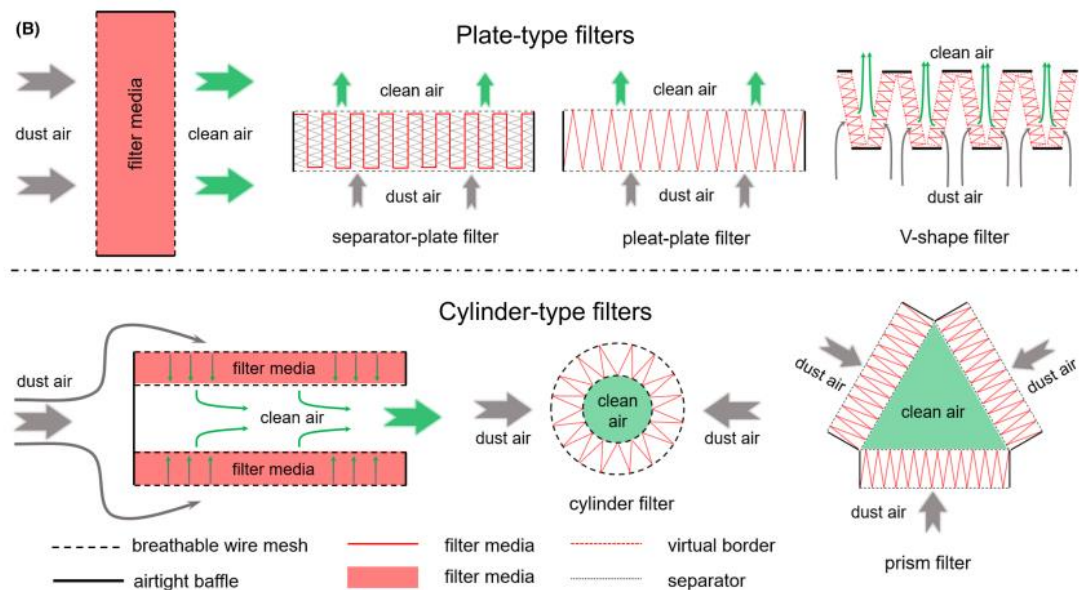


Figure 9: Different filter structures (Source: W. Zhang et al., 2022).

5.3.2. Filter materials

Traditional filters are made of glass fiber. Their capturing efficiency mainly depends on mechanical filtration. However, due to their high initial pressure drop and the fact that the material cannot be degraded in a natural way, different alternatives have been developed.

Fibrous filter media that consist of polypropylene fibers and have static electricity on the surface or inside of the filter are called electret media. These filters contain an electrostatic filtration effect. In comparison to the traditional filter, they have a lower initial resistance and a higher filtration efficiency. During dust loading this effect decreases till it is no longer present (W. Zhang et al., 2022).

The use of nanofiber filter media for indoor pollution control is also being investigated. Due to their lower initial pressure drop, the overall operational cost for these filters would be lower. These filters could effectively control particulate air pollution in the indoor environment but have a shorter lifetime than the traditional glass fiber filters (Xia & Chen, 2021). These filters are made through electrospinning. Electrospinning provides a flexible structure and a functional design that is obtained via nanofiber nets that are spun from a polymer material. The choice for nanofiber can result in certain added qualities of the filter medium, like flame-retardancy or high-strength. Nanofiber filters are lightweight and better for the environment. The fiber diameter is in general $<1\mu\text{m}$ and in some exceptions $<0.1\mu\text{m}$. The low airflow resistance is the result of a non-zero velocity boundary layer in case of airflow through the filter (X. Zhang et al., 2022).

Similar to the nanofiber filters, PTFE or Poly Tetra Fluoro Ethylene filters have a lower initial pressure drop compared to traditional glass fiber filters. However, their resistance grows a lot faster than in traditional glass fiber filters. This could result in a greater energy use during their life span (W. Zhang et al., 2022).

Lastly, in order to improve the energy consumption of PTFE media, composite PTFE filters were developed. The composite filter was created by replacing the windward material with an efficient media. Since this windward base material of the PTFE filter contributes very little to the capturing efficiency, changing this material could decrease the number of particles that are deposited on the PTFE membrane itself. Possible composites that exist are melt-blown media with electrospinning media, electret media with electrospinning media, electret media with PTFE media, multilayered electret media and multilayered electrospinning media (W. Zhang et al., 2022).

6.

Influential factors for the indoor PM concentration

6.1. Relative Humidity

As stated previously, smaller particles stay airborne longer than bigger particles. Very large particles fall to the ground almost immediately and are therefore inhaled less. But next to size there are also other factors that can influence the deposition rate of an PM particle. Every PM particle has a different composition at hygroscopic level depending on size and exact chemical composition. When the humidity level of an indoor environment is higher (>50%), a significant decrease in PM was measured. This can be explained by the absorption of moisture in the air by small PM particles. The combined mass of the particle and the absorbed moisture increases and therefore improves the sedimentation rate of the particle. Small, damaging particles can be removed from the air a lot quicker when there is enough moisture. This increase in settlement rate continues till about 70% relative humidity. Everything above 70% does not seem to have a significant effect. In larger particles the gravity and the high chance of interception with other particles have a higher influence on the sedimentation than moisture. The influence of relative humidity in those particles is therefore minimal. The deposition of larger particles will continue at a similar rate even when humidity is below 50%. However, the settlement rate of particles with a diameter of around 0.5 μm seems to be low under all humidity conditions. (M. Kim et al., 2023). Even though a relative humidity of 70% is desirable for PM, the danger of mold growth and condensation makes a high humidity too risky. Subsequently, it is favorable to pursue a humidity rate of 50 percent in combination with an airtight building envelop. Additionally, it would be beneficial to increase the humidity rate temporally at certain occasions, when indoor concentrations of PM are high (M. Kim et al., 2023).

6.2. Airtightness

Furthermore, the airtightness of a building shell can also play a significant role in the humidity level in the inside environment. A higher airtightness prevents moisture from leaving the building. This results in a higher relative humidity indoors. Therefore, a higher airtightness correlates with a lower PM concentration. However, a high air change rate makes it impossible to maintain a larger moisture level than the outdoor environment (M. Kim et al., 2023). On the other hand, a larger difference in the humidity conditions between the indoor and outdoor environment will result in a greater difference in the vapor

water pressures. This encourages an active exchange of water vapor and containing pollutants through the cracks and gaps of the building shell. Hence, when a building has a low airtightness, PM and other pollutants can travel relatively unobstructed. This results in a very similar PM concentration both indoors and outdoors (M. Kim et al., 2023).

6.3. Varying outdoor PM concentration

The PM concentration on a location depends on the nearby sources, topography of the site, the season and the weather conditions. Large variations in size distribution and components have been reported in different areas around the world (Scapellato & Lotti, 2007). Meteorological factors that could explain this variation are atmospheric pressure, wind, relative humidity, temperature and dispersion (Mukherjee & Agrawal, 2017).

Generally, there is a seasonal variation in the PM concentration. In the winter, the lower temperature, low relative humidity and the calm weather conditions obstruct the movement of the air. Therefore, the PM particles stay nearby for a longer period of time, resulting in a generally higher PM concentration. In summertime, a higher relative humidity causes PM particles to enlarge and aggravate. This results in faster precipitation of the particles and a decreased PM concentration. However, in countries containing desert areas the opposite trend is apparent. Due to desert storms, the summer tends to have the largest concentration of particulate matter in the air for these regions. Long- range transport of fine dust particles can also result in locally higher concentrations. Factors that account for this effect are mainly wind direction and velocity. A higher wind velocity also correlates with a better dispersion of PM in the atmosphere and subsequent lower concentrations (Mukherjee & Agrawal, 2017).

6.4. Window opening

Another large contributor to the indoor PM concentration is the frequency of window opening. Opening windows can have a positive impact on the indoor air quality. By removing CO_2 and letting in O_2 , the air quality improves notably. Therefore, opening windows can provide a natural ventilation. If not through the opening of windows, a ventilation unit should be used to refresh the air. Using natural ventilation can improve the energy efficiency of a building. The influence on the indoor concentration of pollutants is rarely examined.

The necessity of looking at particulate matter when natural ventilation is in place is highlighted in the research of Belias & Licina (2023b). Usually, when looking at the natural cooling potential of a building, only humidity and temperature are considered. In this research, the influence of the outside

concentration of pollutants and noise pollution on the duration and frequency of natural ventilation, is investigated. Depending on the exact location, the consideration of pollutants can reduce the natural ventilation potential up to 94%. This shows that opening a window can have a significant influence on indoor pollutants like PM. Follow-up research also showed that, when using a boundary concentration of different pollutants as restrictive parameters, PM2.5 and PM10 are the most restrictive in all locations. This is opposed to O_3 and NO_2 , that are only defining in specific location types. SO_2 can even be neglected generally (Belias & Licina, 2024).

Furthermore, by opening several windows for a certain amount of time, a high air change rate can be obtained. This prevents a high indoor humidity level, which can cause a slower deposition rate for smaller particles. On the other hand, it also consistently levels the indoor PM concentration with the outdoor PM concentration. This results in a significant increase of PM when a filter medium is present.

6.5. Filter use

The indoor PM concentration is largely dependent on the presence of an air filter. This filter can be part of a mechanical ventilation system or an individual air cleaner. Parameters that determine the efficiency of these filters include the filter type, machine number, airflow speed, the operational time and the specific conditions of the filter (old vs new) (X. Zhang et al., 2022).

6.5.1. Filter type

All types of filters, with different efficiencies can be used for air purification. In the corresponding standard of each filters, the required filter efficiencies for each type of normalized filter category are given. Specific removal efficiencies can be obtained via testing or via an analytical formula.

According to the research by W. Zhang et al. (2022), the filter efficiency for a certain particle size can be defined as follows:

$$\eta(d_p) = 1 - P(d_p) = 1 - \frac{C_{out}(d_p)}{C_{in}(d_p)}$$

With $C_{out}(d_p)$: Outlet particle concentration in $\frac{\#}{l}$

$C_{in}(d_p)$: Inlet particle concentration in $\frac{\#}{l}$

$\eta(d_p)$: Removal efficiency for a particle diameter d_p

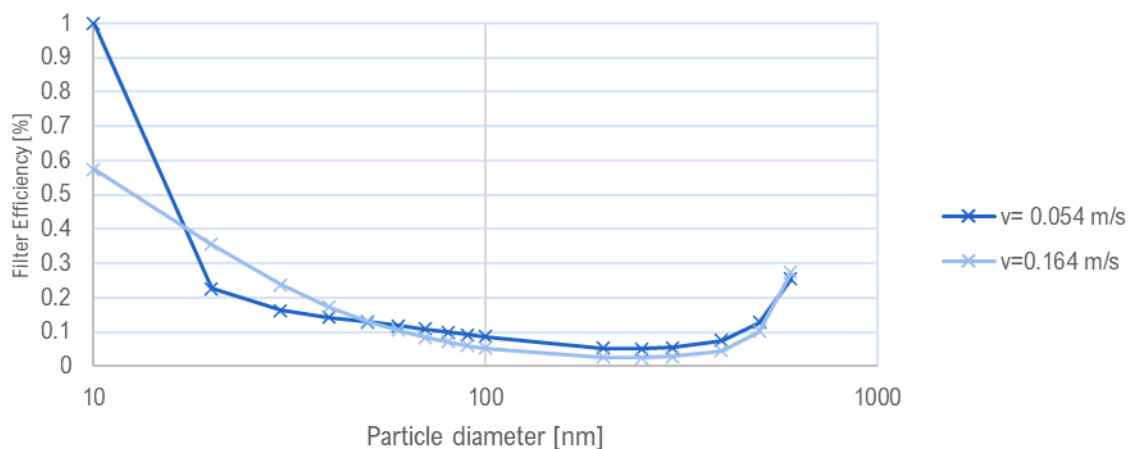
$P(d_p)$: Penetration for a particle diameter d_p

In the research by Chen, Ji and Zhao (2022), size-dependent filtration efficiencies of different kinds of commercially available filter media belonging to different classes and materials, were tested. The tested filters ranged from the best ultrafine particle filter classes to mediocre coarse filters. The filters were tested at two different air flow velocities: 0.164 m/s and 0.054 m/s. The research focusses specifically on the ultra-fine particle sizes, since those are the most dangerous particles due to their large surface area. The removal efficiency was tested for particles with diameters ranging from 0.0146 μm or 14.6 nm to 0.6153 μm or 615.3 nm. To construct the graphs with a logarithmic scale, polynomial equations were used. This was combined with polynomial fitting to find the best fitting equation (C. Chen, Ji, & Zhao, 2019). This equation can be found below.

$$\log[\eta(d_p)] = \text{Intercept} + B_1 * (\log d_p)^1 + B_2 * (\log d_p)^2 + \dots + B_n * (\log d_p)^n$$

With d_p = the size diameter for which one wants to know the efficiency

Now the filtration efficiencies for different particle diameters can be calculated by 10^x , with x representing the result of the logarithmic formula. This can for example be done for a coarse filter, as illustrated in graph 1 and table 4.



Graph 1: Size dependant efficiency distribution coarse filter according to Chen, Ji and Zhao (2022)

G4		v= 0.054 m/s		v=0.164 m/s
<i>dp (nm)</i>	$\log(\eta(dp))1$	$\log(\eta(dp))2$	$\eta(dp)U1$	$\eta(dp)U2$
10	0.09	-0.24	1.00	0.57
20	-0.65	-0.45	0.23	0.36
30	-0.79	-0.62	0.16	0.24
40	-0.85	-0.76	0.14	0.17
50	-0.89	-0.88	0.13	0.13
60	-0.93	-0.98	0.12	0.10
70	-0.96	-1.07	0.11	0.08
80	-1.00	-1.15	0.10	0.07
90	-1.03	-1.22	0.09	0.06
100	-1.07	-1.28	0.09	0.05
200	-1.27	-1.59	0.05	0.03
250	-1.29	-1.60	0.05	0.02
300	-1.27	-1.56	0.05	0.03
400	-1.13	-1.34	0.07	0.05
500	-0.89	-0.99	0.13	0.10
600	-0.59	-0.56	0.25	0.27

Table 4: Size dependant efficiency distribution coarse filter according to Chen, Ji and Zhao (2022)

6.5.2. Filter number

Evidently, more placing numerous air purifiers or filters will decrease the PM concentration as well. However, this comes at a significant higher cost (materials and energy wise). Therefore, other solutions to optimize the one-filter system might be more beneficial.

6.5.3. Filtration area

The specific filter area correlates to the number of particles that can be captured. This can be influenced by the filter structure. The efficiency of a filter structure is contingent with the pleat height and the density of the pleat. The traditional structure of a separator-plate filter has a medium height and density. Other newer structures usually have a higher height and/or density. Due to the dense packing of the fibres, the V-shaped filter has the highest density. Increasing either the height of the filter medium or the pleat density of the filter medium, results in a larger filtration area. Subsequently, this reduces the airflow resistance and thus the initial pressure drop. However, this reduction has its limitations. When the density is too high, the flow path is too narrow, and the resistance will be higher. Increasing the height of the

filter, works only when the medium is strong enough not to break. These two parameters are therefore very delicate to modify. Changing the filter material is usually a better and safer option (X. Zhang et al., 2022).

6.5.4. Ventilation rate

Air can enter the building through 3 kinds of air flows: mechanical ventilation, natural ventilations (windows) and infiltration via cracks in the building envelop. The ventilation rate of each of those three streams will be important to the indoor concentration of pollutants. Infiltration due to cracks and natural ventilation can be calculated using the formulas of Belias & Licina (2024).

$$Q_{inf} = C * (\Delta P)^n$$

$$\text{With } C = \text{flow coefficient in } \frac{m^3}{sPa^n}$$

$$\Delta P = \frac{\text{indoor}}{\text{outdoor}} \text{ pressure difference in Pa}$$

$$n = \text{flow exponent (assumed 0.7)}$$

$$Q_{NV} = C_d * A * \sqrt{\frac{2 * \Delta P}{\rho}}$$

$$\text{With } C_d = \text{discharge coefficient (assumed 0.65)}$$

$$A = \text{window opening surface in } m^2$$

$$\Delta P = \frac{\text{indoor}}{\text{outdoor}} \text{ pressure difference in Pa}$$

$$\rho = \text{air density in } kg/m^3$$

The ventilation rate of the mechanical ventilation system can be adapted to the system itself. As discussed in 6.2, the higher the filter efficiency, the higher the influence of the airtightness and the ventilation rate. Subsequently, when the efficiency is high, the concentration that leaks through the building envelop accounts for a higher percentage of the total concentration of indoor PM. When the flow rate is low, the influence of the airtightness on the total concentration is high as well. Particles from the outside will still enter at the same speed, but the filtered air enters at a slower pace, increasing the PM concentration. This also means that increasing the ventilation flow is a way to decrease the contribution of leaks to the total indoor PM concentration. Consequently, for a low indoor concentration of particulate matter, good airtightness of the building, a high filter efficiency and a high ventilation flow rate are required. (Verheyleweghen, Herreweghe, & Pecceu, 2022).

6.5.5. Operational time

The operational time of a filter can also have a significant influence. When airtightness is low, it is better to just let the ventilation run regardless of the outdoor concentration. Additionally, when efficiency is high, it is preferable to keep the ventilation system in operation, regardless of the airtightness of the building envelope. Because of the high efficiency of the filter, clean filtered air will then dilute the indoor air. When the ventilation system is not equipped with a (high efficiency) filter, switching off ventilation temporarily in case of an outdoor PM peak might be valuable. Only in the case of very poor airtightness the effect will probably be minimal. When the system has moderate filter efficiency, a strategy is less obvious. For these systems there is a tipping point at which the efficiency is too high to turn it off without decreasing the indoor air quality. This tipping point is dependent on the filter efficiency and the airtightness of the building (Verheyleweghen, Herreweghe, & Pecceu, 2022). However, people might not always use the ventilation system to the best of its ability.

6.5.6. Lifetime of a filter

PRESSURE DROP

The pressure drop of a filter can be defined as the difference in pressure between both sides of the filter stream. The higher the pressure drop, the higher the initial airflow needs to be to result in the wanted airflow at the filter outlet. According to the NBN EN 779 standard, a higher filtration efficiency automatically correlates to a higher pressure drop, and thus a larger operating cost. Filter classes with the highest efficiencies are, due to their large pressure drop and coinciding costs, used less frequently. Both HEPA and ULPA filters are usually preceded by a F7 to F9 pre-filter (Todea et al., 2020). The pressure drop of a filter is dependent on the filter structure, media, airflow velocity and the clogging of the filter (X. Zhang et al., 2022).

When only considering ultrafine particles, the research of Bourrous et al. (2016) stated that for a pleated filter, the ultrafine particle deposit can't obtain enough volume within the pleats due to the air flow resistance. So, the behavior of a pleated filter may be considered the exact same. However, when a more complete particle range is considered, there are some differences in the pressure drop of different shapes of filters, under different airflow rates. Research from X. Zhang et al. (2022), tested 23 HEPA filter samples under four different airflow rates. Those filters could have 5 different shapes: V-shape, prism, pleat-plate filters, separator plate filters or cylinder shaped. Results show that the first two can be used more widely at different flow rates, while the pleat-plate filter has the smallest range of applicable flow rates but has a higher pressure drop. The lowest pressure drop belong to the standard separator-plate filter followed by the prism filter. The pressure drops of the V-shaped and the cylinder filters were

both larger than the separator-plate and the prism but lower than the pleat-plate filter, as seen in the figure 10.

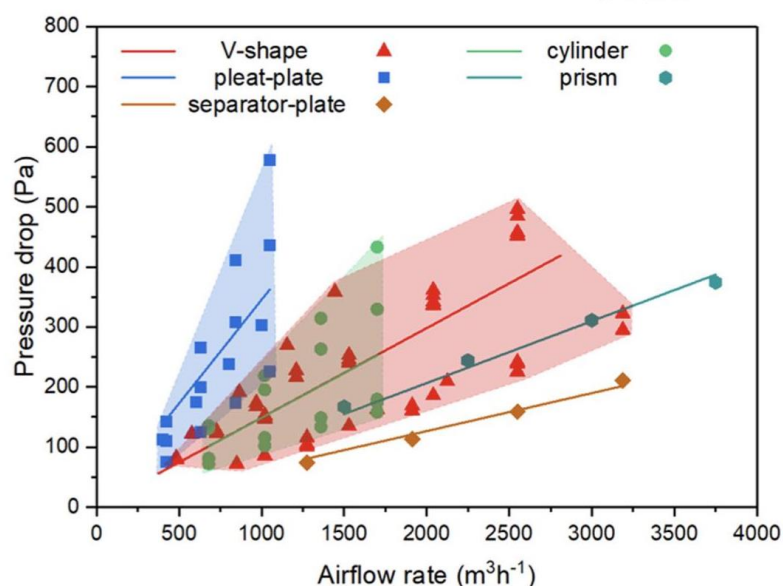


Figure 10: Influence of filter structure on pressure drop (Source: X. Zhang et al., 2022).

There is also a difference in pressure drop between different filter materials. In figure 11 you can see the difference between the pressure drop for a glass fiber filter and different nanofiber filter at different velocities. Since nanofiber filters were developed to have a lower pressure drop than a glass fiber filter, the large difference was to be expected. Nanofiber filters can also operate at larger flow rates. This kind of filters, however, have a very poor fire resistance (X. Zhang et al., 2022).

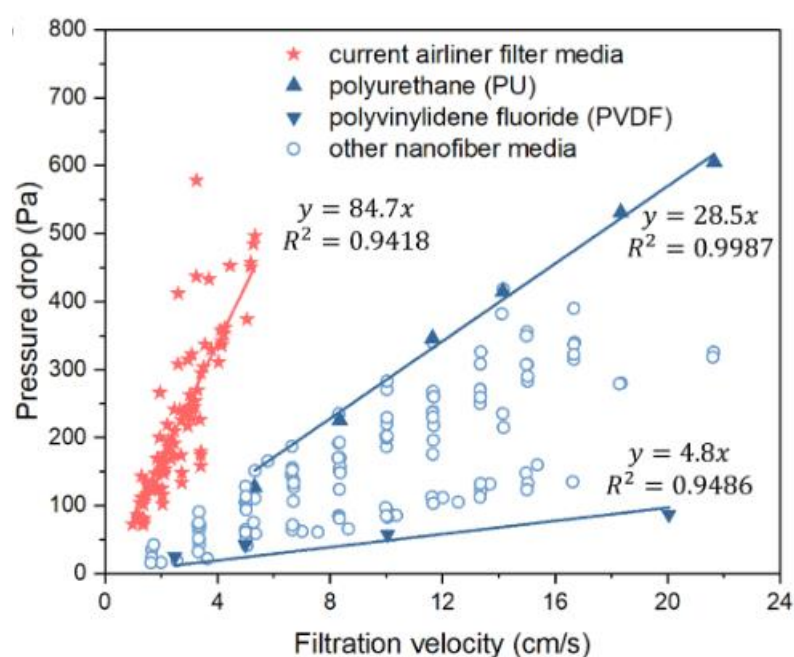


Figure 11: Comparison of pressure drop for different media (Source: X. Zhang et al., 2022)

In case of PTFE filters and PTFE composite filters, the flow resistance gives an indication of the expected proportions for the pressure drop. This can be seen in Figure 12. The glass fiber filters have a higher initial airflow resistance. While the PTFE filter has a lower initial resistance but as the dust loading progresses, the resistance increases significantly. Ultimately the resistance for the PTFE filter surpasses the resistance of the glass fiber filter. Using a composite can delay this incline (W. Zhang et al., 2022). A tendency that is consistent for both different filter structures, is that the higher the filtration velocity, the higher the pressure drop. A higher velocity will send more particles through the filter faster and therefore the resistance and pressure drop will increase faster.

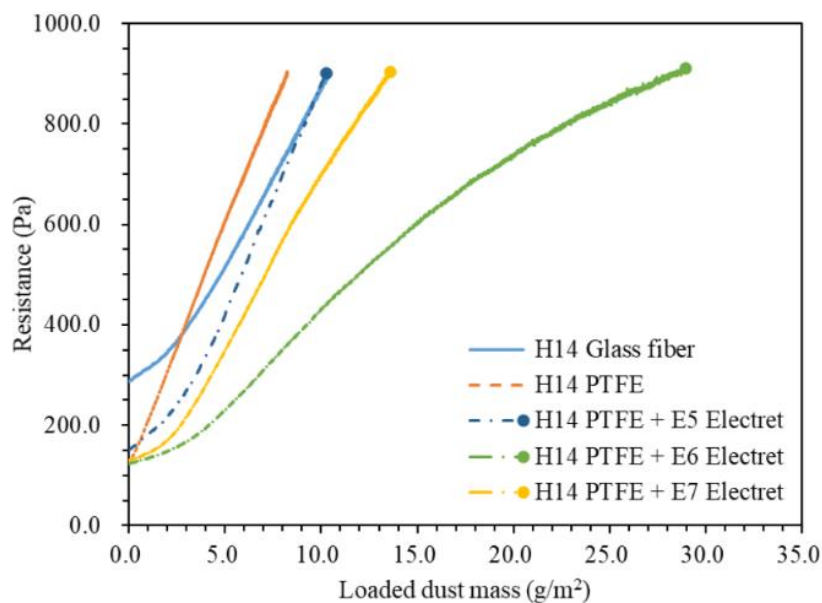


Figure 12: Dynamic resistances for three composite PTFE media (Source W. Zhang et al., 2022).

CLOGGING

The pressure drop varies during varies degrees of clogging. Clogging occurs when particles get stuck in the filter media and start to form a filter deposit inside the media and on the surface. This particle deposit will result in an increased resistance against airflow and an increased pressure drop. To overcome this pressure drop, a larger amount of energy will be necessary. At some point, the pressure is too high to justify the costs. If we can predict the pressure drop that obstructs the proper functioning of the filter, we will be able to predict the ultimate life time of a filter (Bourrous et al., 2016).

According to the model developed by Mocho and Ouf (2011), the resistance of the filter increases with decreasing particle size, since the dust cake will be more compact in the case of smaller particles. This means that ultrafine particles will be especially clogging particles (Bourrous et al., 2016). Since the smallest particles are also the most damaging, it is essential to 1) prevent this particles from entering

the indoor environment and 2) being able to predict when a filter does no longer filter these particles efficiently enough and needs to be replaced.

The overall mechanisms in place during the clogging process are shown in figure 13. The clogging of a filter can result in a significant decline in air flow rate. Additionally, the corresponding pressure drop and RPM (revolutions per minute of the fan) increase. And, to compensate for the air flow reduction, the output power of the installation rises as well (S.-H. Park et al., 2021).

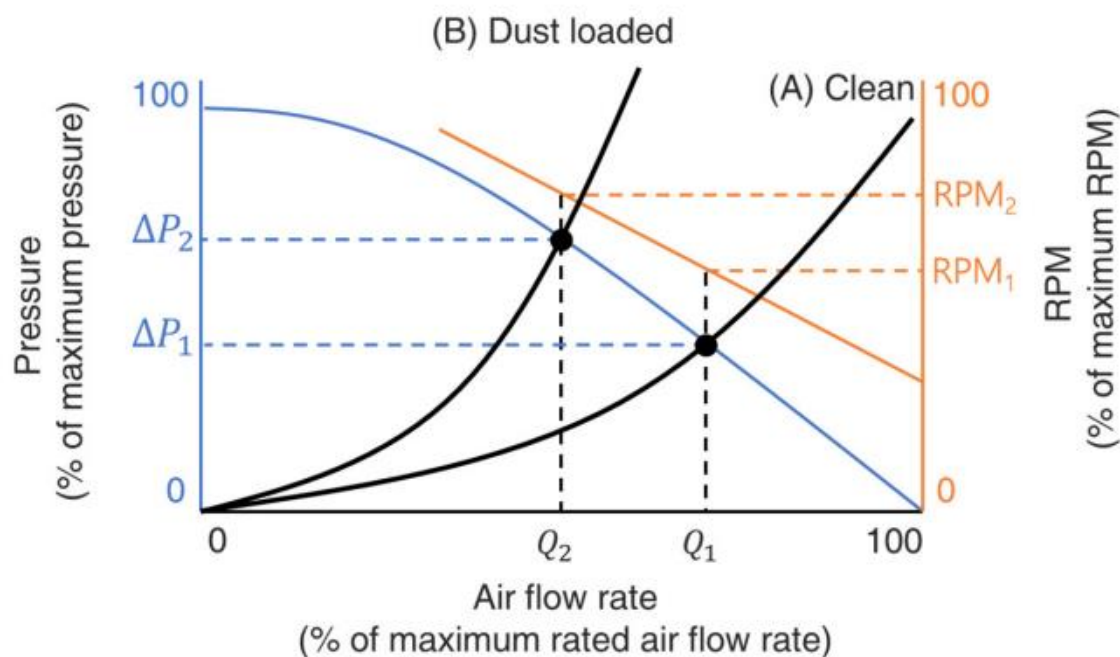


Figure 13: Schematic of changes in SA fan operation characteristics according to the filter conditions (Source: S.-H. Park et al., 2021)

The clogging phenomenon of a filter can be described by two phases. In the first phase, particles are absorbed by deep filtration inside the filter medium. This first absorption of particles will block the pores of the filter medium. This blockage will initially create an increase in the airflow resistance of the filter. This phase can be characterized by a penetration profile of the different particles deposited in the filter. In the next phase, or surface filtration, a deposit is formed on the outside of the filter. This phase goes together with a significant enlargement of the pressure drop. This increased pressure drop is mainly determined by the porosity of the deposited particles, the specific area and the diameter of the particles that are deposited on the surface of the filter medium. There is also an intermediate phase where particles are collected both on the filter medium and inside the filter medium (Bourrous et al., 2016). In case of a pleated filter, a third phase can be observed. This will be called the surface reduction. This occurs when

the mass of the particle cake is sufficiently high, resulting in a reduction of the cake due to the weight. This third phase is accompanied by an acceleration in the increase of the pressure drop. This phenomenon is dependent on the air flow and the pleat geometry. During surface filtration pressure drop can be characterized by a linear increase in value. The assumption is usually made that depth and surface filtration regimes are not dependent on the filter geometry. Pleated filters act the same as flat filters when it comes to deep filtration. So deep filtration does not depend on the geometry of the filter medium (Bourrous et al., 2014).

In certain instances an inhomogeneous deposition occurs in the pleat. This results in the reduction of the efficient filtration surface before the pleat is fully loaded with particles. This could be explained by Brownian diffusion. However, this only seems to be the case when a very low velocity is used (Bourrous et al., 2014).

CLOGGING AND EFFICIENCY

According to the American standards, a filter should be replaced when the pressure drop is twice the initial pressure drop. However, most brands recommend replacing every 6 to 12 months. Knowing when a replacement is necessary is therefore difficult for the user (S.-H. Park et al., 2021).

How the efficiency of filter is influenced by the clogging of the filter depends on the specific dust characteristics in the specific building. More important, however, is the type of filter installed. The type of filter can determine whether the efficiency increases or decreases for different particle sizes when the dust-load increases. When a filter media is not charged, the efficiency often increases for increasing dust-load. Bigger particles will pile up in the inside of the filter, which results in additional filtration capacity. However, when a filter does utilize a charge, the efficiency often decreases over time. In the small particle size fractions this effect can be significant. Here, up to 50% of the original removal efficiency can be lost. The many small particles will be deposited on the medium and decrease the effect of the charge. This can be combined with an increase of efficiency due to larger particles, similar to uncharged filters (Owen & Kerr, 2020).

PRESSURE DROP CALCULATION

To estimate and model the pressure drop for ultrafine particles in case of deep filtration, the method from Bourrous et al. (2014), can be used. Via a simplified capillary approach, a clogging model for flat fibrous media was developed based on the penetration profile, the porosity of the particle deposit and the filter characteristics. This model also gives a clogging point which stands for the particle mass accumulated in the filter when the transition from deep filtration to surface filtration occurs. Additionally,

this can be defined as the start of the linear incline in pressure drop. Possible inconsistencies will only be present on the exterior of the filter. An illustration of this can be seen in figure 14. (Bourrous et al., 2016).

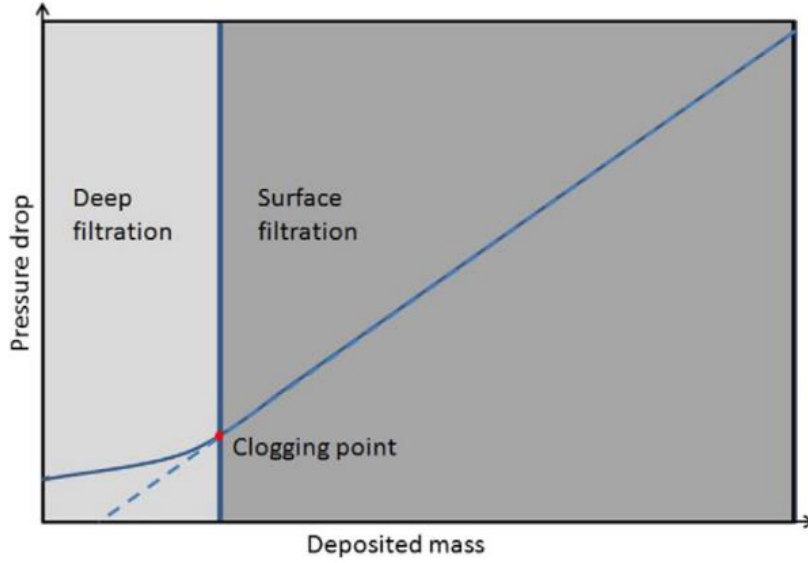


Figure 14: Illustration of the pressure drop phases in function of deposited mass (Source: Bourrous et al., 2016)

Firstly, the penetration profile k is determined by using a relation with the fraction of particles at a certain depth $\alpha(x)$ in the medium.

$$\alpha(x) = \frac{m}{S} * \frac{ke^{-kx}}{\rho_p(1-e^{-kZ})}$$

with m = mass deposited

S = filtration area

Z = actual thickness of the medium

ρ_p = density of the constituent material of the aerosol

The filter medium is assumed to be homogeneous with a packing density of α_m . First, the clogging point was determined. After this point the direct correlation between the pressure drop and thickness of the deposit can be used to obtain the linear course of the surface filtration.

The volume fraction that is occupied by the deposit at a depth x in the filter can be described as follows.

$$\alpha_{tot}(x) = \alpha_m + \frac{\alpha(x)}{\alpha_g}$$

with α_m = packing density of the medium

α_g = porosity of the accumulated deposit in the medium in exponential form

A deposit on the surface is formed when all the empty filter volume is filled with particles. ($\alpha_{tot}(0) = 1$)
. At this point, the mass in the filter can be described by:

$$m_{cp} = \frac{S \cdot \alpha_g \cdot \rho_p \cdot (1 - e^{-kZ}) \cdot (1 - \alpha_m)}{k}$$

The pressure drop for this laminar flow, through a capillary is expressed by Poiseuille's relation

$$\Delta P = \frac{128 \mu Q Z^*}{\pi d^4}$$

with d = diameter of an empty capillary

Q = flow through a capillary

Z^ = thickness of the equivalent medium = $Z \cdot \alpha_m$*

The flow through a capillary:

$$Q = \frac{\pi \cdot d^2 \cdot U_f}{4}$$

with U_f = filtration velocity

The pressure drop for the clogged section can be determined using the Kozeny-Novick relation.

$$\Delta P_g = h_k \frac{36 \cdot \alpha_g \cdot \mu \cdot Q \cdot d \cdot m}{C_u \cdot \varepsilon^3 \cdot \rho_p \cdot d_p^2 \cdot S^2}$$

With $h_k = 4.7 + e^{(14(\varepsilon - 0.8))}$

In order to calculate the pressure drop at a certain depth, parameters Q and d will be dependent on x .

To use this model knowledge of the packing density, thickness of the medium, size of the primary particles and the porosity of the deposit is necessary. However, the model is only valid when a homogeneous particle accumulation of the particles can be assumed.

In order to calculate the initial pressure drop, the formula from the research of Chen, Ji and Zhao (2019) could also be used. This formula was obtained for different filter classes for two different flow rates (0.054 m/s and 0.164 m/s). The pressure drop for fibrous filters can be described as follows.

$$\Delta P = \frac{16\mu\alpha L}{(Ku)(d_f^2)} U = KU$$

With μ : dynamic viscosity [Pa * s]

α : packing density

Ku : Kuwabara hydrodynamic factor

L : filter depth of thickness [m]

d_f : fiber diameter [m]

U : airflow velocity $\left[\frac{m}{s}\right]$

K : ratio $\frac{\text{Pressure drop}}{\text{velocity}}$ $\left[Pa \frac{s}{m}\right]$

Via linear fitting the coefficient K was calculated for every filter class. These can be observed in table 5. These results were compared with previous studies and were very similar. But they still present a simplification, so some differences with real life tested values can be expected.

	K	Pressure drop v = 0.054 m/s	Pressure drop v = 0.164 m/s
G1	28	1.512	4.592
G2	40	2.16	6.56
G3	37	1.998	6.068
G4	79	4.266	12.956
F5	122	6.588	20.008
F6	169	9.126	27.716
F7	276	14.904	45.264
F8	439	23.706	71.996
F9	1002	54.108	164.328
E10	1035	55.89	169.74
E11	2578	139.212	422.792
E12	3704	200.016	607.456
H13	4286	231.444	702.904
H14	5816	314.064	953.824
U15	7141	385.614	1171.124
U16	4778	258.012	783.592
U17	5508	297.432	903.312

Table 5: Different K coefficients for different filter classes (Source: Chen, Ji and Zhao, 2019)

6.5.7. Energy consumption

Ventilation can be held responsible for up to 50% of the operational energy demand in a well-insulated building. A ventilation system is used to improve indoor air quality but also to save energy in case of favorable outdoor conditions. Natural ventilation can be used to reduce the cooling demand and bring down operating costs. If the outdoor air, is cleaner than the air present indoors, the air quality will also

improve. Finding a balance between a good indoor air quality and minimal energy demand is a constant search. This tradeoff can be strongly influenced by meteorological variations of the outdoor environment and the variations of the particulate matter. Sufficient information about both is necessary to obtain the best possible outcome for both objectives (Belias & Licina, 2024).

When a filter is present in this ventilation system, the energy consumption of a ventilation system will consist of two parts. Firstly, the ventilation system itself will require energy to work. Secondly, when a filter is present in the ventilation system, the energy use of the installation will not be constant due to clogging of the filter. So, the lower the pressure drop the lower the energy consumption (X. Zhang et al., 2022). And, the longer the filter is in place, the more energy the system requires (W. Zhang et al., 2022). A minimal initial pressure drop and a as slow as possible increase is therefore very important in terms of energy consumption.

The power of the ventilator fan can be calculated following the EN 16798-3 standard (Belias & Licina, 2024).

$$P_{fan} = \frac{Q_v * \Delta P_{tot}}{\eta_{tot}}$$

With P_{fan} = usefull power of the fan in Watt

Q_v = airflow through the fan in $\frac{m^3}{s}$

ΔP_{tot} = total pressure drop of the ventilation system in Pa

η_{tot} = total efficiency of the fan

In case of demand-controlled ventilation, the reduced power of the ventilator fan can be calculated according to the ASHRAE standard 90.1 (Belias & Licina, 2024).

$$P_{fan}(t) = P_{nom} * P_f(t)$$

With $P_{fan}(t)$ = usefull power of the fan at moment t in Watt

P_{nom} = nominal useful power of the fan for maximal airflow in Watt

$P_f(t)$ = full – load power of the fan

And:

$$P_f(t) = 0.0013 + 0.1470 * \frac{Q(t)}{Q_{max}} + 0.9506 * \left(\frac{Q(t)}{Q_{max}}\right)^2 - 0.0998 * \left(\frac{Q(t)}{Q_{max}}\right)^3$$

With $Q(t)$ = airflow at time t in m^3/s

Q_{max} = maximum desgined airflow in m^3/s

The operation energy of a ventilation system can be reduced by using demand-based ventilation and/or energy recovery. This results in a reduction of the demand with 25.9 %. When ventilation is based on the indoor/outdoor temperature difference this reduction is 29.2%. But when outdoor air pollution (with boundary values depending on the WHO 2006 guidelines) limits this ventilation, the reduction is only 13.5 %. If the most recent air quality guidelines would be used, this would probably be even less. So the influence of pollutants is significant.

The question is, however, does the cost of a ventilation system with filter, exceed the additional cost of PM related health effects. Outdoor PM_{2.5} contributes to 62% of the total DALY's associated with air pollution exposure in the indoor environment in case of standard ventilation rates. The optimization of ventilation rates, the addition of filters and the limitation of indoor sources can result in a reduction of this number by 20-44% in Europe. Using natural ventilation instead of mechanical ventilation with filtration of particles increases the PM exposure and ultimately results in additional health care costs of about 130-207 million dollars per year. This shows that the cost of health care due to DALY's associated with bad air quality is higher than the cost of energy (Belias & Licina, 2024).

7.

Research questions and hypotheses

After this detailed analysis of the existing literature, certain research questions can be posed. The existing standards only consist of a very limited PM size division. This division makes a generalization of the toxicity of different particle sizes. Considering the large diversity in health effects for various diameters of PM, this makes it difficult to link certain concentrations to an expected level of health. More size divisions are necessary to access and resolve PM related health effects.

Furthermore, filters are used to minimize the indoor pollution concentration of PM as part of a mechanical ventilation system or an air purifier. However, the effectivity of different filters is usually tested in laboratory conditions with particle generators and only one size particles. In real life, the particulate matter concentration is not only subject to the particles that are able to penetrate the filter, but also by various other factors like the outdoor concentration, indoor PM sources and window ventilation. In a real-life environment, a filter might not prove effective enough.

Additionally, it is evident that by filtration, particles are captured in the filter medium. This has an impact on the filter efficiency but also on the service life of the filter due to the enlargement of the pressure drop. At one point in time, the pressure drop will exceed a boundary value for which the energy use of the filter is too high to be sensible. Whether or not this turning point proceeds or exceeds the 6-12 months guidelines in a real-life situation should be investigated.

Lastly, when particulate matter is filtered from the indoor environment, other pollutants might turn out to be more dangerous than the remaining PM concentrations.

Concretely, the following research questions can be posed. Based on the reviewed literature, a hypothesis can be formulated for each question.

1. Is a filter able to significantly reduce the indoor PM concentration to provide a healthy indoor air quality?
 - a. A HEPA filter can provide a significant decrease; however a coarse or fine filter is not.
2. What parameters (apart from the filter) influence the indoor concentration of particulate matter the most?
 - a. The biggest influential factors are the use of window opening for natural ventilation and the presence of cooking.
3. At what point in time is the replacement of the filter required to maintain an acceptable energy use?
 - a. Replacement of the filter is necessary between 6-9 months after installation.
4. What pollutants are most dangerous when the particulate matter concentration is filtered from the air?
 - a. The most dominant pollutants when controlling PM are NO_2 and formaldehyde.

8.

Research method

8.1. Procedure

Firstly, to be able to better understand the danger of different PM particle sizes in the indoor environment, a new division of particulate matter is developed. This division is based on the literary research regarding possible health effects. Additionally, the filter efficiency curve and airtightness is also taken into account. This will provide a better understanding of the risk a certain environment encompasses. Additionally, it could result in a more targeted strategy for PM concentration control.

Secondly, to investigate the different parameters that can influence the indoor PM concentration, simulations were used. These simulations were performed using Dymola software based on the Modelica language. For these simulations, a filter model is developed. The model is designed so that each parameter can be derived from the test report according to the current filter standards. This allows for maximal adaptation in other circumstances. In this dissertation, the focus is on the HEPA filter classified filters. HEPA filters can remove 99.97 % of particles with a particle size greater than or equal to 0.3μ .

Next, this filter model is placed in an existing multizone building environment, developed by De Jonge (2023), as part of a ventilation system. This model is used to simulate different real-life scenarios. Additionally, an adapted filter model is developed to simulate filter lifetime. Due to simulations in a real-life environment, it can then be determined, whether or not a HEPA filter provides sufficient particulate matter declination.

In order to determine whether or not the particulate matter concentration in the indoor environment is low enough to avoid health effects, the guidelines of the World Health Organization are used. Therefore, the average concentration of particulate matter in each room is determined as well as the number of days when the maximal daily concentration is exceeded. This provides a good initial indication of whether or not the air is sufficiently healthy. Lastly, the DALYs metric is used to determine the amount of health life years lost in various scenarios. This will also allow the comparison between PM and other pollutants.

The most critical remaining pollutants can be appointed. Ultimately, the goal is to be able to determine how efficient filter use in a real-life environment is and how likely possible health effects of this remaining PM concentration could be.

8.2. Bin sizes

The classical divisions of PM resulted in the current regulations that are in place for particulate matter. Only limit values for fine and coarse PM are considered in current guidelines. This limits the adaptability of those limit values to the exact risks a person is subjected to when exposed to a specific size of particulate matter. A more elaborate division of PM is needed in order to make those risk more tangible and susceptible.

8.2.1. Bin sizes based on health effects

Very large particles ($>10\text{ }\mu\text{m}$) are inhaled less frequently due to their faster settling rate. Additionally, when these particles do enter the body via nose or mouth, they are usually effectively filtered by the defense mechanisms provided by the respiratory tract, the cilia, and the mucus. Therefore, this fraction is lodged in the nose and throat. This may result in discomfort like coughing and sneezing but no lasting damage to the overall health. For that reason, they are not considered PM and can also be considered unharmed. (Kim et al., 2015).

PM particles smaller than $10\text{ }\mu\text{m}$ are capable of penetrating inside the respiratory tract and travel from the nasal passages to the deeper parts of the system. Particles that belong to the fraction with a size between 5 and $10\text{ }\mu\text{m}$ will most likely reach the tracheobronchial tree and get deposited there. This will result in mostly respiratory responses, causing inflammation and irritation.

Particles smaller than $5\text{ }\mu\text{m}$ but bigger than $1\text{ }\mu\text{m}$ can get through to the respiratory bronchioles and the alveoli. This is the smallest element of the respiratory tract (Kim et al., 2015). Overall, PM concentrations with sizes varying from $10\text{ }\mu\text{m}$ to $2.5\text{ }\mu\text{m}$ seem to be responsible for the bigger part of, oxidative stress responses and reduction of cell mitochondrial activity (Yang et al., 2023).

Particles smaller than $1\text{ }\mu\text{m}$ have gas like behavior due to their size. Such particles are therefore able to penetrate through the alveoli. Since the alveoli are responsible for gas exchange with the circulation system, particles can travel further to the rest of the body once they have passed the alveoli and can even reach the cell tissue (Kim et al., 2015). Health effects like DNA damage, an impaired cell membrane or growth of sub G1/G0 population (all endocrine mechanisms) can be allocated to these smallest particle sizes. Furthermore, PM_{0.5} could, according to some studies, be responsible for cardiopulmonary disease mortality (Yang et al., 2023).

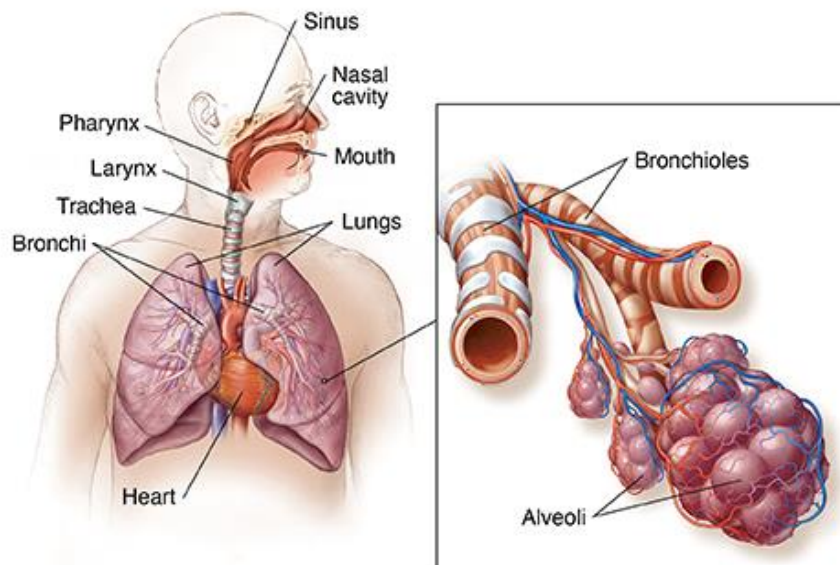


Figure 15: Schematic of the Pulmonary tract and its parts (Source: Kyung & Jeong, 2020).

Particles in the range from 0.25 -0.4 μm seem to increase the risk for inflammation, coagulation and vasoconstriction (Manigrasso et al., 2022). While particles smaller than 0.24 μm are small enough to be able to penetrate the placental barrier and can consequently damage fetuses. If the particle size is smaller than 0.2 μm , PM can reach the brain and cause impairments in that region (Lowther et al., 2020). A schematic overview of the conclusions above can be found in figure 16. The full image can be found in Annex A.

PM BIN SIZES BASED ON HEALTH EFFECTS

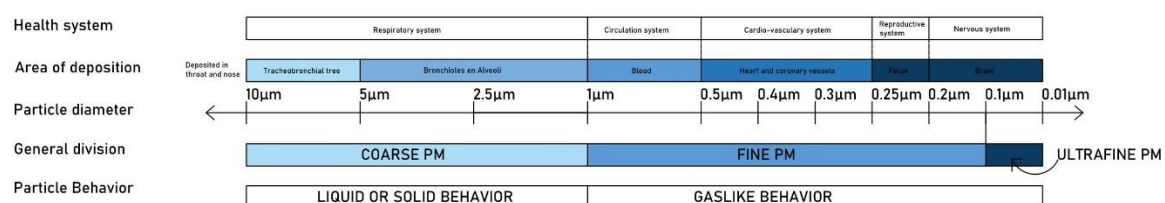


Figure 16: PM bin sizes based on health effects, see annex A.

8.2.2. Bin sizes based on removal efficiency

Generally, the fractional efficiencies of a random filter can be depicted a V shaped curve. (Todea et al., 2020). Each filter has a different MPPS or most penetrating particle size, for which the minimal filter efficiency of the filter is reached. Typical coarse and medium filters have a MPPS of about 200-300 nm. Whereas for H and U filters, the through of the curve lies around 120 nm to 250 nm. According to research of Lowther et al. (2020) the removal efficiency of a H-filter is consistently the least at an

intermediate particle size of 100-400 nm or 0.1-0.4 μm . With the particle sizes between 200-250 nm or 0.2-0.25 μm being the most difficult sizes to remove. In the research of Maria Todea et al. (2020), the MPPS for an ISO 30 E and ISO 35 H filter was placed at 130 nm. This results in the scheme below.

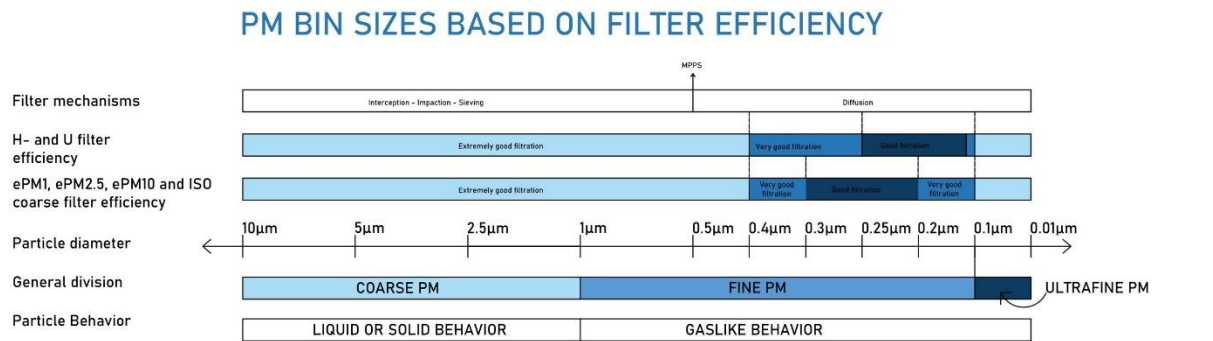


Figure 17: PM bin sizes based on the filter efficiency, see annex A.

8.2.3. Bin sizes based on the building shell

According to Lowther et al. (2020), the particle sizes that are least effectively removed by filters (0.2-0.3 μm) are also the particle fractions that can penetrate through the building envelop most effectively. With particles of 0.2 μm being the easiest to penetrate.

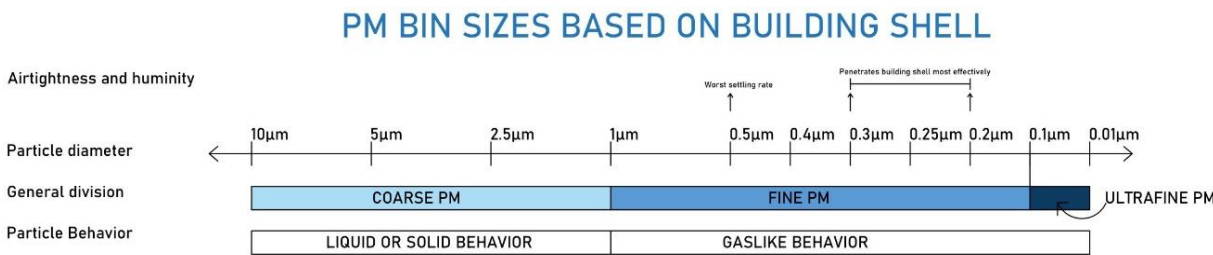


Figure 18: PM bin sizes based on the building shell, see annex A.

8.2.4. Bin sizes conclusion

Favourable bin sizes according to the different parameters can now be described. As an upper limit, it is obvious to impose a limit of 10 μm . From a health perspective, boundaries are added every time the particles of a certain size can travel further into the body. The boundary values of PM2.5 and PM1 are also used since these are the current boundary values. Furthermore, the outer limits of the efficiency throughs and particles that penetrate the building shell more easily, are considered and added when necessary.

The lower limit for the bin sizes is placed at 0.01 μm . Particles smaller than this value are extremely difficult to measure in test setups and therefore often not included in real life data results. Additionally,

these particles are not easy to simulate. The overall knowledge of these ultra small particles is therefore limited. They could however be even more damaging than other sizes of PM, but further research is necessary to confirm or deny this.

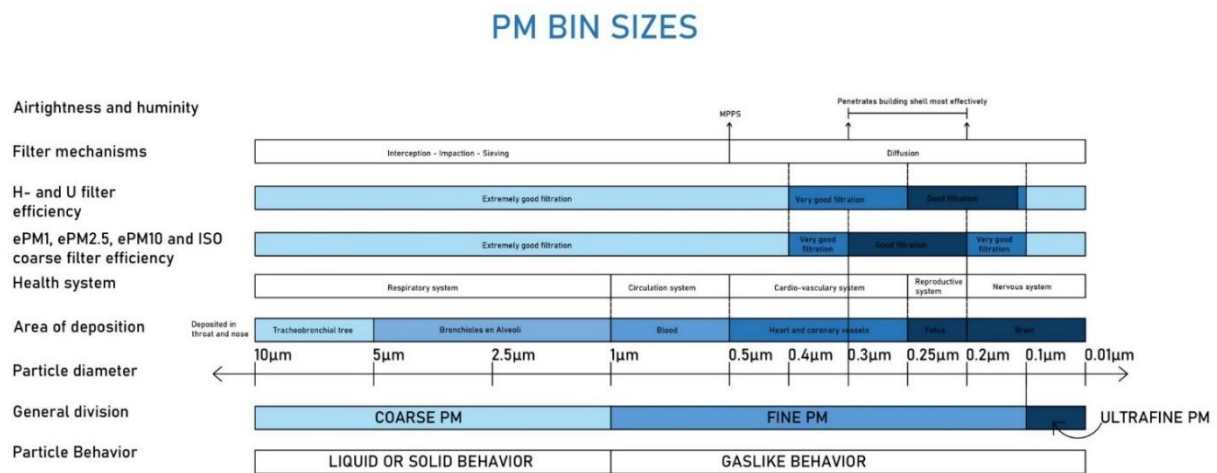


Figure 19: PM bin sizes overview, see annex B.

8.3. Building simulation model

8.3.1. Test procedure coarse filters

Coarse filters are evaluated according to NBN EN ISO 16890 standard. The tests are performed at the nominal air flow rate for which the filter was designed by the manufacturer. If no air flow is specified, the tested rate is 0.944 m³/s for filters with dimensions 610x610mm. During the testing procedure the resistance to airflow is measured. Furthermore, an efficiency curve is constructed at this air flow rate, both for an initial and an unconditioned filter as for a conditioned filter. At least 5 points are necessary for this curve. By conditioning the filter, the possible reliance on the electrostatic removal mechanism can be determined. Afterwards these curves are used to calculate the average fractional efficiency curve. The percentual efficiencies for each ePM class can then be calculated. The initial and final pressure difference is also determined.

Group designation	Requirement			Class reporting value
	ePM _{1, min}	ePM _{2,5, min}	ePM ₁₀	
ISO Coarse	—	—	<50 %	Initial grav. arrestance
ISO ePM ₁₀	—	—	≥50 %	ePM ₁₀
ISO ePM _{2,5}	—	≥50 %	—	ePM _{2,5}
ISO ePM ₁	≥50 %	—	—	ePM ₁

Figure 20: Filter groups according to EN ISO 16890-1 2017.

8.3.2. Test procedure high efficiency filters

For Europe, the efficiency of high efficiency filters, is tested using the methods described in the first part of the EN 1822 standard. The filtration performance is determined by the efficiency of the filter for its MPPS, which is the diameter of the aerosol particles. The nominal air flow rate at which the filter is tested is determined by the manufacturer.

The first step in the testing procedure itself, consists of measuring the pressure drop in unloaded state (initial pressure drop). Next, the aerosol mixed with air is send through the filter and both the number of particles before and after the filter are counted. This results in the efficiency E of the filter for its MPPS. The penetration percentage is expressed using following equitation:

$$P = \frac{C_{N,d}}{C_{N,u}}$$

With

$$C_{N,d} = \text{the number concentration downstream equal to } \frac{N_d}{V_{s,d} * t_d}$$

$$C_{N,u} = \text{the number concentration upstream equal to } \frac{k_D * N_u}{V_{s,u} * t_u}$$

With

N_u = number of particles counted upstream

N_d = number of particles counted downstream

k_D = dilution factor

$V_{s,u}$ = sampling volume flow rate upstream

$V_{s,d}$ = sampling volume flow rate downstream

t_u = sampling duration upstream

t_d = sampling duration downstream

The efficiency E can then be expressed by:

$$E = 1 - P$$

The efficiency of a flat filter medium is tested for a range of particle sizes at the nominal filter medium velocity. Subsequently the MPPS is determined from the resulting curve that show the efficiency versus the particle size. This curve is drafted with the help of at least six interpolation points. Each procedure needs to be performed for at least five samples of the filter material. The material needs to be at least 200mm x 200 mm. The mean pressure difference is also determined during the first procedure before aerosols are loaded onto the filter (Source: EN 1822-1:2019).

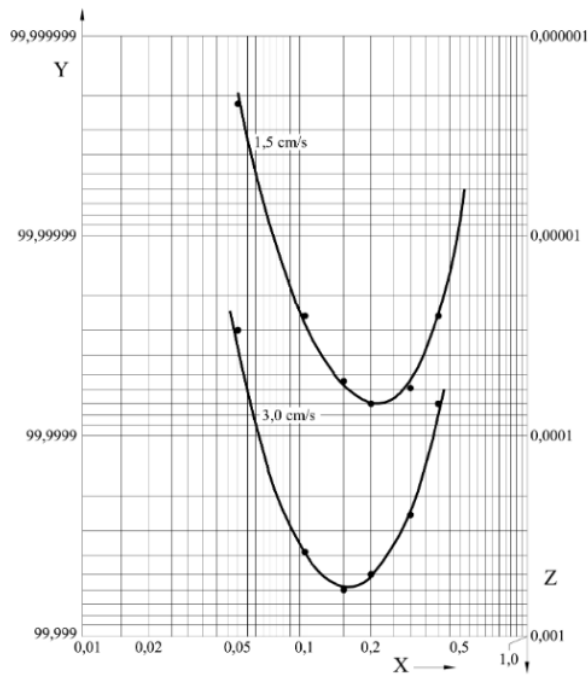


Figure 21: Example of a fraction efficiency curve in function of the particle diameter d_p (Source: NBN EN 1822-1, 2019).

In short, following the testing procedures described in the NBN EN ISO 16890 and NBN EN 1822 norm results in a curve that plots the efficiency versus the size of the particle. The MPPS and integral efficiency are also calculated. Furthermore, the mean pressure difference is analyzed as well. The nominal air volume flow rate, face area and subsequent nominal filter medium face velocity is determined by the manufacturer.

8.3.3. Development of the filtermodel

The previously described testing standard will be used as the basis for the filter model. The model should work with only the output parameters of the norm. This way, different filters could be compared using the model, when manufacturers would make this data available. Applied data will, therefore, be viewed as possible output data of the standard.

However, some remarks need to be made. Since the relative humidity can influence the settling rate, the broad requirement of a relative humidity below 75%, can pose quite some differences. The higher the humidity, the better for the settling rate. A narrower range of relative humidity required for the testing of filters could allow for a better comparison between filters. Furthermore, the nominal air volume flow rate for which the filter will be tested is specified by the manufacturer. This poses a problem for the comparison of different filters. A different airflow will result in a different filter efficiency curve. Therefore, filters tested at different velocities cannot be compared. This makes deriving general conclusions about

airflow exceedingly difficult. In a future updated standard it would be beneficial to include certain standard velocities that need to be tested in order to facilitate the comparison between different filters.

Now, to further investigate the influence of different filters on the indoor PM concentration in different circumstances, a filter component was modelled in Dymola. This filter accounts for the filtration efficiency that will later be applied to the PM concentrations from the outdoor environment. In order to be able to connect conclusions regarding the health impact of particulate matter, the ten different bin sizes as described in 8.1 were used. Each bin size represents a trace substance with a specific size diameter range.

To ensure that the model works with the outputs of a standard test rapport according to the standard, some parameters needed to be added to the filter component. Input parameters that are determined by the manufacturer are:

- Surface area of the filter medium [m^2]
- Nominal air volume flow rate [m^3/s]
- Nominal face velocity [m/s]

When the nominal air volume flow rate and the surface area are known, the face velocity can be calculated.

- Nominal face velocity [m/s] = Nominal air volume flow rate [m^3/s] / Surface area of the filter medium [m^2]

Unfortunately, since the testing standard does not require a specific flow rate, these flow related parameters will be separate entities from the other parameters. If a connection should become available, this can be easily added. A last parameter will be the pressure drop [Pa] that the filter is subjected to at its initial condition. Subsequently, each tracer will need to be connected to its matching filter efficiency. In order to make the model applicable for each filter, the number of tracers and efficiencies can be adapted in the model itself.

The basis scheme then consists of the filter component that requires the filter efficiency as input, a component that represent the PM particles that go through the filter (“air bubble”) and components that represent the flow rate that goes through the filter medium. Different concentrations of particulate matter of different sizes can be added to the “air bubble”, and consequently also the different corresponding filtration efficiencies. The result of this model gives filtered concentrations for particulate matter particles of various sizes. Figure 22 shows the graphical Modelica implementation of the filter.

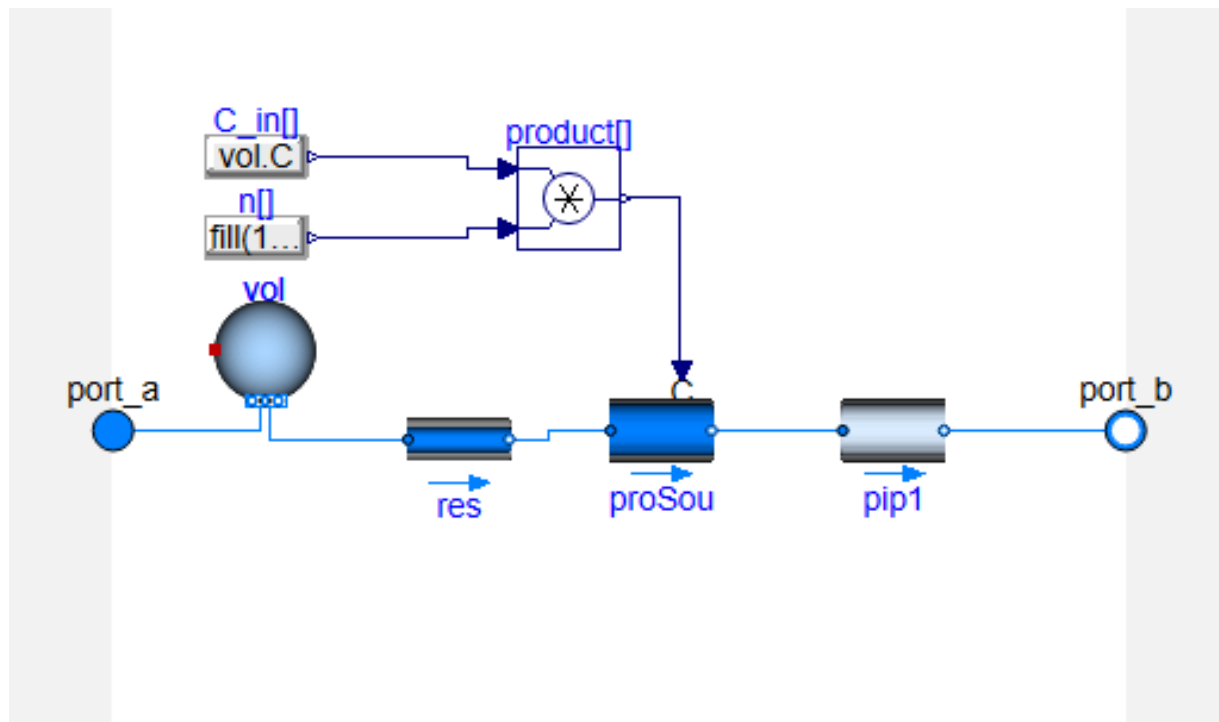


Figure 22: Modelica representation of the filter model

8.3.4. Validation of the filter model

In order to validate this model, the Out2In project by Buildwise (previously WTCB) is used. During this research the filtration efficiency of different filters belonging to different (coarse) filter classes were measured. The tests were carried out with the real PM concentration of the outdoor air in Brussels. Therefore, this research is especially suitable for the validation of the Dymola model. A measuring range of $0.01\mu\text{m}$ till $10\mu\text{m}$ was implemented. The boundary conditions of the test setup were as follows (Van Herreweghe et al., 2019).

- Round metal ductwork with $\varnothing$ 160 mm
- Constant flow fan of $150\text{ m}^3/\text{h}$

Conversion of these parameters gives the parameters from the NBN norm:

- Surface area of the filter medium $[\text{m}^2] = 0.08^2 \cdot \pi = 0.021\text{ m}^2$
- Nominal air volume flow rate $[\text{m}^3/\text{s}] = 150\text{ m}^3/\text{h} \cdot 1\text{h}/3600\text{s} = 0.042\text{ m}^3/\text{s}$
- Nominal face velocity $[\text{m/s}] = 0.021/0.042 = 0.5\text{ m/s}$

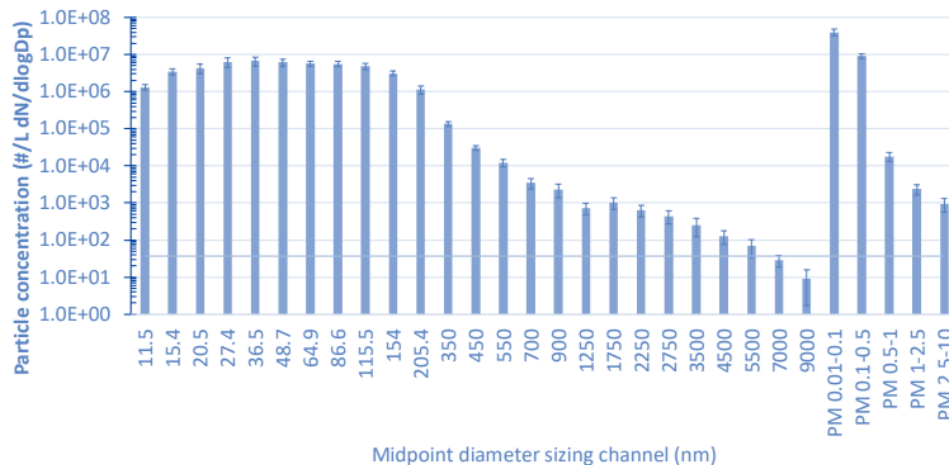
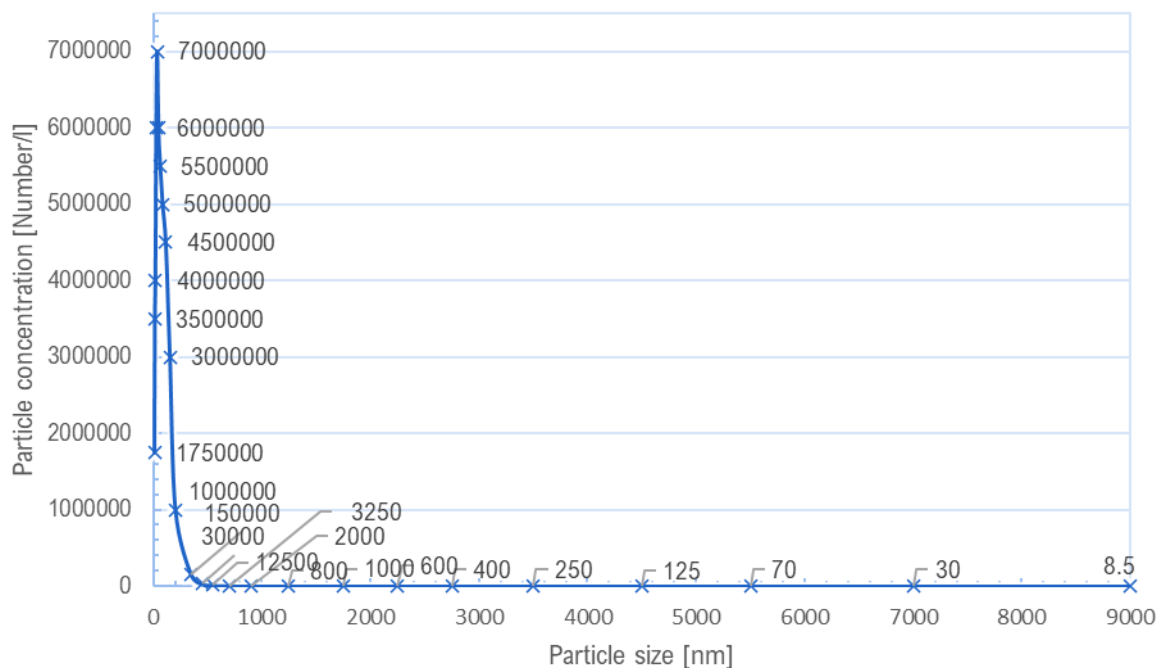


Figure 23: Typical profile of the particulate load of the Brussels outdoor air according to the Out2nNproject by Van Herreweghe et al. (2019).

In figure 23 the real-life particle load of the outdoor environment in Brussels is displayed as used in this project. For various particle diameters the number of particles per liter of air is depicted. It is obvious that the particle load is dominated by the smaller particle classes. Particles with a size above $0.5 \mu\text{m}$ represent a minimal percentage of the total particle load of 0.044% (Van Herreweghe et al., 2019). In order to be able to compare different filters tested with the standard, the different concentrations were combined in graph 2. Afterwards ten inputs were determined, each corresponding to the average size value for one of the bin sizes, these can be found in table 6. The same method could be used in case of different bin sizes/particle size ranges.



Graph 2: Particle size distribution derived from the particle sizes and concentrations from figure 23.

	dp (nm)	dp (µm)	Particle concentration (#/L)
Bin size PM0.01-0.1	55	0.055	5805556
Bin size PM0.1-0.2	150	0.15	3155844
Bin size PM0.2-0.25	225	0.225	884786
Bin size PM0.25-0.3	275	0.275	590871
Bin size PM0.3-0.4	350	0.35	150000
Bin size PM0.4-0.5	450	0.45	30000
Bin size PM0.5-1	750	0.75	2938
Bin size PM1-2.5	1750	1.75	1000
Bin size PM2.5-5	3750	3.75	219
Bin size PM5-10	7500	7.5	25

Table 6: Particle concentration for each bin size calculated by interpolation.

To validate the model, a coarse 60 % FP is used. In order to simulate the output of the standard testing method, a graph was constructed with the different efficiencies of this filter for different particle sizes.¹ This is similar to the fractional efficiency curve that the test procedure from the EN 1822-1 standard provides.

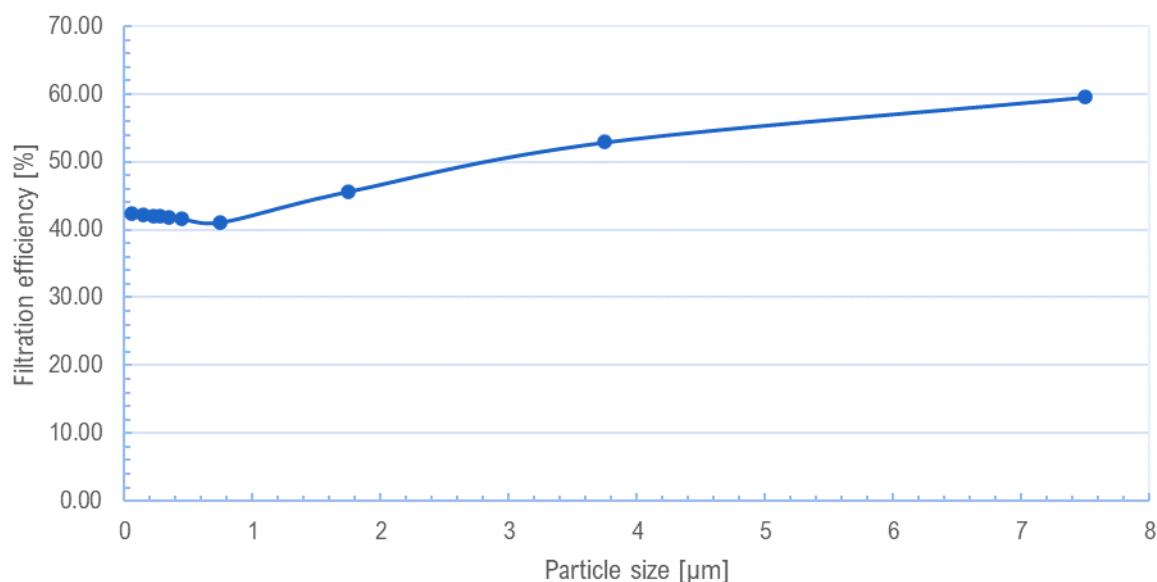
In the project, the filter efficiencies were measured for each ePM particle class according to the updated standard. This means, however, that except for the first class, filter efficiencies for different sizes are accumulated. The removal efficiency of ePM10 also consists of the efficiencies for ePM2.5 and ePM1. Therefore, one should keep in mind that for some bin sizes these efficiencies could prove to be an over- or underestimation.

The efficiencies for ePM0.1, ePM1, ePM2.5 and ePM10 were assumed to be the inputs for their largest size diameters. Interpolation was used to determine efficiencies for the average diameter of each bin size. The resulting efficiencies can be observed in graph 3 and table 7.

¹ Data derived from the research project Out2In conducted by Buildwise with the financial support of Innoviris Brussels

	dp [μm]	Efficiency [%]
Bin size PM0.01-0.1	0.055	42.32
Bin size PM0.1-0.2	0.15	42.23
Bin size PM0.2-0.25	0.225	42.08
Bin size PM0.25-0.3	0.275	41.98
Bin size PM0.3-0.4	0.35	41.84
Bin size PM0.4-0.5	0.45	41.64
Bin size PM0.5-1	0.75	41.06
Bin size PM1-2.5	1.75	45.62
Bin size PM2.5-5	3.75	52.89
Bin size PM5-10	7.5	59.55

Table 7: Filter efficiency for each bin size for a coarse 60% FP filter (Source: Van Herreweghe et al., 2022).



Graph 3 : Efficiency curve of a coarse 60% FP filter derived from data of the Out2In project of Van Herreweghe et al. (2022).

The different efficiency/particle size couples are now used as input for the filter in Dymola. To validate the filter, a test model was developed. The graphical implementation of this test model can be seen in Figure 24. The filter efficiencies were used as filter input, while the outdoor PM concentrations were added to the outside air. Simulating this model, resulted in the expected particle concentrations.

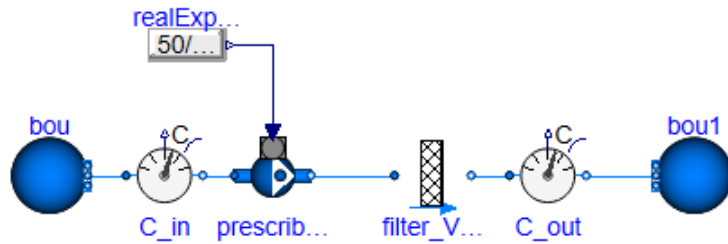


Figure 24: Modelica test model with previously developed filter component.

8.4. Multi-zone model

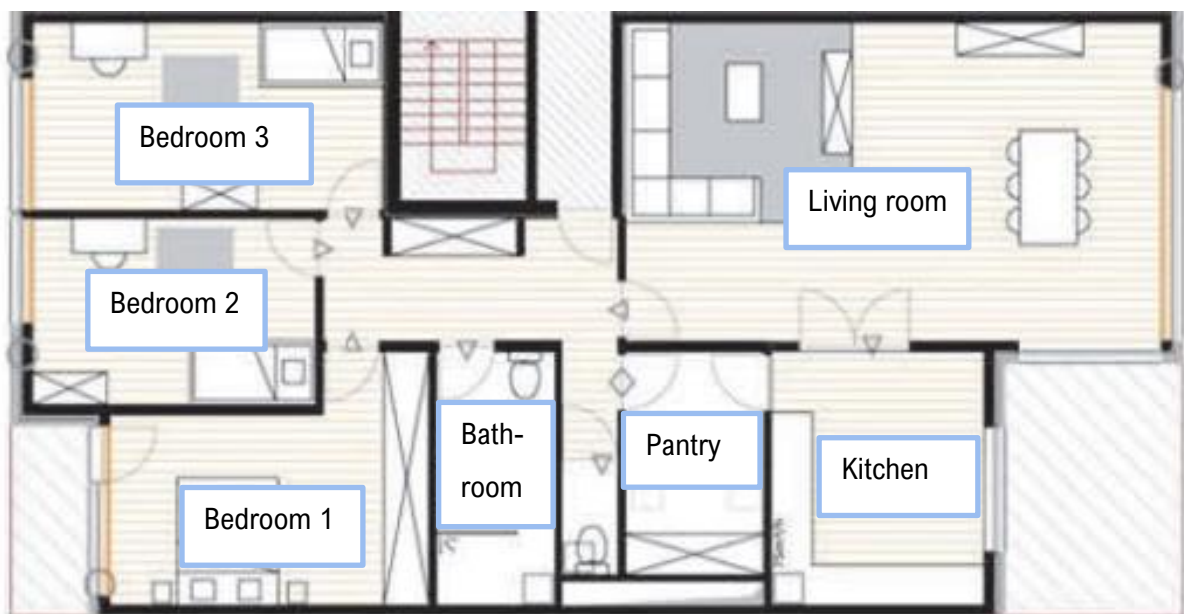


Figure 25: Floorplan of the reference apartment of the multi-zone model (Source: De Jonge, 2023).

In order to be able to test the filter in a real-life environment, this filter needs to be tested as part of a larger virtual system that encompasses a living area closely related to reality. For these tests, the multizone model from the dissertation of dr. ir. arch. Klaas De Jonge (2023) was used. In this dissertation: ‘Holistic Performance Assessment of Residential Ventilation Systems’ the 3-room apartment in figure 25, was modelled in Dymola using Modelica software. This model, which will be referred to as the multi-zone model. The existing model was modified and simplified for the different case studies of this dissertation to allow the testing of different conditions and filters.

The apartment consists of 3 bedrooms, a living room with adjacent kitchen, a back-pantry that also contains a separate toilet and is also accessible from the corridor, a bathroom and 3 bedrooms connected to the corridor. There is a communal stairway that contains the front door for this apartment. The north

of the apartment is placed at the long side of the living room. The bedrooms are numbered, with the one in the West being the first (the masters) and the one in the North the third. The apartment is located in a 6-story apartment building and has a total living area of 119.4 m² for an occupation of 1 to six people.

8.5. Base scenario

In order to evaluate the influence of different factors on the indoor PM concentration, the multi-zone model was adapted to encompass particulate matter.

This base scenario does not include any occupants, the apartment is empty but functional as if it were not. By making this simplification, particulate matter generated by the activities of the occupants can be excluded in this first step. Additionally, the windows of the apartment stay closed. Air cleaning will be provided by a D- ventilation system (both mechanical exhaust and supply) that works at 100% of the time with a constant flow rate. There will not be a filter present in this ventilation system. The supply and exhaust values and locations can be found in Table 8. The airtightness of the building is assumed to have a default value (De Jonge, 2023).

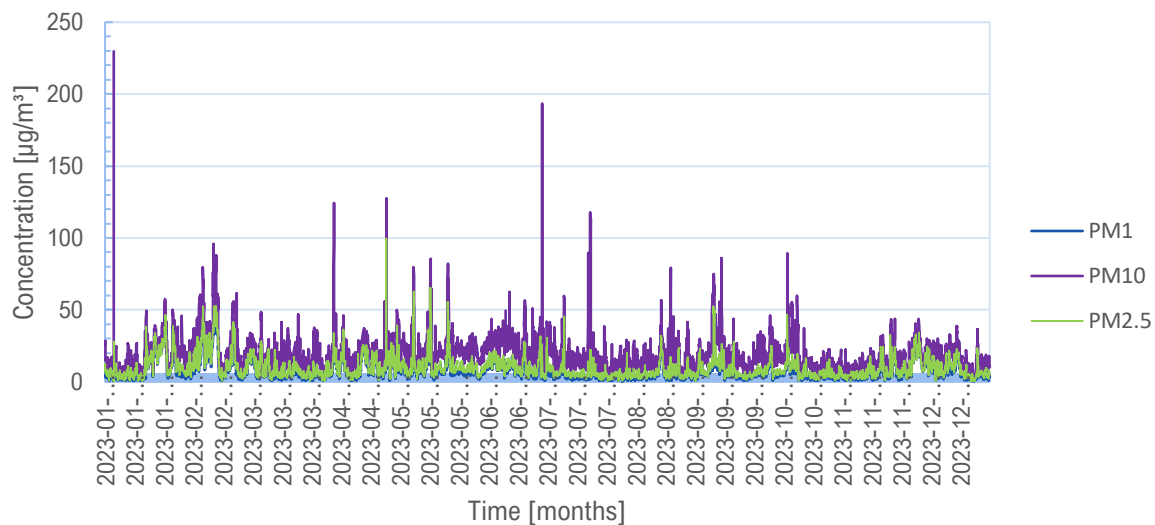
	Zone	Function	Flow rate [m ³ /h]
Supply	1	Bedroom 1	43.92
	2	Bedroom 2	35.26
	3	Bedroom 3	38.88
	4	Living space	108.32
Exhaust	5	Hall	16
	6	Kitchen	60
	7	Bathroom	60
	8	Toilet	30
	9	Service room	60

Table 8: Ventilation flow rates per zone (Source: De Jonge, 2023).

In order to adapt the existing multi-zone model, PM tracers were added to the air medium and the outside air. Those tracers correspond to the different bin sizes determined in chapter 8.2.4. To each tracer a decay rate was allocated based on literature. Since the existing literature maintains few particle size divisions, many tracers share the same decay rate. For the bin sizes below 2.5µm, the estimated decay rate from the research of Liu Sun & Lance Wallace (2020) was used, which was 1.1/h. For bin sizes between PM2.5 and PM10, a value of 1.51/h was obtained from the research of Kang et al. (2019).

The outdoor concentrations of PM for the different bin sizes are variable. These concentrations are obtained via irCELine.be. Here, the variation in PM concentration for PM1, PM2.5, PM10 can be downloaded for different measurement stations in Belgium. Values were downloaded for the year 2023 in $\mu\text{g}/\text{m}^3$ with hourly intervals. The weather file of the multizone model corresponds to Uccle. However, since more detailed concentrations could be found in Vilvoorde, these were preferred. The geographical difference between the two is minimal.

It is obvious that this data imposes some flaws. Because data is only available for existing PM divisions, some divisions also partially encompass others (e.g. PM10 also includes PM2.5). Technically an interpolation could be made, but because the size distribution and concentration of PM0.1 is unknown, and very geographically variable, this will not give an accurate idea of the subsequent health effects. Using the best corresponding data for each bin size, can be seen as a worst-case scenario in which all particulate matter of its size fraction belongs to that particular bin size. If at one point, more data should become available, this should be revised more accurately. The variation in PM concentration throughout the year in Vilvoorde can be observed in graph 4.



Graph 4: Variation of PM throughout the year 2023 in Vilvoorde

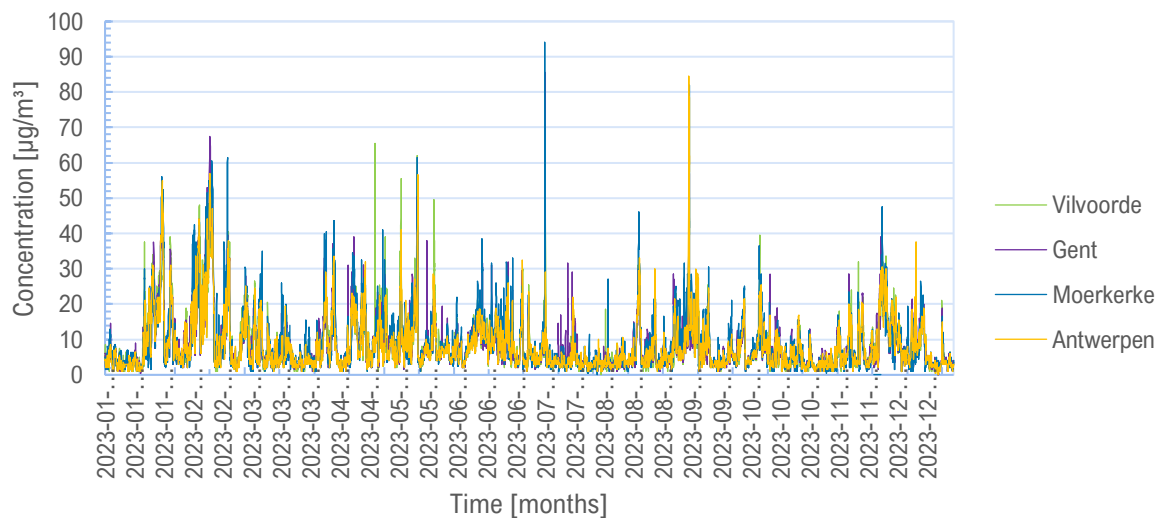
In order to add this varying concentration to the Dymola model, a table was constructed with the concentrations for each of the tracers dependent on time. In case of missing values, interpolation was used. This table was added to a combiTimeTable component and connected to the ventilation system and the infiltration air. The results of this base scenario will be used as a baseline for the analysis of the results from the further case studies.

9.

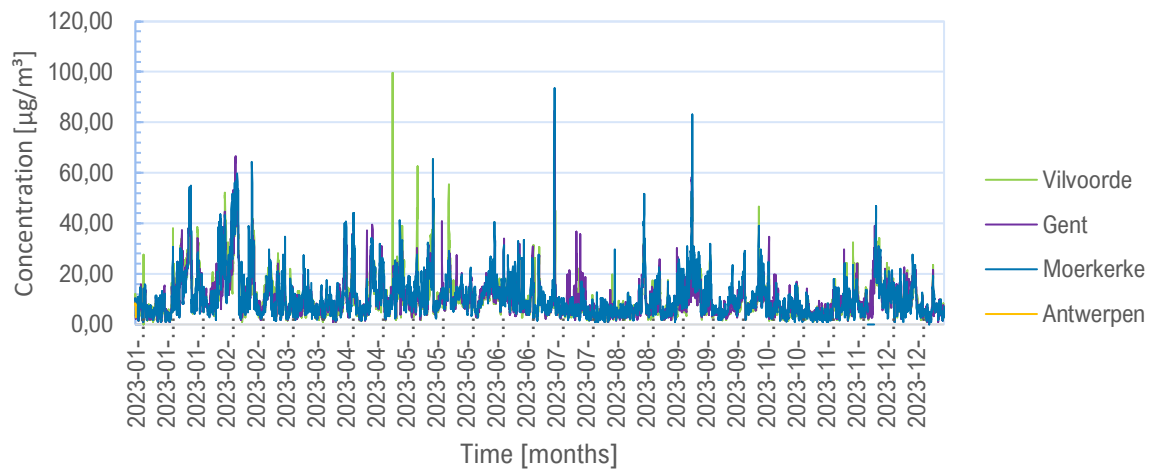
Case studies

9.1. Case study 1 : Outdoor PM concentration

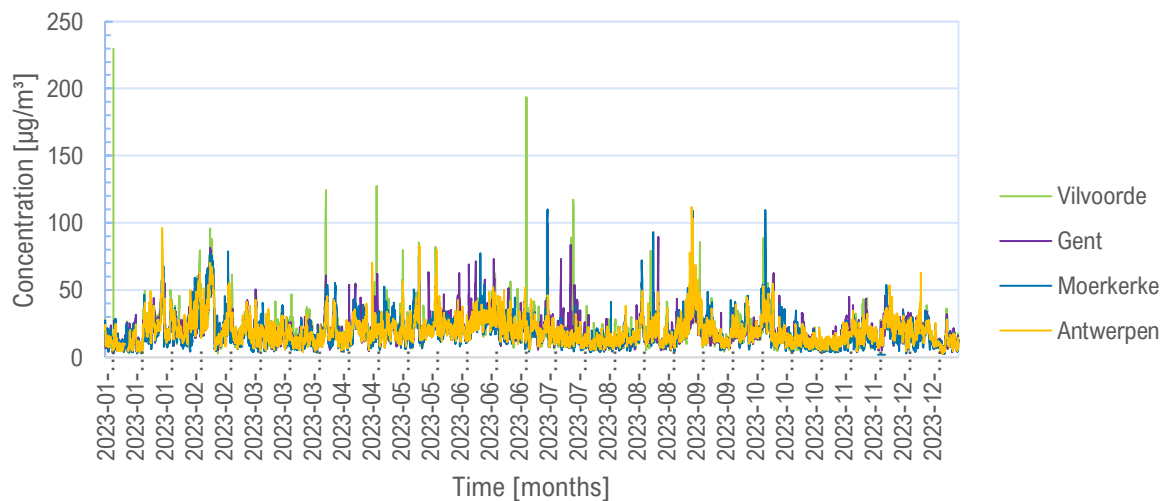
In this first case study, a few other outdoor PM concentrations will be simulated. The measuring station of Vilvoorde is situated just outside the ring road of Brussels, next to a busy road. In order to determine the geographical diversity of PM, three additional measuring stations are selected. The first, located in the centre of Gent, the second located on the Antwerp ring road and the third located in a more rural area in Moerkerke. Graphs 5, 6 and 7 provide an initial insight in the differences between the locations for different PM size ranges.



Graph 5: Comparison of the outdoor PM₁ concentration in different environments



Graph 6: Comparison of the outdoor PM2.5 concentration in different environments



Graph 7: Comparison of the outdoor PM10 concentration in different environments

When observing these three diagrams, the overall similarity between them stands out. For PM1, the highest peaks are in Moerkerke and Antwerp. Although there are also some high points in Gent and Vilvoorde. In case of PM2.5 the highest values belong to Vilvoorde, but concentrations in Moerkerke are again very present. For PM10, highest peaks are observed for Vilvoorde, Gent and Antwerp. In table 9 the different average concentrations and peaks are given for each location. The mean values are quite similar for all locations.

	Vilvoorde			Ghent			Antwerp			Moerkerke		
	PM1	PM2.5	PM10	PM1	PM2.5	PM10	PM1	PM2.5	PM10	PM1	PM2.5	PM10
	µg/m³			µg/m³			µg/m³			µg/m³		
Average	8.07	9.6	18.5	8.27	9.96	18.9	8.1	9.8	19.3	8.8	10.4	18.2
max	65.5	99.64	229.9	85.5	84.6	99.9	84.5	84.17	111.4	94	93.53	110.4
min	0	0.47	2.9	0	0.94	2.9	0	0	2.9	0	0	0

Table 8: Average, minimal and maximal concentrations for Vilvoorde and Ghent.

Some slight differences can be seen. The highest average PM1 and PM2.5 concentration is measured in Moerkerke, before Ghent and then Vilvoorde and Antwerp, which have a remarkably similar concentration. PM10 is most abundantly present in Antwerp, before Gent, Vilvoorde and Antwerp. This seems to speak against some conclusions of the literary study. It seems like the rural area contains the finest particles, while the traffic areas have the largest concentrations of PM2.5 and PM10. However, this could also be due to a few remarkably high peaks. Highest peaks for PM2.5 and PM10 are found in Vilvoorde, while the highest PM1 peak can again be found in Moerkerke. The lowest values are measured in Moerkerke as well.

9.2. Case study 2 : Filters

Simulating the overall building with different kinds of filters, gives an accurate prediction of the influence of filters as opposed to a ventilation system without filters. In order to test this, influence the filter model is added to the overall multi-zone model in the ventilation system between the source and pump. Figure 26 shows the graphical representation of this addition to the already existing ventilation system.

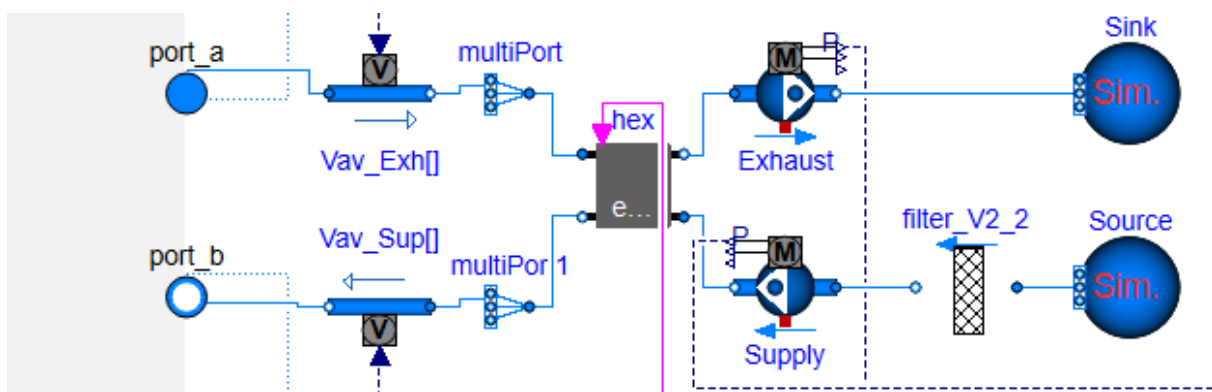


Figure 26: Ventilation system D multi-zone model with addition of the filter.

In total, 3 different kinds of filters will be tested. In order to determine the filter efficiency, two different methods will be used.

9.2.1. Coarse filters

The data from the Out2In project contains filter efficiencies for a coarse 60% filter and an ePM1 55% filter (both folded panel). The efficiency for each bin size, as determined via the method described in 8.3.4, are inserted in the filter model. These values can be found in table 9.

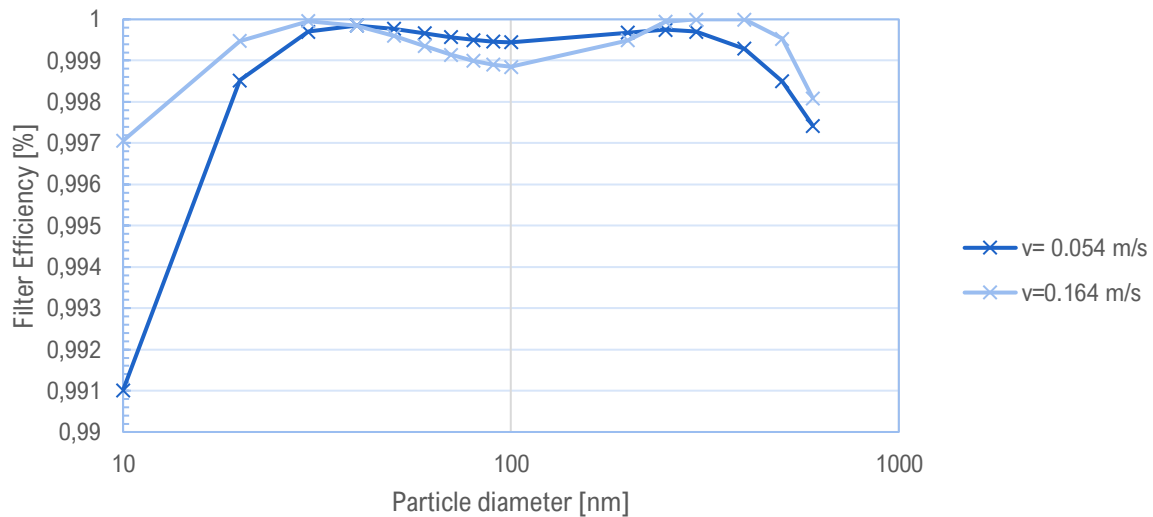
	Coarse 60% Filter	ePM1 55% filter
Bin size PM0.01-0.1	42.32	53.58
Bin size PM0.1-0.2	42.23	52.33
Bin size PM0.2-0.25	42.08	50.46
Bin size PM0.25-0.3	41.98	49.20
Bin size PM0.3-0.4	41.84	47.33
Bin size PM0.4-0.5	41.64	44.83
Bin size PM0.5-1	41.06	37.33
Bin size PM1-2.5	45.62	36.17
Bin size PM2.5-5	52.89	44.13
Bin size PM5-10	59.55	52.74

Table 9: Filter efficiencies for a Coarse 60% and ePM1 55% filter based on data from the Out2In research.

9.2.2. High efficiency filters

For the higher efficiency filters, a more analytical approach needs to be used. These types of filters are not commonly used in residential ventilation systems and were therefore not part of the Out2In study. In order to obtain efficiencies for these filters, the research by Chen, Ji and Zhao (2019) was used. However, a remark needs to be made. The filtration efficiency formula calculated by Chen, Ji, and Zhao (2019), is only applicable in the range from 0.0146 μm or 14.6 nm to 0.6153 μm or 615.3 nm. The determined bin sizes go up to 10 μm . This leaves quite a large range of PM (0.6153-10 μm) unaccounted for. The effect of this can however be minimized, since particulate matter with a diameter larger than 0.5 μm only accounts for 0.044 of the PM pollutant load in the Brussels outdoor air. An assumption can be made for these larger particle sizes (Van Herreweghe et al., 2019).

Furthermore, the filters from the Out2In project were tested at an airflow velocity of 0.5m/s, while the mathematical obtained efficiencies for the other filters were based on a velocity of 0.054 or 0.164 m/s. This difference will most likely be significant, as discussed before. Unfortunately, this is an expected disadvantage of the current testing standard that does not provide mandatory velocities. This is a real problem that is also present when comparing filters in real life conditions. In Graph 8, efficiencies for both possible velocities for a H13 according to the research by Chen, Ji, and Zhao (2019) can be found. This shows the difference varying testing velocities can make. Both the though, start and end positions are remarkably different. At the current time, no solution was found to overcome this problem. The efficiencies for this research were assumed to be the same at the velocity of the coarse and fine filter.



Graph 8: Size dependant efficiency for a H13 filter.

Since the second velocity is closest to the one used in Out2In, these efficiencies will be used in the next simulation for a H13 filter. They can be found in table 10. For the bin sizes exceeding the tested sizes, a linear continuation was assumed, with a maximum value of 0.99999.

	H13 filter
Bin size PM0.01-0.1	0.99947
Bin size PM0.1-0.2	0.99902
Bin size PM0.2-0.25	0.99973
Bin size PM0.25-0.3	0.99999
Bin size PM0.3-0.4	0.99999
Bin size PM0.4-0.5	0.99999
Bin size PM0.5-1	0.99999
Bin size PM1-2.5	0.99999
Bin size PM2.5-5	0.99999
Bin size PM5-10	0.99999

Table 10: Filter efficiencies used in the simulation for a H13 filter.

9.2.3. Other pollutants

The use of filters will also influence some of the other tracers. Firstly, tracers that are not present in the outdoor air, can also not be subject to filtration. For these tracers an efficiency of zero will be assumed. This leaves only CO_2 , NO_2 and O_3 . For O_3 a filtration efficiency of 10% was assumed. The effect of the filters on NO_2 and CO_2 was considered negligible (Belias & Licina, 2024).

9.3. Case study 3 : Indoor PM source

In this next case study, occupants are introduced. Even though the occupants themselves don't emit particulate matter directly, the activities they participate in, do. Several indoor sources of particulate matter could be considered, as described in the literary research. However, the most important source of particulate matter in the indoor environment is cooking. This is a general, universal PM source that can be applied to all households.

The emitted PM concentration is dependent on the specific meals that are made, the equipment that is used and the duration of the cooking activities. Some activities will emit a lot more PM than others. The kind of heat source is also important (Kim et al., 2015).

The difference in the influence of cooking is investigated when there is no filter, a coarse filter, or a HEPA filter present. These are the best and worst filter tested in previous case study.

9.3.1. Modelica implementation

First step towards the implementation of cooking related particulate matter is the construction of a cooking schedule. This schedule is based on schedule 127 of the existing occupancy schedules in the multi-zone model (De Jonge, 2023). This schedule represents a 4-person family, with two children that go to school and one parent that goes to work. The remaining parent is a stay-at-home parent. The cooking schedule is based on the schedule of the stay-at-home parent. It is assumed that the larger meals of the day are cooked by this parent during the periods he/she is in the kitchen for a longer period of time.

Following the method of O'Leary et al. (2019), a worst-case scenario is considered, where 3 warm meals are cooked per day. The emission rate and duration for different meals can be found in table 11. Breakfast will consist of some toasted bread. Lunch will be a pasta Bolognese and to end the day, the occupants will enjoy a chicken and vegetable stir-fry.

Meal	Meal type	Average Emission Rate	Duration
		[µg/min]	[minutes]
Breakfast	Toast	220	3.85
Lunch	Pasta Bolognese	1900	28
Dinner	Chicken and vegetables stir-fry	3200	17

Table 11: Emission rate and duration for the selected meals (Source: O'Leary et al., 2019)

These values and the occupant schedule of the adults who cooks, can now be combined into a PM source schedule for the kitchen. In this schedule, preparing lunch will be the average situation, which will correspond to an emission value of 1. For the emission rate of the other meals a conversion factor will be used. These values can be found below in table 12. Every time the stay-at-home parent is in the kitchen for a longer period, the conversion factor of the corresponding meal will be placed next to the entry time, the emission will continue for the duration given in table 11. This schedule is then linked to the moisture production of the cooking activity to represent the connection between humidity and settling rate.

Meal	Conversion factor
Breakfast	0.116
Lunch	1
Dinner	1.684

Table 12: Conversion factors for the PM source schedule.

Furthermore, the emission rate of the pasta is converted from $\mu\text{g}/\text{min}$ to kg/s and entered in Dymola as the source emission rate. The conversion to source strengths for the other meals happens in the PM sources schedule via the conversion factors. However, to become the total emission rate for each bin size some extra calculations need to be made.

Firstly, there is also a range hood present in the kitchen. This hood is connected to the humidity schedule of the kitchen and turns on when the occupant is cooking. The capture efficiency of this hood is 70%. This percentage will be reduced from the source emission rate.

Secondly, the total emission rate of the cooking related source is not divided equally over all bin sizes. A cooking related size distribution can be calculated using the research by Pikmann et al. (2023). Here, cooking was found to have a lognormal distribution with a standard deviation of 5nm. For the mode diameter, values from the same research were used. For the lunch, this value was 40 nm. Via the formula of the lognormal distribution and the mean particle sizes for each diameter a percentage of the total emission was calculated for each bin size. In this research only PM_{2.5} is considered. Therefore, the higher sizes fractions will be assumed to not take part in the emission of PM. This is probably very close to reality, since 99.999999999999450 % of the particles emitted by the activity were situated in the lowest bin size. The resulting values can be found below in table 13.

Bin Size	Percentage of emission [%]
PM0.01_0.1	0.999999999999450
PM0.1_0.2	5.5341038E-14
PM0.2_0.25	1.2221436E-18
PM0.25_0.3	6.0361890E-21
PM0.3_0.4	9.6021842E-24
PM0.4_0.5	1.0450616E-26
PM0.5_1	6.2603217E-33
PM1_2.5	5.4037982E-44
PM2.5_5	0
PM5_10	0

Table 13: Percentage of total PM emission by cooking per bin size.

These values are multiplied by the emission rate of the pasta meal for each bin size. After calculating the 30% that is not removed by the range hood, this is multiplied by the PM source schedule and moisture relations. This results in the PM emission per bin size and can be seen as a good indication of the influence of cooking on the indoor PM concentration.

A graphical implementation of these components can be seen in figure 27.

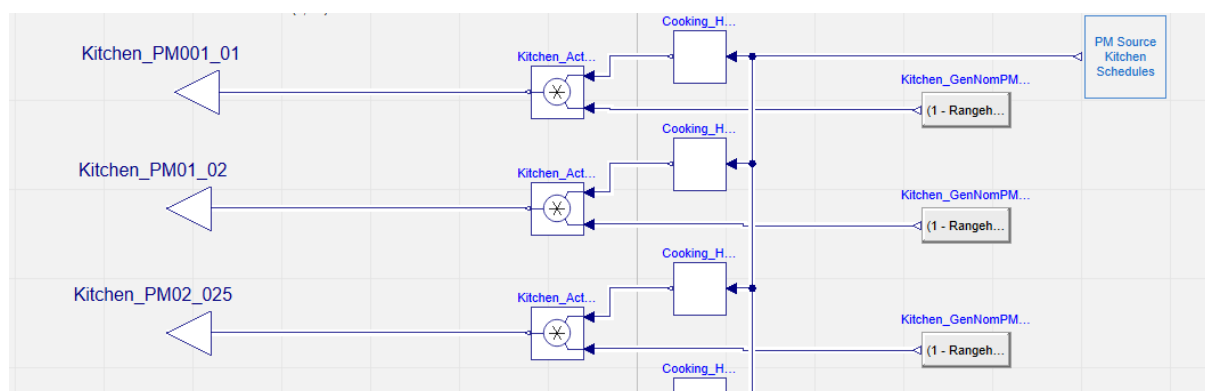


Figure 27: Graphical implementation of cooking-related PM in the occupancy model for some bin sizes.

9.4. Case study 4 : Demand based ventilation

In previous case studies, the ventilation system was operating 100 % of the time. But since an occupant has been added to the model, a demand-based system can also be tested. In the dissertation by De Jonge (2023) multiple demand-controlled ventilation systems were developed. For this research the D5 system will be simulated. A principle sketch off this system can be found in figure 28.

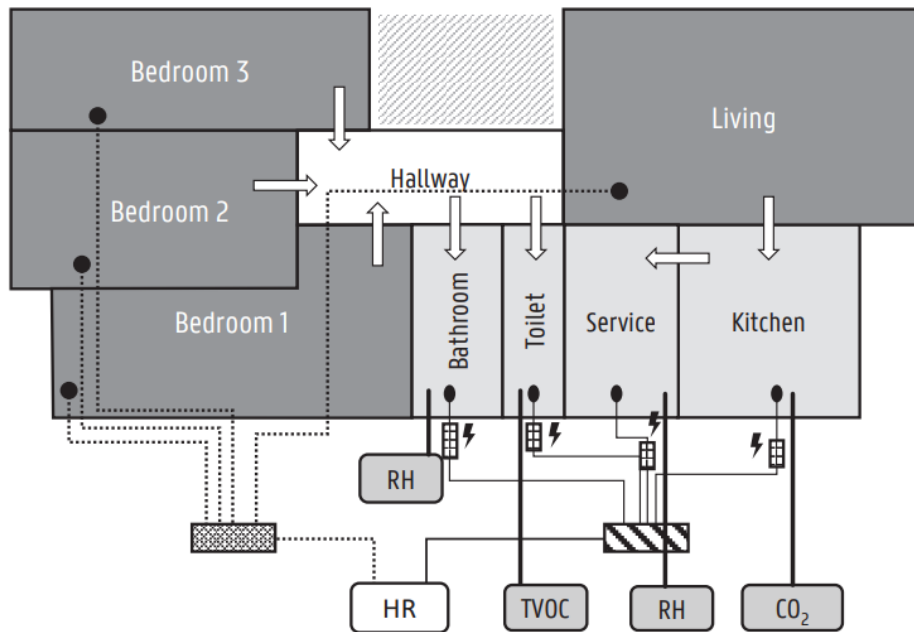


Figure 28: Principle and operation of ventilation system D5 (Source: De Jonge, 2023).

In this system, the ventilation flow rate is adjusted to the needs of the wet rooms (kitchen and bathroom). This results in a better energy performance, but it does not consider particulate matter. Subsequently, a higher humidity correlates to a faster settlement of particles, so this might not be beneficial for the PM count in the wet rooms either.

9.5. Case study 5 : Window opening

In this next case study, the influence of window opening and therefore natural ventilation is investigated. According to literary reports, this is one of the largest influences on the indoor particulate matter concentration. When a window is opened, outside air can directly enter a space, without filtration. Therefore, the particulate matter concentration that enters the building via opened windows, as opposed to via a ventilation system with filter, could be significant.

This will be investigated by opening certain windows in the multi-zone model for certain time periods. The location of the windows can be observed in figure 29. The windows are located on the North, South and West facades.



Figure 29: Floorplan multi-zone model with indicated windows (Source: De Jonge, 2023).

9.5.1. Opening schedule

In the base scenario and all previously investigated case studies, these windows were considered to be closed. The only ventilation present was contributed to mechanical ventilation and air leakage. In order to investigate the influence of open windows, some of these windows will be opened based on a schedule. This schedule was constructed based on the doctoral dissertation of Silke Verbruggen (2021) “Window Use Habits as an Example of Habitual Occupant Behaviour in Residential Buildings”. Based on surveys, the window-use habits in Belgium are discussed in this dissertation. This window use is not only dependent on habits, but also on seasons. Therefore, a division between winter and summer use will be made. November, December, January, February, March, and April will follow the winter schedule. While May, June, July, August, September, and October will follow the summer schedule. Concretely, the adapted schedule for each window can be found in table 14.

Window	number	Opening Winter	Opening Summer
Large window living room	1	Always closed	Always closed
Small window living room	2	Always closed	Open 9-5
Window kitchen	3	Always closed	Always open
Small window bedroom 1	4	Open 8-9	Always open
Big window bedroom 1	5	Open 8-9	Always open
Window bedroom 2	6	Open 8-9	Always open
Window bedroom 3	7	Open 8-9	Always open

Table 14: Window numbers and opening schedule.

9.5.2. Window component

Additionally, in order to implement this schedule for each window, two specific components have to be made. These components are based on components developed in the same research from Silke Verbruggen (2021). A schematical representation of this component can be seen in figure 30. The window-like component represents the airflow originating from a window through the equation of Phaff. This contains the open area of the window through which it lets in outside air. This component is then linked to the window use, so it only exchanges air when the window is opened. The resulting air flow is then assigned to the air flow of its corresponding flow with air flow pumps.

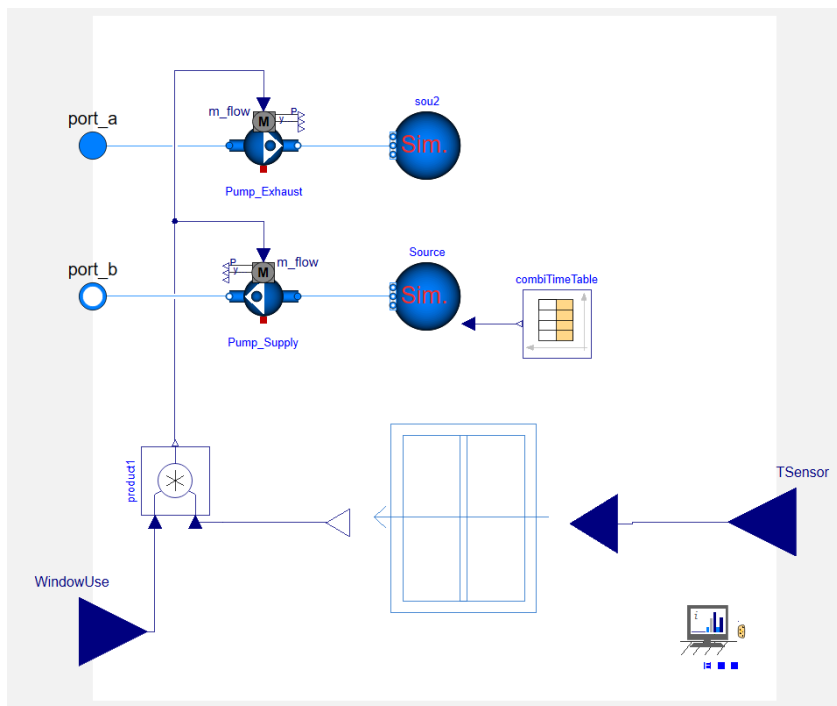


Figure 30: Air flow component slide window.

An important parameter for this component is the open surface area of the window. This parameter is dependent on the type of window, whether it is a tilted, turning or sliding window. According to the window-habits survey, tilted windows are the most commonly used. This is, however, also dependent on the kind of room. Furthermore, access to the terraces can sometimes happen through one of the windows. Based on this information and the floorplan, an assumption can be made. The windows in the bedrooms are considered to be tilted windows. The living room window adjacent to the terrace will be a sliding window. The other living room window will be closed permanently, due to its size. Lastly, the kitchen window will be a turning window. The update schedule can be found in table 15.

Window	number	type	Opening Winter	Opening Summer
Large window living room	1	Closed	Always closed	Always closed
Small window living room	2	Slide	Always closed	Open 9-5
Window kitchen	3	Turn	Always closed	Always open
Small window bedroom 1	4	Tilted	Open 8-9	Always open
Big window bedroom 1	5	Tilted	Open 8-9	Always open
Window bedroom 2	6	Tilted	Open 8-9	Always open
Window bedroom 3	7	Tilted	Open 8-9	Always open

Tabel 15: Updated window schedule

Next step is the implementation of the window component in the structure model. In order to do this, the type of component is connected to a corresponding time schedule for each window. Subsequently, the Tsensor is also connected to this component. Ultimately, the air flow due to the component is connected to the airflow of the corresponding zone.

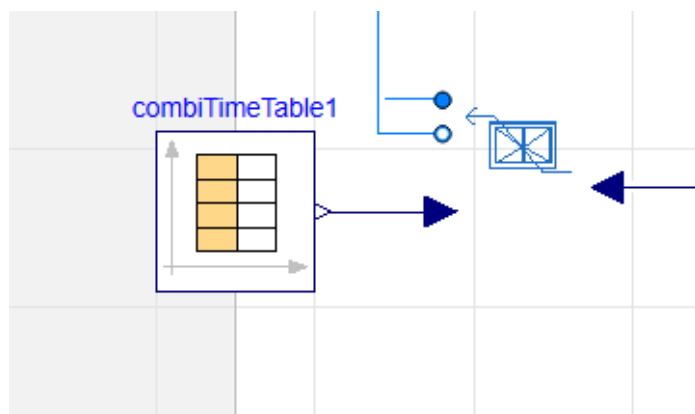


Figure 31: Schematic Implementation window component in the structure model

9.6. Case study 6 : Filter lifetime

9.6.1. Varying efficiency

In this next case study, the lifetime of a filter is investigated, when clogging occurs. During the clogging of filter, the efficiency changes as well. For a coarse 60% filter, the effect of particle accumulation on filter efficiency for different size ranges can be overviewed in figure 32 from the Out2In research by Herreweghe et al. (2022).

A

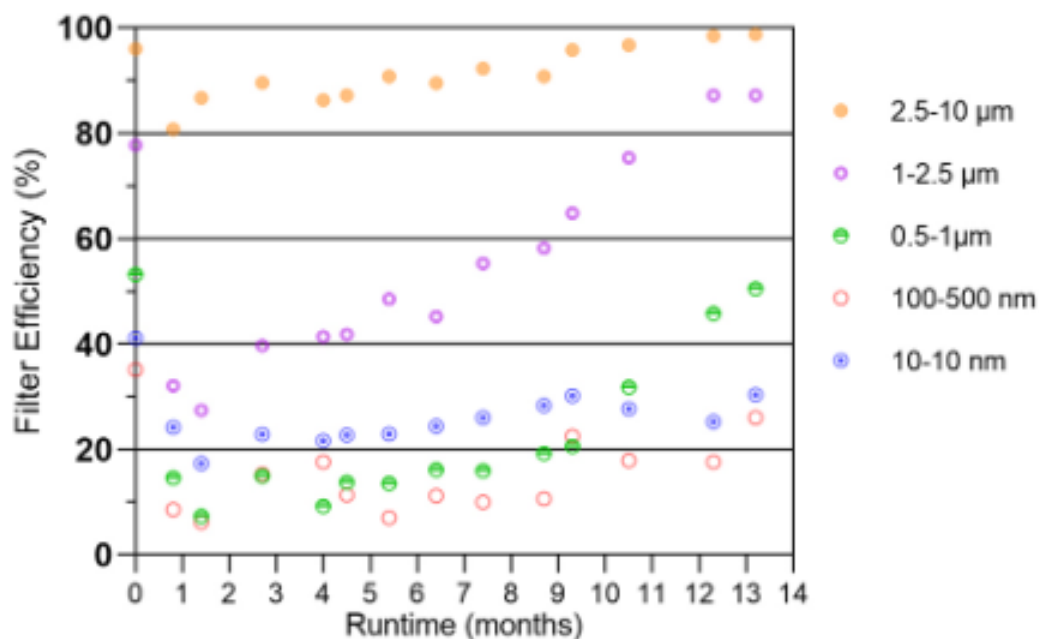
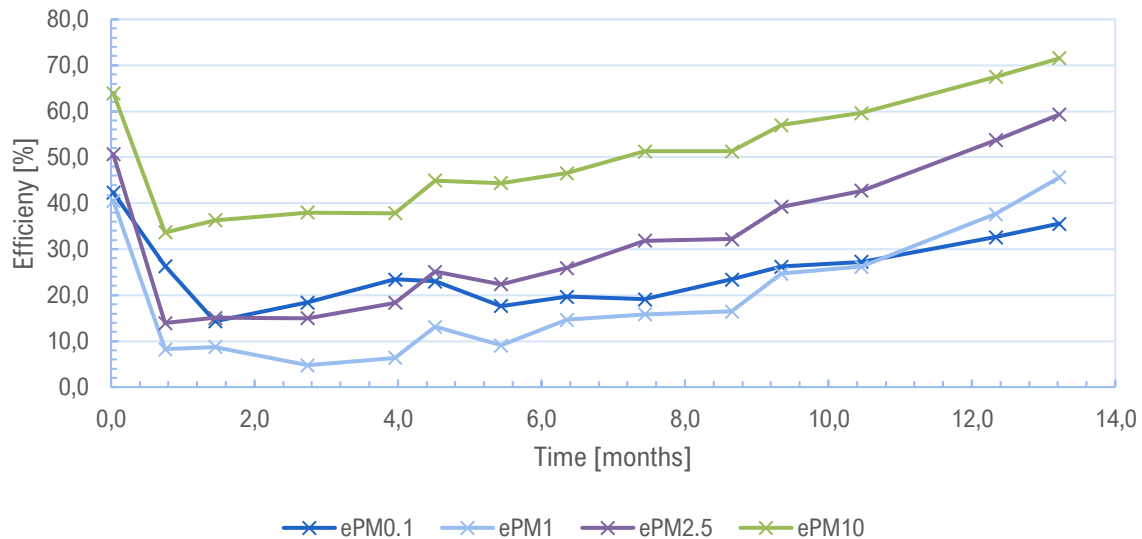


Figure 32: Filter efficiency during clogging of a coarse Folded panel filter (Source: Herreweghe et al., 2022)

There is a large gap between the efficiency of a clean filter and that of a filter that has been used for a month. This can be attributed to the static electric charge that the filter medium can receive during its production. This results in an extra filter mechanism in the form of electrostatic interaction. When particles get stuck in the filter media, this effect decreases till it eventually disappears (Herreweghe et al., 2022).

After this initial drop however, the efficiency increases again and ultimately exceeds the initial efficiency. In case of a coarse 60% filter, the filter efficiency variation can be directly plotted based on the data from the research project Out2In.² In this research the varying efficiency was measured about once a month for a period of 13 months (Herreweghe et al., 2022). The given efficiencies for the coarse filter and different PM size fractions can be seen in graph 9.

² Data derived from the research project Out2In conducted by Buildwise with the financial support of Innoviris Brussels



Graph 9: Visual representation of the filter efficiency variation throughout time for a Coarse 60% filter (Source: Herreweghe et al., 2022)

To enter this varying efficiency in Dymola, some simplifications need to be made. Firstly, the initial efficiency drop is discarded. Using the lowest efficiency as the starting point makes it possible to fit a curve through the various efficiencies. Additionally, this could also be seen as the worst-case scenario for the indoor environment, leaving through the most particulate matter. Furthermore, the efficiencies from the Out2In project were observed for 4 particle ranges according to the standard NBN EN ISO 16890. This means, unfortunately, that values for some of the bin sizes, are the same. Interpolation will be used to calculate the efficiencies that corresponded to each bin.

In case of a H13 filter, the situation is a bit more difficult. Here, the efficiency cannot be based on real life data. The initial efficiencies were determined in the second case study. This resulted in efficiencies very close to 100%. For the efficiency increase a similar incline was assumed as for the coarse filter, with the same maximum value of 0.99999. The efficiency increase is therefore a lot less significant.

It is evident that this method has some flaws. Since the efficiencies are analytically determined, they will not be an exact match to the real efficiencies. The efficiency curves were also not determined based on real life data or an analytical formula but on a similar curve of a very different filter. This could be different in real testing conditions as well. Additionally, since there wasn't a lowest value of the efficiency to use as a starting point, there is also no efficiency drop. However, since HEPA filters are very high efficiency filters and the difference between different bin sizes is very small, these flaws will most likely have a minimal influence.

9.6.2. Adapted filter component

Subsequently, a feedback loop for the filter efficiency is added to the previously developed filter component. Here the mass that remains in the filter (=clogging) is added together via an integrator and then reformed to an efficiency curve by using different gains and an atangens function. By placing this filter in a test model, the efficiency curves can be constructed. This test model contained the outdoor concentrations of the Out2In project, in order to be able to mimic the real-life efficiency curves (Herreweghe et al., 2019). With the help of sweep parameter, the correct combination of gains could be found for each particle range and a fitting efficiency curve could be obtained. A schematic of this composition can be found in figure 33.

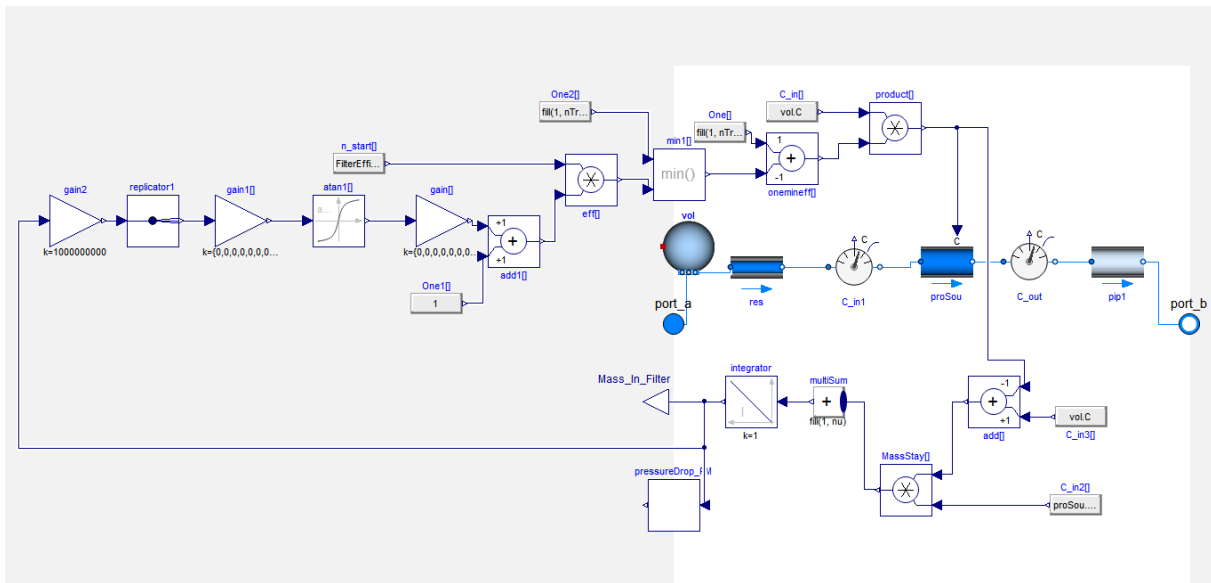


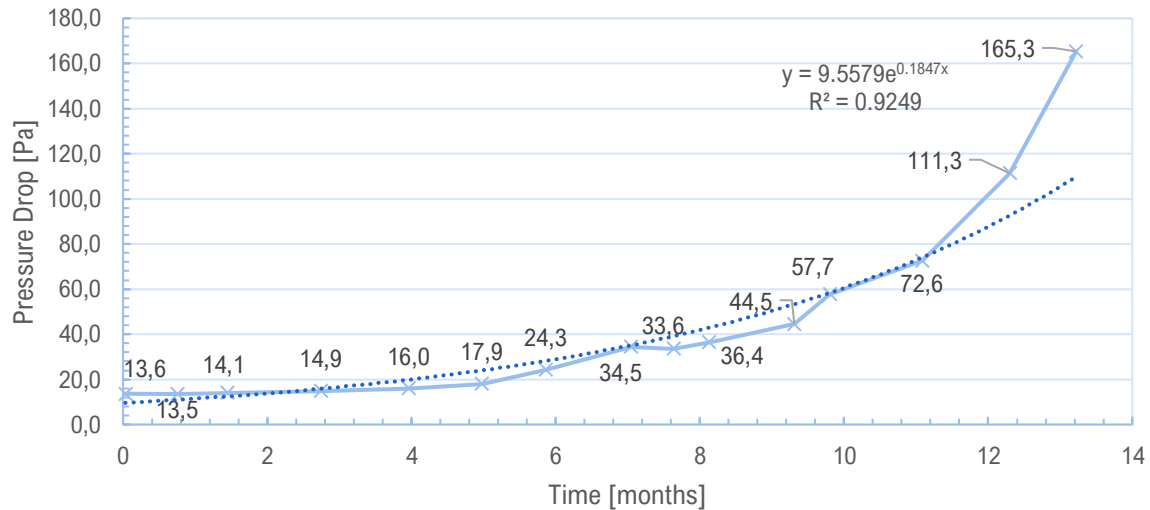
Figure 33: Dymola representation of a filter with varying efficiency

9.6.3. Pressure Drop

Simulating the model with varying efficiencies also shows the particle mass that stays behind in the filter. This can be linked to the pressure drop and subsequent lifetime of the filter. In order to make this possible, the assumption is made here that there is no compaction between the particles. There is also no division between different particle sizes in terms of clogging. Each particle is assumed to have the same contribution to the clogging phenomenon. To determine the influence of the particle accumulation in the simulation environment of Dymola, a graph needs to be compiled that links pressure drop and particle load.

For a Coarse filter, this graph can be compiled based on the data from the Out2In project. Here, the pressure drop and particles removal efficiencies is measured at the same time. The initial pressure drop of a coarse 60% filter is 13.6 Pa. The pressure drop evolution for this filter can be found in graph 10.

This filter should be replaced when the pressure drop reaches 150 Pa. This graph shows that this is the case just before 13 months with the outdoor Brussels particle load and this specific flow rate of 150 m³/h (Herreweghe et al., 2022). Other flow rates will result in different pressure drops.



Graph 10: Pressure drop evolution for a coarse 60% filter.³

To use this data in the Modelica environment, the outdoor PM concentrations in particles/L from this project need to be converted to µg/m³. Therefore, some assumptions will need to be made. The particles are assumed to have a circular shape. According to the research by Pitz et al. (2003), the particle density varies strongly throughout the day. The mean density on an hourly basis was about 1400 to 1500 kg/m³. While the overall densities range from 1000 to 2500 kg/m³. The lowest values were measured in the morning and the highest in the afternoon. This is similar to the value used in the CONTAM software (1200 kg/m³). With an assumed particle density of 1400 kg/m³, the mass of one particle can be calculated as follows. For example, for the last bin size PM₅₋₁₀, the mean particle diameter is 7.5µm or the radius is 3.75 µm. This results in the following calculations.

$$\rho = \frac{m}{V}$$

$$m = \rho * V$$

³ Data derived from the research project Out2In conducted by Buildwise with the financial support of Innoviris Brussels

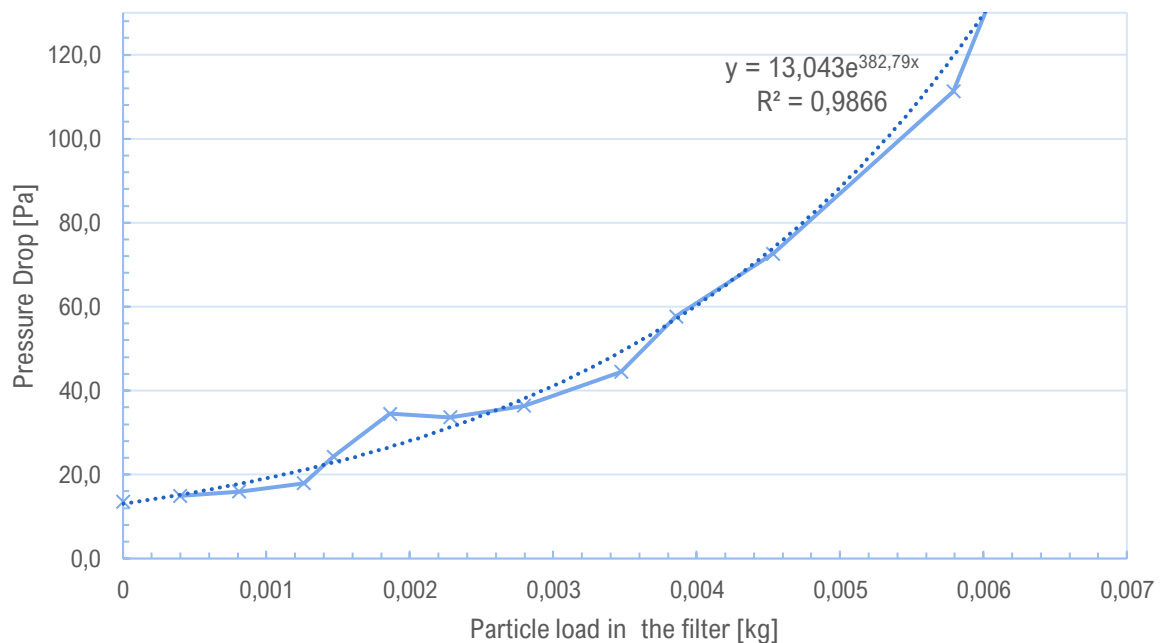
$$\begin{aligned}
m &= 1400 \frac{kg}{m^3} * \frac{4}{3} \pi r^3 \\
&= \frac{1400 kg}{m^3} * \frac{4}{3} \pi (3.75 * 10^{-6} m)^3 \\
&\approx 3.09 * 10^{-13} kg = 3.09 * 10^{-4} \mu g
\end{aligned}$$

This factor can be multiplied with the concentration of particles in one liter air and converted to a value in $\mu g/m^3$. In this case, the total mass for each m^3 is $7.62 \mu g/m^3$. A similar calculation can be executed for each bin size. The results can be found in table 16.

	Radius	Volume	mass/particle	mass/particle	number/L	total mass/l	total mass/ m^3
	m	m^3	kg	μg	l	$\mu g/l$	$\mu g/m^3$
PM0.01-0.1	2.75E-08	8.71E-23	1.22E-19	1.22E-10	5805556	7.08E-04	0.71
PM0.1-0.2	7.50E-08	1.77E-21	2.47E-18	2.47E-09	3155844	7.81E-03	7.81
PM0.2-0.25	1.13E-07	5.96E-21	8.35E-18	8.35E-09	884786	7.39E-03	7.39
PM0.25-0.3	1.38E-07	1.09E-20	1.52E-17	1.52E-08	590871	9.01E-03	9.01
PM0.3-0.4	1.75E-07	2.24E-20	3.14E-17	3.14E-08	150000	4.71E-03	4.71
PM0.4-0.5	2.25E-07	4.77E-20	6.68E-17	6.68E-08	30000	2.00E-03	2.00
PM0.5-1	3.75E-07	2.21E-19	3.09E-16	3.09E-07	2938	9.08E-04	0.91
PM1-2.5	8.75E-07	2.81E-18	3.93E-15	3.93E-06	1000	3.93E-03	3.93
PM2.5-5	1.88E-06	2.76E-17	3.87E-14	3.87E-05	219	8.46E-03	8.46
PM5-10	3.75E-06	2.21E-16	3.09E-13	3.09E-04	25	7.62E-03	7.62

Table 16: Conversion of PM concentrations from number/l to $\mu g/m^3$.

Now, the filter component with varying efficiency as described in 9.6.1, calculates the mass that remains in the filter. This ultimately results in a cumulative mass curve. From this, the mass in the filter at each measuring time can be determined. For the coarse filter, this results in point couples of masses and pressure drop at each measuring time. These points can then be used to construct graph 11 with pressure drop in function of the particle load. An exponential trend line gives a function for this correlation.



Graph 11: Pressure drop evolution for a rising particle load of a coarse 60% filter.

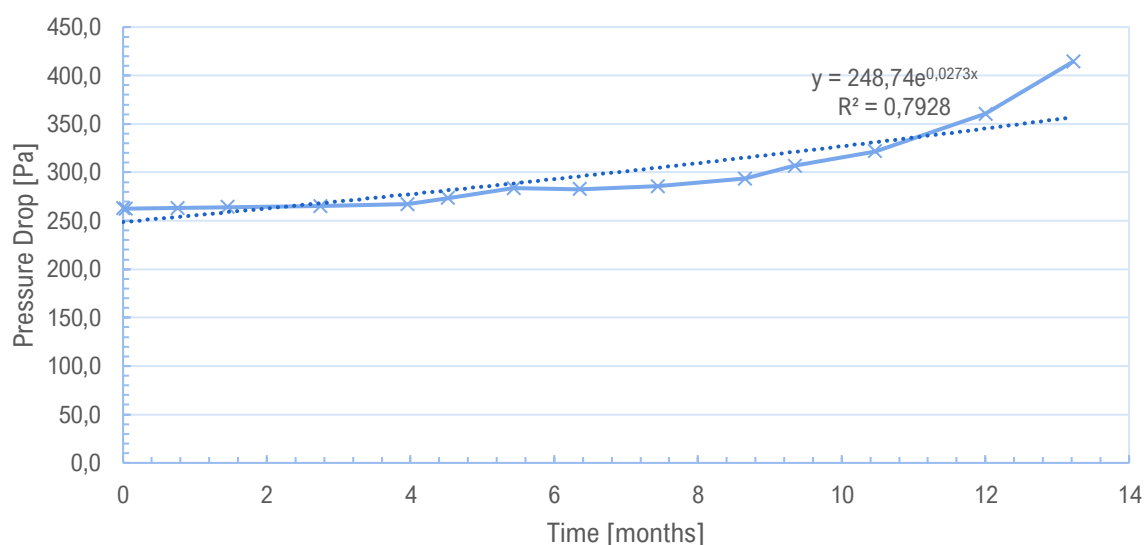
With the equation below, the pressure drop of a coarse 60% filter can be determined when the particle load is known. This correlation is added as a new component in Dymola, in order to calculate the pressure drop directly.

$$y = 13.043e^{382.79x}$$

Unfortunately, since parameters like the porosity and packing density are not tested in the testing standard, the more detailed pressure drop formulas can't be used in this case.

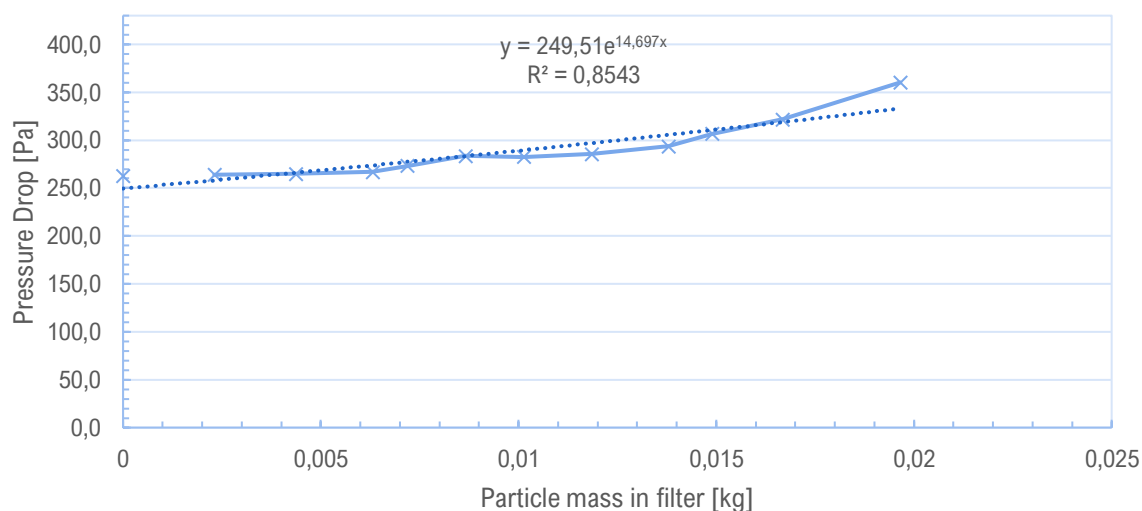
Again, in case of a HEPA filter, the problem is more complex. In order to be able to calculate a correlation between the particle load in the filter and the pressure drop, some variables are missing. Therefore, some additional assumptions will need to be made. For the initial and final pressure drop, the pressure drop of a HEPA filter on the market is used. The Viskron-Aire SuperCel 24 V Max HEPA filter experiences an initial pressure drop of 249 Pa, and needs to be replaced at a final pressure drop of 622 Pa. A

production file for this filter can be found in annex H. The pressure drop increases is determined with the pressure drop/time equation of the coarse 60% filter. The results can be seen in graph 12.



Graph 12: Pressure drop evolution for a H13 filter.

This pressure drop variation then results in a similar particle load/pressure drop correlation for a H13 filter. This can be seen in graph 13. The corresponding equation can also be found below.



Graph 13: Pressure drop evolution for a rising particle load of a H13 filter.

$$y = 249.51e^{14.697 x}$$

This correlation will allow the pressure drop to be estimated for other outdoor concentrations.

10.

Result and discussion

10.1. Base scenario

After running the simulation, different parameters can be calculated for each of the spaces in the apartment. These parameters can be compared to the air quality guidelines by WHO. However, since these guidelines only pose values for PM_{2.5} and PM₁₀, an interpolation was used to determine guideline values for the other bin sizes. Guideline values for each of these bin sizes were developed based on the highest diameter size in the corresponding bin size and an exponential curve. These values can be found below, in table 17. The 24-hour average is allowed to be exceeded for maximum 3 days a year.

Pollutant	Bin Size number	Averaging Time	2021 standards
			µg/m ³
PM _{0.001-01}	1	Annual	3
		24-hour	10
PM _{0.1-0.2}	2	Annual	3
		24-hour	10
PM _{0.2-0.25}	3	Annual	3
		24-hour	10
PM _{0.25-0.3}	4	Annual	3
		24-hour	10
PM _{0.3-0.4}	5	Annual	3
		24-hour	11
PM _{0.4-0.5}	6	Annual	3
		24-hour	11
PM _{0.5-1}	7	Annual	4
		24-hour	12
PM _{1-2.5}	8	Annual	5
		24-hour	15

PM2.5-5	9	Annual	7
		24-hour	21
PM5-10	10	Annual	15
		24-hour	45

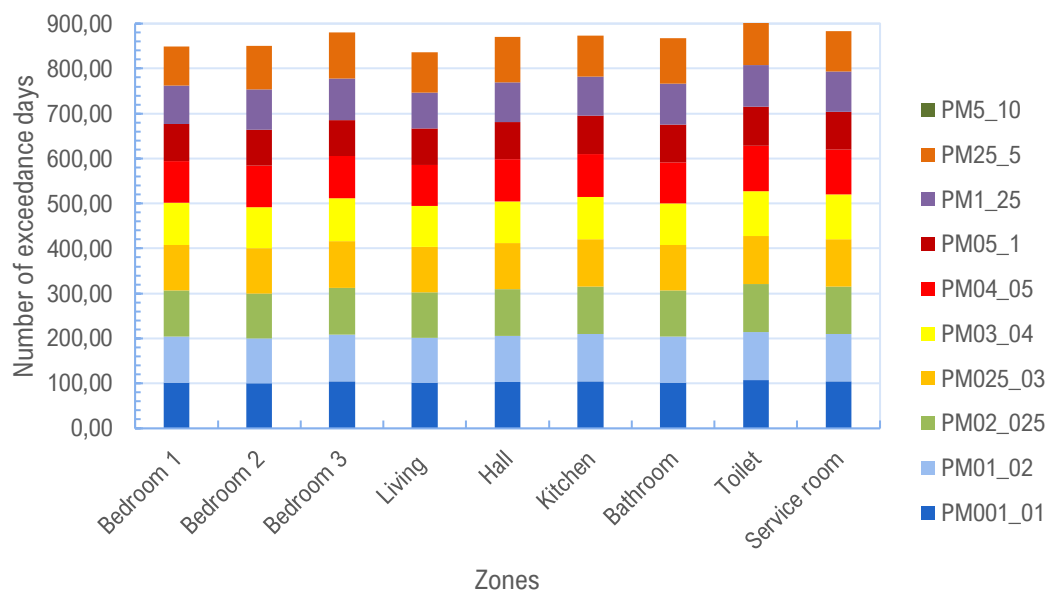
Table 17: Air quality guidelines World Health Organization in regards to particulate matter (2021)

For this base scenario, the average annual concentrations of PM that were measured, can be found in table 18. If the yearly mean values are compared to the guidelines opposed by the WHO, it is clear that for each zone, the limit is exceeded. For the smallest bin sizes, the average surrounds 7 $\mu\text{g}/\text{m}^3$, while about 3 $\mu\text{g}/\text{m}^3$ is recommended. For the bin size between 1 and 2.5 μm , the difference with the guidelines is about 4 $\mu\text{g}/\text{m}^3$ and for those above 2.5 μm the difference is even higher. Health effects corresponding to all size fractions of PM can therefore be expected.

Annual average Vilvoorde	0.01-0.1 μm	0.1-0.2 μm	0.2-0.25 μm	0.25-0.3 μm	0.3-0.4 μm	0.4-0.5 μm	0.5-1 μm	1-2.5 μm	2.5-5 μm	5-10 μm
	$\mu\text{g}/\text{m}^3$									
TARGET VALUE	3	3	3	3	3	3	4	5	7	15
Bedroom 1	7.25	7.25	7.25	7.25	7.25	7.25	7.25	9.06	17.49	17.64
Bedroom 2	7.27	7.27	7.27	7.27	7.27	7.27	7.27	9.01	17.62	17.72
Bedroom 3	7.24	7.24	7.24	7.24	7.24	7.24	7.24	9.06	17.63	17.59
Living	7.15	7.15	7.15	7.15	7.15	7.15	7.15	8.89	17.27	17.32
Hall	7.27	7.27	7.27	7.27	7.27	7.27	7.27	9.06	17.52	17.50
Kitchen	7.24	7.24	7.24	7.24	7.24	7.24	7.27	8.91	17.52	17.43
Bathroom	7.21	7.21	7.21	7.21	7.21	7.21	7.24	8.88	17.50	17.63
Toilet	7.51	7.51	7.51	7.51	7.51	7.51	7.51	9.31	18.17	17.93
Service room	7.31	7.31	7.31	7.31	7.31	7.31	7.31	9.08	17.72	17.66

Table 18: Average PM concentration in each room in Vilvoorde in 2023.

The highest numbers are found in the rooms with only exhaust, especially the toilet and service room. The air that enters these rooms already comes from other rooms. The living room has the lowest concentration of PM. This could be due to good ventilation and the size of the room. Subsequently, the number of days that the daily average exceeds the guidelines for a maximum daily concentration of the WHO can be investigated. Graph 14 represents a summary of the exceedance days for the different rooms.



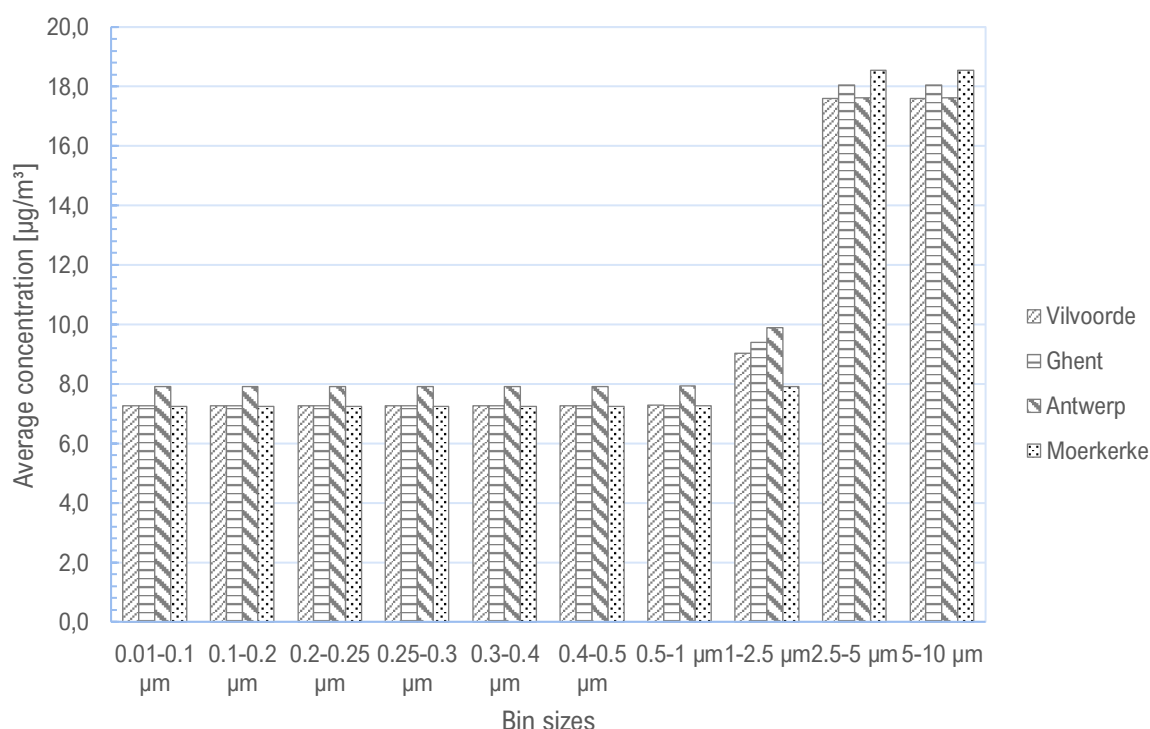
Graph 14: Number of days exceeding the 24-hour limit opposed by the WHO in 2023 in the base scenario.

For the largest bin size, no exceedance days are measured. These are the least harmful size fractions. Therefore, the threshold is also quite high. Since these larger fractions settle very quickly this is a logical outcome. For all other size fractions, the number of exceedance days is between 80-105, which is a lot more than the acceptable 3. Especially, the smallest bin sizes (up till $0.3\ \mu\text{m}$) have higher exceedance day numbers. These are also the most harmful fractions, so getting this number to decrease is vital. The highest numbers are observed in the smallest rooms with exhaust air: service room and toilet. The lowest numbers can be found in the living area and first two bedrooms (all supply). A table of these results can be found in annex B. These concentrations will serve as a baseline for the researched case studies.

10.2. Outdoor PM concentration

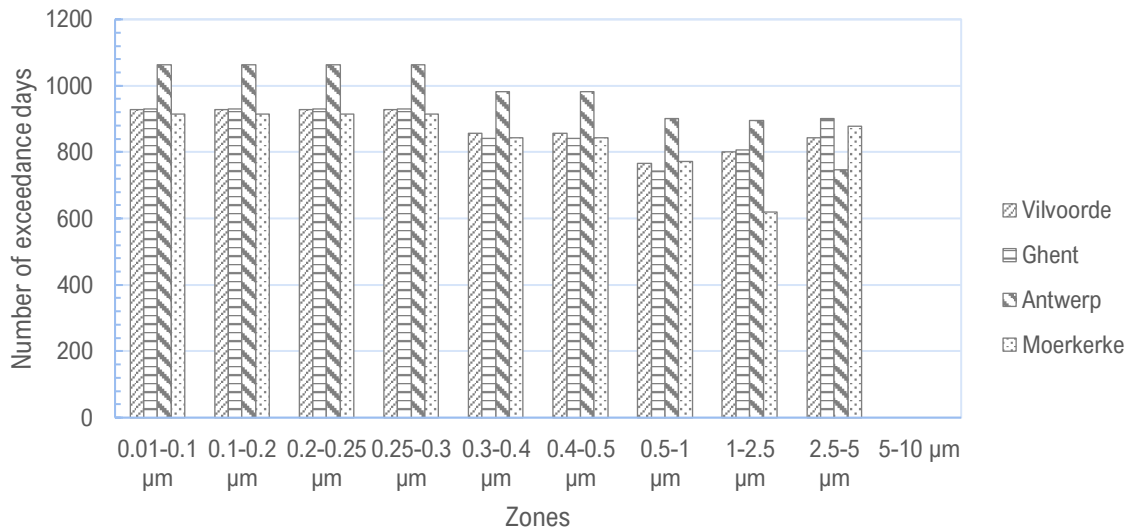
The average of the annual average in all rooms for the 4 different locations can be compared with graph 15. The average concentration for bin sizes below $1\ \mu\text{m}$ is the highest in Antwerp. Vilvoorde also displays a slightly higher concentration for these bin sizes as opposed to Ghent and Moerkerke. For PM1-2.5, the highest concentrations are again found in Antwerp, followed by Ghent and Vilvoorde. The lowest concentration for this fraction was found in Moerkerke. The difference could be attributed to traffic or industrial processes. For the two largest bin sizes, the highest concentration is observed in Moerkerke. Usually more natural PM fractions are more present in this rural environment. These particles are known to have a larger size diameter. This is contrary to what the average and peak concentrations indicated. It seems that the influence of the peaks was indeed the cause for those values. The second highest concentrations are found in Ghent and the lowest, next to the traffic sites in Antwerp and Vilvoorde.

Nevertheless, all values are higher than the recommended guidelines. Full tables with the annual average and number of exceedance days in each room can be found in annex C.



Graph 15: Difference in average indoor PM concentration per bin size.

Subsequently, the number of days where the 24-hour average is exceeded is analysed. For the bin sizes below 2.5 µm, the boundary concentration is exceeded more often in Antwerp than in the other cities. Usually, the number of Vilvoorde is also slightly higher than for Ghent and Moerkerke. This is probably explained by the more abundant presence of traffic in Antwerp and Vilvoorde. Moerkerke has the lowest number of exceedance days for all these bin sizes. For bin size 2.5-5 µm, the number of exceedance days is higher for Ghent and Moerkerke as opposed to Antwerp and Vilvoorde. No exceedance days are counted for the highest bin size in any of the environments.



Graph 16: Comparison of the number of exceedance days for different environments.

10.2.1. Conclusion

The lack of variation in the data, probably means that in the whole of Belgium, particulate matter concentration is quite similar, and the influence of a specific location is therefore minimal. Belgium is a very small country. Local sources are spread very easily throughout the country. However, small differences in size distribution can be observed. Traffic seems to attribute to PM1 and PM2.5, while more rural activities could be attributed to PM10. Since smallest particles are most dangerous to the human health, living in a rural area is better than living in the city centre and living in the city centre is better than living next to a large traffic site. Although the difference is minimal.

All these concentrations exceed the limit of the World Health Organization from table 17 easily. The 24-hour mean value in every room is also exceeded on around 80-110 days for each bin size, apart from the largest, in each case. Which is a lot more than the 3 exceedance days that are allowed. From a health perspective the indoor PM concentration is therefore too high, independent of in which of the four locations the building may be constructed. Subsequently, when this air enters a building, the indoor air is also insufficient. Taking measurements to prevent these concentrations from entering the building is therefore necessary. On a larger scale, source control would be necessary to lower the overall concentrations in Belgium.

10.3. Filters

From looking at the values in table 19, it is evident that placing a filter has a significant effect. Even only a coarse filter, improves the air quality remarkably. The annual average concentrations for PM smaller than 1 μm are now a little higher than 4 $\mu\text{g}/\text{m}^3$, instead of 7 $\mu\text{g}/\text{m}^3$. With an ePM1 55% filter, which has a higher efficiency for the smallest particles, this value goes down till around 3 $\mu\text{g}/\text{m}^3$. The guidelines require values below 3 or 4 $\mu\text{g}/\text{m}^3$. Even though this is not yet the case, the improvement is bound to have a positive effect on the general health. For particulate matter between 1 and 2.5 $\mu\text{g}/\text{m}^3$, the annual concentration with a coarse filter is around 4 $\mu\text{g}/\text{m}^3$ as well. The target value of 5 $\mu\text{g}/\text{m}^3$ is therefore met. However, when an ePM1 55% filter is used the concentration for this bin size increases with 1-2 $\mu\text{g}/\text{m}^3$ and is again higher than 5 $\mu\text{g}/\text{m}^3$. For the bin size 2.5-5 $\mu\text{g}/\text{m}^3$, the values are still a bit too high for both filters.

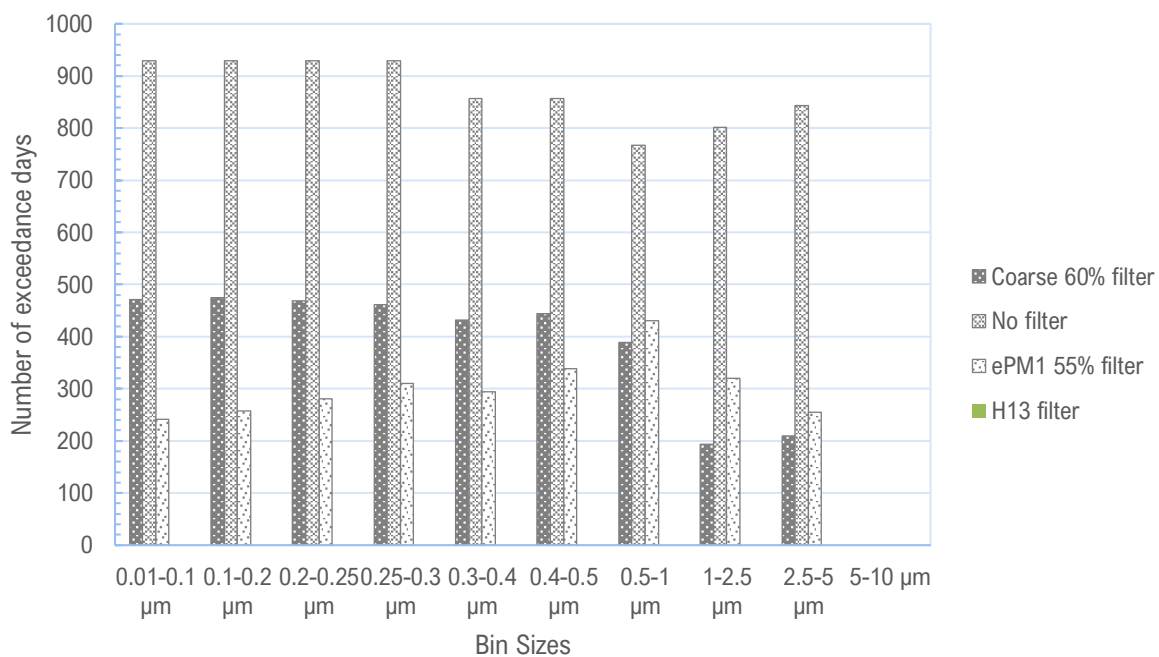
For PM larger than 5 μm , the concentration indoors decreases from about 17 $\mu\text{g}/\text{m}^3$ till about 7-9 $\mu\text{g}/\text{m}^3$. These values stay below target value for both filters. As opposed to the base scenario, the highest values are measured in the kitchen, before the toilet, service room and hall. These are again all rooms where ventilation exhaust is present.

A H13 filter is a very high efficiency filter, with efficiencies very close to 100%. Therefore, a large influence is to be expected. The average concentrations are near zero for all bin sizes. The highest concentrations can be found in the kitchen, even though there are still no indoor sources present. Furthermore, higher concentrations are also present in the living area. Due to the size of the room, air leakage probably more influence, explaining this higher concentration. The concentrations are in all cases lower than the guidance values. Full results can be found in annex D.

Annual average Different filters	0.01-0.1 μm	0.1-0.2 μm	0.2-0.25 μm	0.25-0.3 μm	0.3-0.4 μm	0.4-0.5 μm	0.5-1 μm	1-2.5 μm	2.5-5 μm	5-10 μm
	$\mu\text{g}/\text{m}^3$									
TARGET VALUE	3	3	3	3	3	3	4	5	7	15
Base Scenario	7.3	7.3	7.3	7.3	7.3	7.3	7.3	9.0	17.6	17.6
Coarse 60% filter	4.5	4.4	4.5	4.5	4.5	4.5	4.6	4.7	9.3	7.5
ePM1 55% filter	3.2	3.3	3.5	3.6	3.8	4.1	4.8	5.5	11.1	9.2
H13 filter	0.2	0.2	0.2	0.1	0.1	0.1	0.2	0.2	0.3	0.3

Table 19: Average PM concentration in each room with different filters.

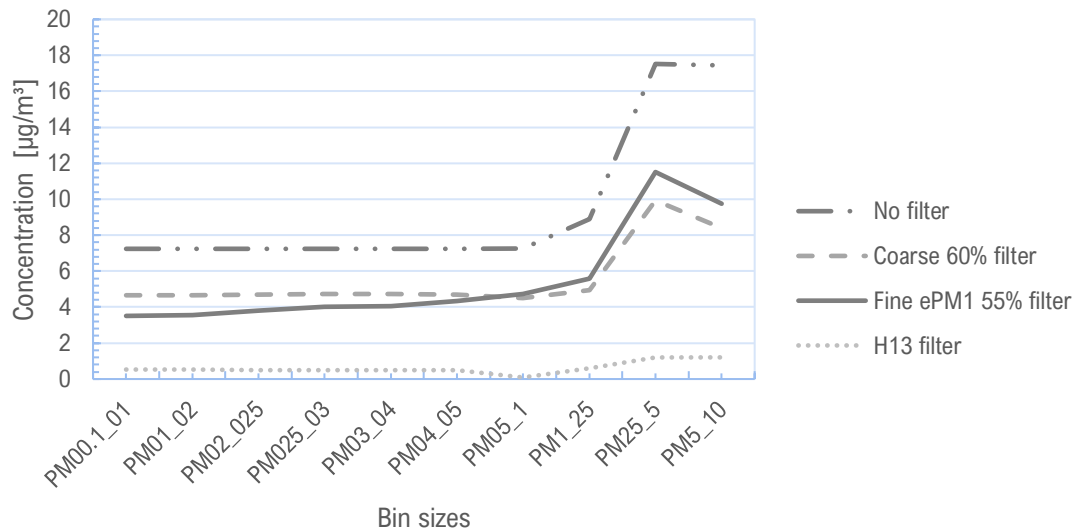
Furthermore, the exceedance days also decreased noticeably for all filters. In case of a coarse filter, the number of exceedance days are cut in half for the smallest PM particles. Exceedance days for PM1-2.5 and PM2.5-5 even go from about 90 till around 25 days. With an ePM1 55% filter, the decrease of exceedance days for the smallest bin sizes is even higher. Other bin sizes count more exceedance days as compared to the coarse filter. Especially, the bin size 1-2.5. Just like in the base scenario, the daily average concentration for PM larger than 5 μm is not exceeded in any case. With a H13 filter, the maximal daily average is not exceeded in any room. The visual comparison of the coarse, fine, H13 and no filter can be found in graph 17.



Graph 17: Number of days exceeding the 24-hour limit opposed by various filters.

10.3.1. Conclusion

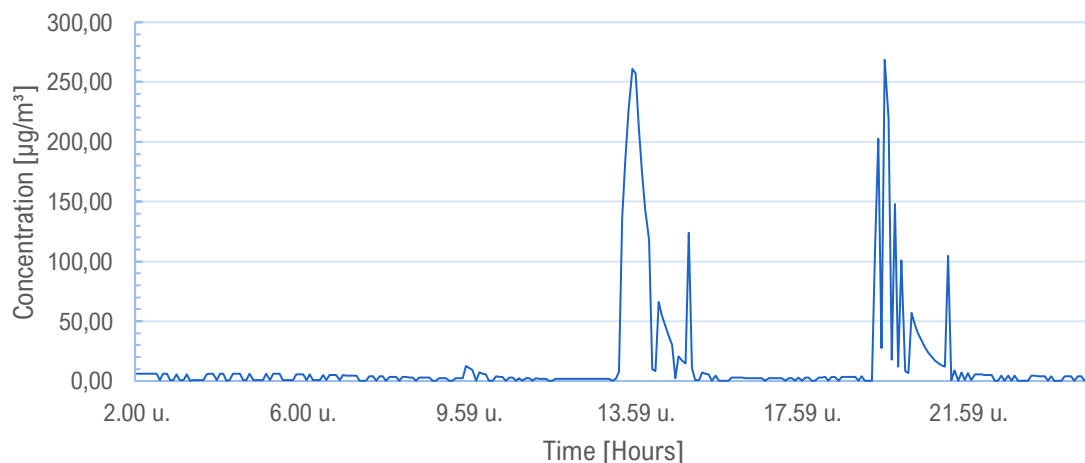
When looking at a visual comparison for the PM concentration in the kitchen, that can be found in graph 18, it is clear that placing a filter has a significant influence in all cases. But the concentration is only consistently lower than the target values when a HEPA filter is used. Additionally, only with a HEPA filter, the number of exceedance days is sufficiently low. Therefore, only the HEPA filter is sufficient for the general health. Fine and coarse filters only reach the target values for a few bin sizes. The exceedance days for most size fractions are still to numerously present to speak of a healthy environment as well. Considering the higher toxicity of lower size diameters, the ePM1 55% filter will allow a better air quality and less health effects than the coarse filter.



Graph 18: Comparison of the average annual concentrations for each bin size in the kitchen.

10.4. Indoor PM source : Cooking

Since almost all cooking-related PM belonged to the first bin size (PM0.01-01), the highest effect is expected there. Furthermore, this emission is also place dependent. In graph 19, the influence of cooking on the PM concentration in the kitchen on a daily basis (no filter), can be observed more closely. For the displayed day (Sunday), all three meals were cooked. During lunch and dinner time, large peaks can be observed up till 250 $\mu\text{g}/\text{m}^3$. During breakfast there is also a small peak of about 9 $\mu\text{g}/\text{m}^3$. The peaks created by cooking are very high and very dangerous.



Graph 19: PM concentration in the kitchen PM0.01-0.1 on a Sunday.

For the annual average, the highest effect is observed in the kitchen and service room. The comparison of the concentration for these two rooms for different cases, can be found in table 20 and 21. The full results for each room can be found in Annex E.

When no filter is present, the smallest PM size fraction in the kitchen increases in annual average concentration till $13.25 \mu\text{g}/\text{m}^3$, which is almost twice the concentration of the base scenario. For the surrounding rooms, there is an influence especially on the service room. Here the concentration increases from $7.31 \mu\text{g}/\text{m}^3$ till $9.07 \mu\text{g}/\text{m}^3$.

Evidently, the PM emitted by the cooking does not pass the filter. Therefore, a significant increase can be expected even when a filter is present. However, since the kitchen is responsible for exhaust, a filter can remove the cooking-related PM more quickly. The difference between no cooking and cooking is about the same for the scenario without filter as with a coarse or fine filter. When a coarse filter is used, the average concentration for the smallest bin size when cooking related PM is emitted is about $10.6 \mu\text{g}/\text{m}^3$. In the service room, there is also a similar decrease of $2 \mu\text{g}/\text{m}^3$ as opposed to the simulation without filter.

The presence of a H13 filter is also not able to compensate for the cooking related increase of PM. The annual average still exceeds the target value of the smallest bin size for the kitchen. The increase in indoor PM concentration is about $6 \mu\text{g}/\text{m}^3$ when cooking is added, this is higher than for the other filters. For the service room, the limit value is achieved so the circulation of PM seems to be reduced.

Annual average Cooking Kitchen	0.01-0.1 μm	0.1-0.2 μm	0.2-0.25 μm	0.25-0.3 μm	0.3-0.4 μm	0.4-0.5 μm	0.5-1 μm	1-2.5 μm	2.5-5 μm	5-10 μm
	$\mu\text{g}/\text{m}^3$									
LIMIT VALUE	3	3	3	3	3	3	4	5	7	15
Base Scenario	<u>7.24</u>	7.24	7.24	7.24	7.24	7.24	7.27	8.91	17.52	17.43
No filter	<u>13.25</u>	7.19	7.19	7.19	7.19	7.19	7.28	8.95	17.37	17.49
Coarse 60% Filter	<u>10.56</u>	4.49	4.57	4.6	4.57	4.64	4.39	4.86	9.78	8.32
H13 filter	<u>6.55</u>	0.52	0.52	0.52	0.52	0.52	0.09	0.6	1.19	1.21

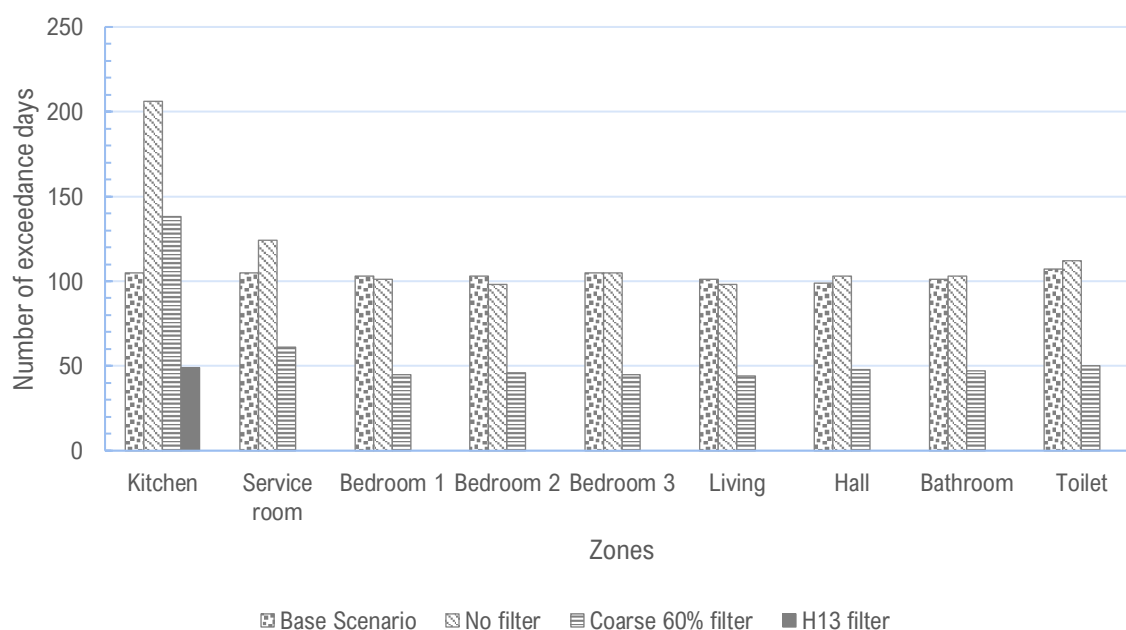
Table 20: Average annual PM concentration in the kitchen with cooking activities present.

Annual average Cooking Service Room	0.01-0.1 μm	0.1-0.2 μm	0.2-0.25 μm	0.25-0.3 μm	0.3-0.4 μm	0.4-0.5 μm	0.5-1 μm	1-2.5 μm	2.5-5 μm	5-10 μm
	$\mu\text{g}/\text{m}^3$									
LIMIT VALUE	3	3	3	3	3	3	4	5	7	15
Base Scenario	<u>7.31</u>	7.31	7.31	7.31	7.31	7.31	7.31	9.08	17.72	17.66
No filter	<u>9.07</u>	7.32	7.32	7.32	7.32	7.32	7.32	9.13	17.75	17.7
Coarse 60% Filter	<u>6.15</u>	4.43	4.46	4.46	4.48	4.45	4.58	4.75	9.5	7.82
H13 filter	<u>2.24</u>	0.26	0.26	0.26	0.26	0.26	0.26	0.3	0.53	0.52

Table 21: Average annual PM concentration in the service room with cooking activities present.

A comparison in exceedance days between the base scenario and the model with occupant and cooking can be found in graph 20. Due to cooking, the kitchen the number of exceedance days almost double in the kitchen when no filter is present. In the service room about 20 extra exceedance days can be observed. Some of the rooms that are adjacent to the kitchen also experience a slight incline in exceedance days.

The number of exceedance days decreases till about 140 days when a coarse filter is added. This is about 60 days less than with no filter present. The decrease in exceedance days is bigger for the service room. Here the concentration is exceeded 61 days as opposed to 124 days when no coarse filter was present. With a H13 filter, the number of exceedance days in the kitchen is about 50 days, as opposed to 138 when a coarse filter was present. So, there is a large difference, but compared to the scenario without cooking that had 0 exceedance days, the air quality is not sufficient.



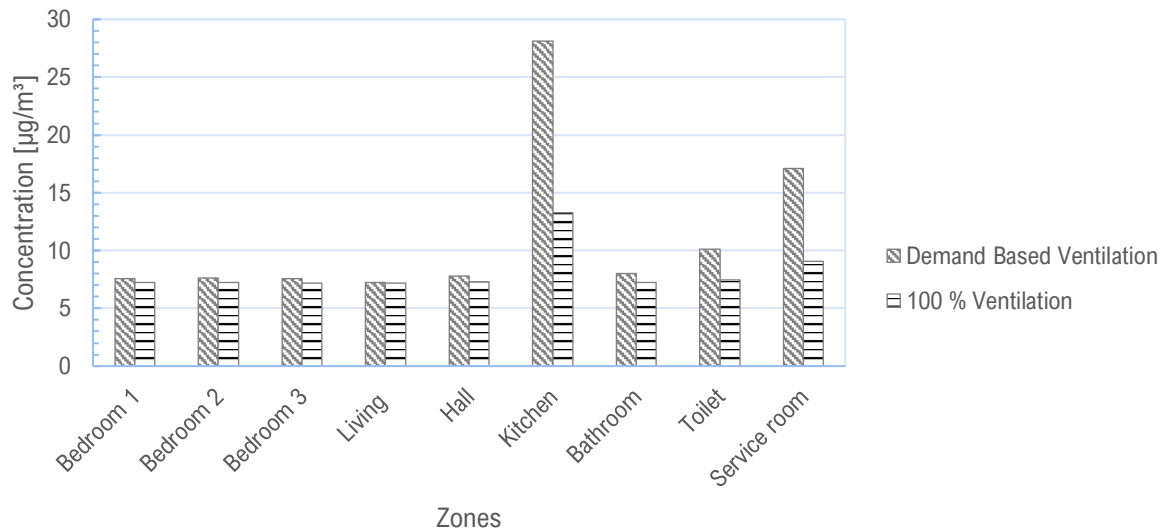
Graph 20: Number of exceedance days for the smallest bin size for each room.

10.4.1. Conclusion

Overall, the influence of cooking on the indoor environment is very significant. The concentrations for the smallest and most toxic particle size increase notably. This can lead to numerous health effects, like brain and nerve damage. All other organs can be affected as well. Compared to the situation without filter, the use of a coarse or high efficiency filter results in a significant improvement that will reduce the risks significantly. However, the decrease in the indoor PM concentration due to a coarse or H13 filter is not significantly enough to compensate for the cooking-related increase. And the higher the efficiency of the filter, the higher the influence of cooking on the overall concentration. Creating a completely healthy environment is therefore not possible in the kitchen. In order to obtain such environment a cooking hood with higher efficiency for those particles could be used. The installation of an air cleaner in the kitchen could also be an improvement. Nevertheless, the H13 filter is able to prevent a higher concentration in the neighboring rooms.

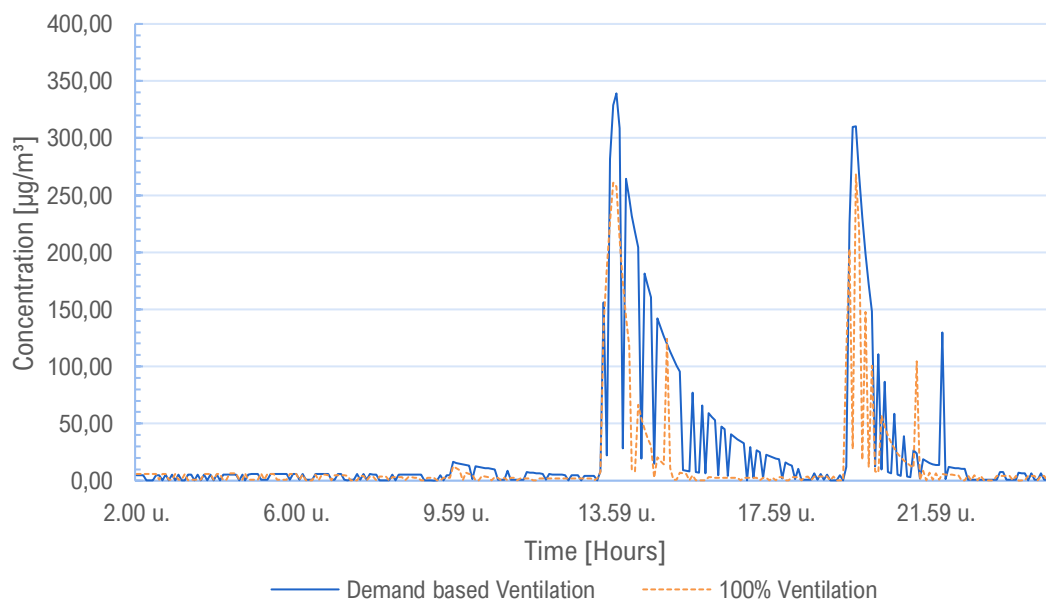
10.5. Occupant based ventilation

Demand based ventilation, results in average concentrations as can be seen in graph 21 for bin PM0.01-0.1, as the difference was most pronounced for this bin size. Other results can be found in annex F. Overall, the average concentration in the rooms increase. For the dry rooms, this increase is only minimal. But in the wet rooms, the influence of demand-based ventilation is significantly higher. In the kitchen the smallest bin size increases till about $28\mu\text{g}/\text{m}^3$. This is more than twice the concentration with 100% ventilation. The demand-based ventilation is not able to sufficiently capture the PM emitted by cooking. The same phenomenon can be observed for the service room. Furthermore, in the toilet and bathroom, the overall concentrations also increase. This can probably be explained by the decrease in humidity. The demand-based ventilation results in an overall higher indoor PM concentration, especially in the rooms that are controlled based.



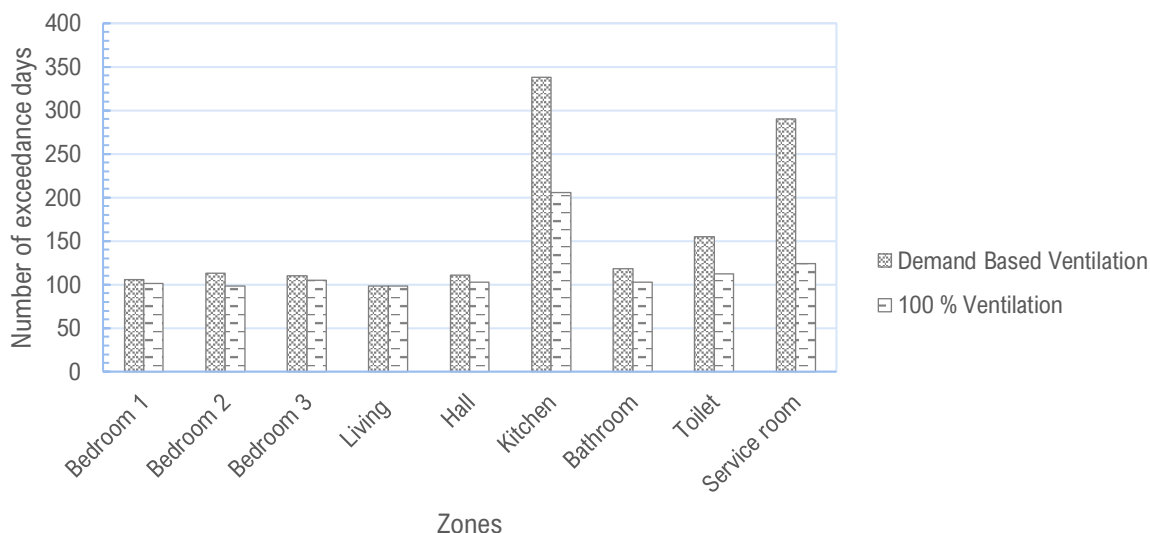
Graph 21 : Annual average concentration in different rooms for bin size PM_{0.01-0.1} with and without demand based ventilation.

This influence can be investigated more closely. The PM_{0.01-0.1} concentration for the kitchen can be seen in graph 22. The cooking-related peak has increased from about 250 $\mu\text{g}/\text{m}^3$ till about 320-350 $\mu\text{g}/\text{m}^3$ with demand-based ventilation. By the time the ventilation system starts to work, the concentration is already higher than in the case of a continuous system. The concentration also decreases faster if the system is constant since the system is always on. Therefore, the demand-based ventilation, results in a kitchen environment with higher concentrations when the occupants are present.



Graph 22: Average concentration of PM in the kitchen with 100% ventilation and demand based.

The number of exceedance days can be compared via graph 23. Here, a similar pattern is revealed. The daily concentration in the kitchen is exceeded almost every day. The overall number of exceedance days also increases significantly for the other wet rooms (bathroom and toilet). More detailed numbers can be found in annex F.



Graph 23: Average concentration of PM in the kitchen with 100% ventilation and demand based.

10.5.1. Conclusion

Overall, a demand-based system has a negative influence on the indoor PM concentration. Constant ventilation is better in case of particulate matter source removal.

10.6. Window opening

The influence of the opening of windows in scenario's with and without a filter can be observed in table 22. Full results can be found in annex G.

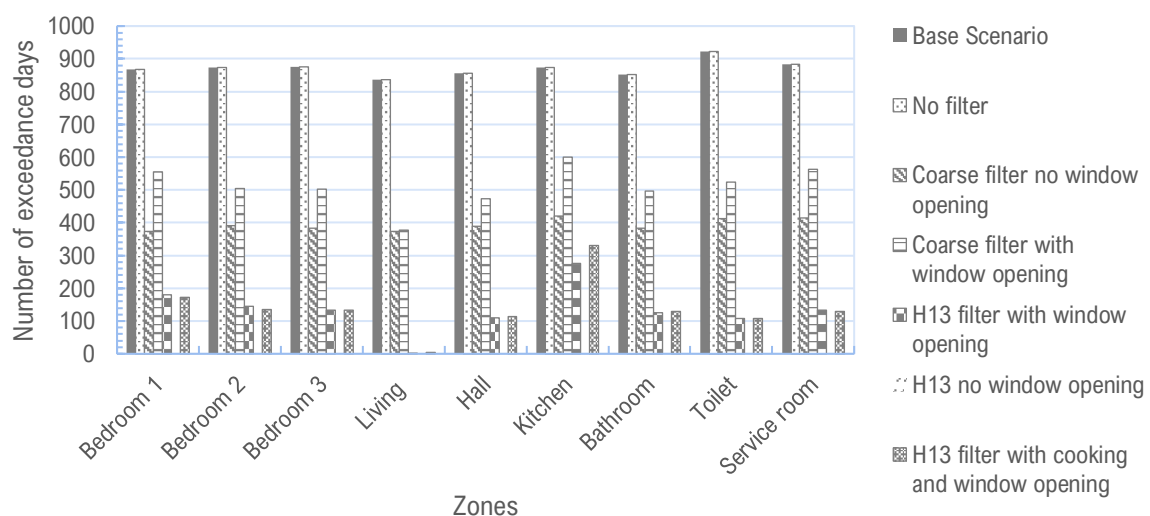
Annual average Window opening	0.01-0.1 μm	0.1-0.2 μm	0.2-0.25 μm	0.25-0.3 μm	0.3-0.4 μm	0.4-0.5 μm	0.5-1 μm	1-2.5 μm	2.5-5 μm	5-10 μm
	$\mu\text{g}/\text{m}^3$									
LIMIT VALUE	3	3	3	3	3	3	4	5	7	15
Base Scenario	7.3	7.3	7.3	7.3	7.3	7.3	7.3	9.0	17.6	17.6
No filter	7.2	7.2	7.2	7.2	7.2	7.2	7.2	8.9	17.5	17.5
Coarse Filter	5.3	5.4	5.4	5.4	5.4	5.4	5.5	6.2	12.5	11.6
H13 filter	2.3	2.3	2.3	2.3	2.3	2.3	2.3	2.9	7.2	7.2

Table 22: Average annual PM concentration for all bin sizes with window opening.

When no filter is present, window opening does not have a large influence. It even seems to decrease the concentration. In the absence of a filter, the ventilation system constantly brings in air mixed with pollutants. Opening the windows, may allow some more PM to enter, but some can also leave. It appears that in this case, more PM is leaving than there is entering. This effect is largest for the bin sizes below 1µm, which are the sizes that stay airborne the longest. Due to their higher settling rate, the larger size diameters probably escape less easily.

If a coarse filter is present in the system, a clear influence is visible. The average concentration goes up significantly in all cases, especially for the larger bin sizes. These concentrations are more abundantly present in the outdoor environment. Therefore, the influence of an unfiltered fraction is larger for these concentrations. However, the target value is still reached for the largest bin size. The other concentrations are too high to comply with these targets. If non-filtered outdoor air enters, the influence is significant. And the work of the ventilation system is partially undone.

The opening of windows when a H13 filter is present has an even higher influence. The yearly average concentration goes from between 0 and 1 µg/m³ to about 2 µg/m³ for the smallest bin sizes and about 7 for the two largest bin sizes. In most cases the target values are still within reach. Only for the kitchen, this value is exceeded a little bit. Additionally, the number of exceedance days can also be compared in graph 24. Here, it is clearly visible that the number of exceedance days decreases when a filter is placed but increases again when windows are opened. For a H13 filter without window opening there were no exceedance days. Unfortunately, there are quite a few exceedance days when window opening is present. Overall, about 10-20 extra days of exceedance are added in all bin sizes, except the largest.



Graph 24: Number of exceedance days in each room in different scenarios.

For the living room, the influence is smaller. It seems like, due to the size of the room, the outdoor concentration has a smaller influence on the indoor concentration. For the H13 filter with window opening, the living room only has a few exceedance days. In the kitchen, the number of exceedance days is higher, even though no cooking is present at this stage. This could be explained by the presence of the turning window. Therefore, more air is exchanged than with a tilted window.

Now, the influence of cooking can also be investigated. A comparison between different scenarios for the kitchen can be found in table 20. The other rooms can be reviewed in annex G. The addition of the occupants causes a 4 $\mu\text{g}/\text{m}^3$ increase in the average annual concentration for the first bin size in the kitchen. This concentration (6.99 $\mu\text{g}/\text{m}^3$) is only slightly higher than when cooking was present without window opening. Window opening, therefore, has a small negative influence on indoor PM concentration when cooking related PM is present.

Annual average Window opening Kitchen	0.01-0.1 μm	0.1-0.2 μm	0.2-0.25 μm	0.25-0.3 μm	0.3-0.4 μm	0.4-0.5 μm	0.5-1 μm	1-2.5 μm	2.5-5 μm	5-10 μm
	$\mu\text{g}/\text{m}^3$									
LIMIT VALUE	3	3	3	3	3	3	4	5	7	15
Base Scenario	<u>7.24</u>	7.24	7.24	7.24	7.24	7.24	7.27	8.91	17.52	17.43
H13 filter	<u>3.19</u>	3.15	3.17	3.11	3.11	3.11	2.11	4.16	9.28	9.35
H13 filter + cooking	<u>6.55</u>	0.52	0.52	0.52	0.52	0.52	0.09	0.6	1.19	1.21
H13 filter + cooking + window	<u>6.99</u>	3.11	3.17	3.17	3.17	3.17	2.11	4.07	9.30	9.33

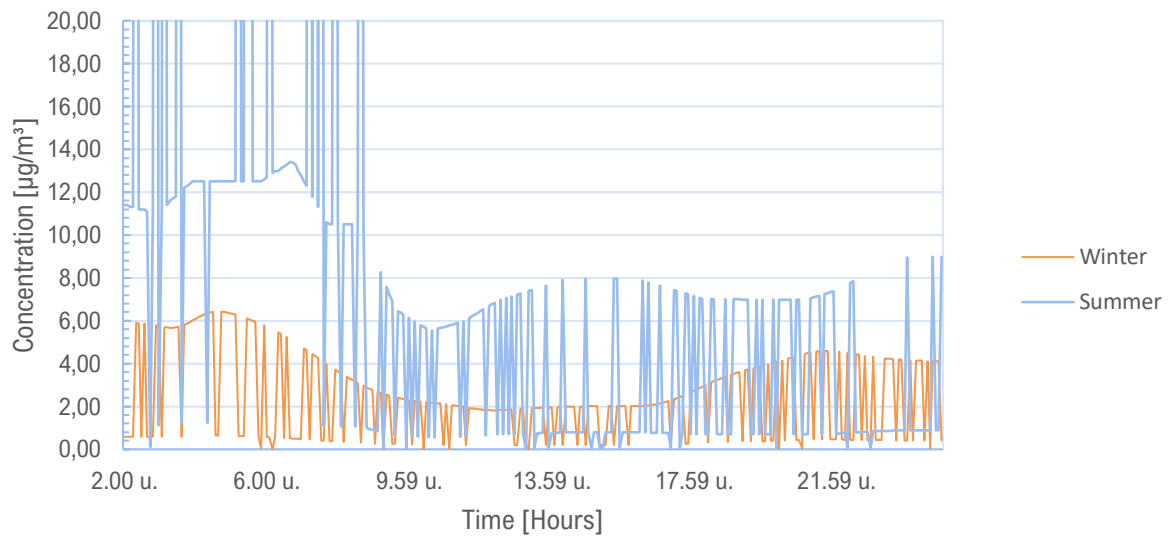
Table 20: Average annual PM concentration in the kitchen with window opening.

Additionally, the number of exceedance days more than doubles with the addition of an occupant. The window opening is not able to prevent that. As can be seen in graph 27.

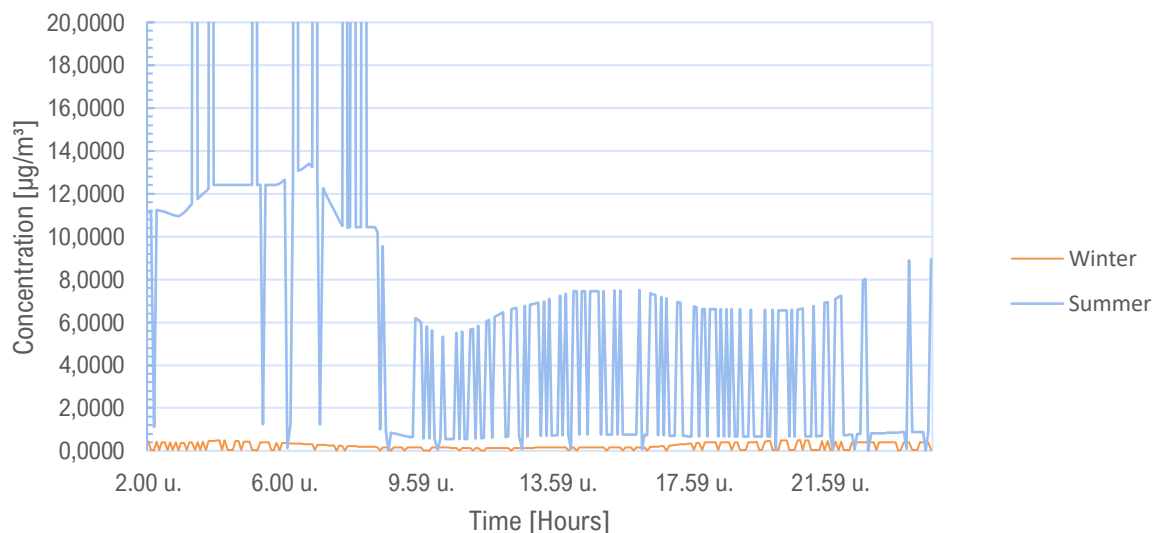
Since window opening is seasonal, the difference between summer and winter can be observed more closely in order to get a better understanding of the influence.

In graph 25, the difference in daily concentration between summer and winter can be seen. The concentration in the summer is significantly higher. In graph 26, a H13 filter is added. No indoor cooking source is present. The difference in the concentration can mainly be contributed to the summer. In the kitchen, the window is continuously open in this case. Therefore, the indoor concentration follows the

outdoor concentration. In the winter, when the windows are closed, incoming air is filtered by the H13 filter. This effect is completely gone in the summer.

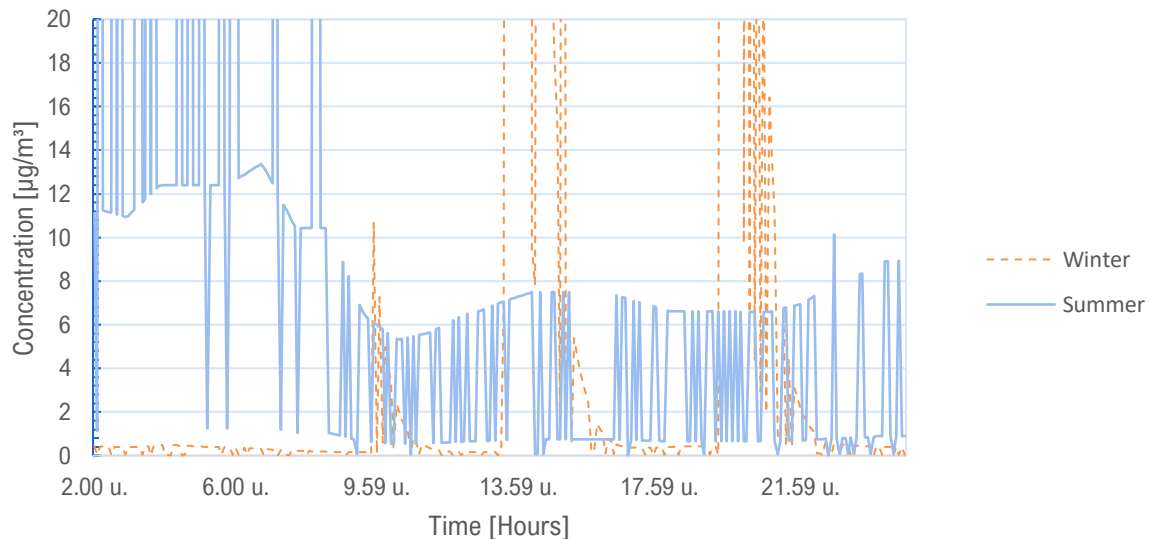


Graph 25: Daily concentration of PM_{0.01-0.1} in the kitchen on a summer and winter day when no filter is present.



Graph 26: Daily concentration of PM_{0.01-0.1} in the kitchen on a summer and winter day when a H13 filter is present.

When cooking is present and the windows are open, the cooking related peaks are basically gone. This can be observed in graph 27. The annual average concentration only increases due to the cooking related PM of the winter. Therefore, opening a window does have a positive influence on an indoor PM source. The negative influence of the window opening in other moments results in the overall higher concentration of the annual average.



Graph 27: Daily concentration of PM_{0.01-0.1} in the kitchen on a summer and winter day when a H13 filter is present and windows are opened in summer.

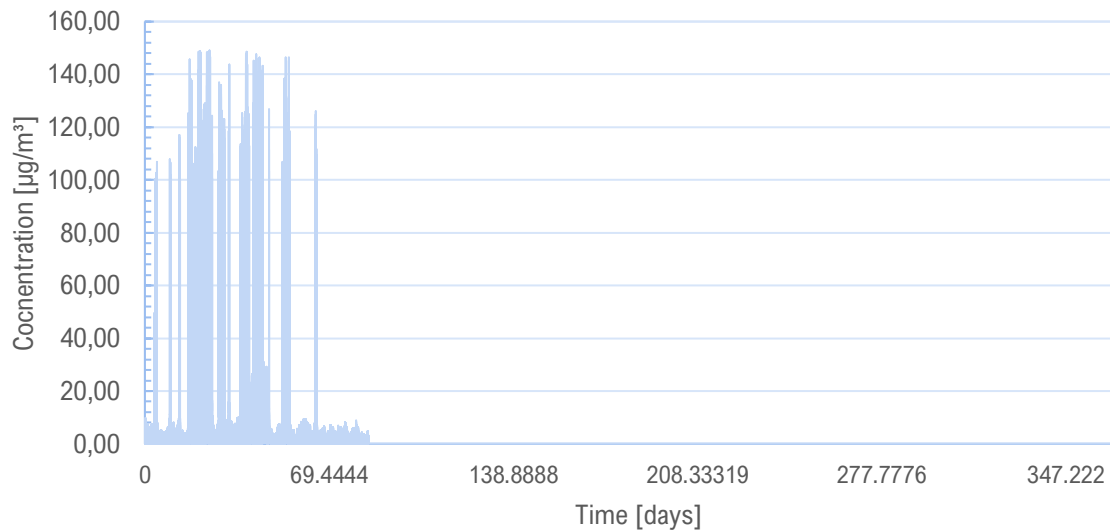
10.6.1. Conclusion

The influence of window opening on the indoor PM concentration is strongly correlated with the ventilation system. When no filter is present, the influence of window opening is minimal, even slightly positive. When a filter is added, the opening of windows lets in unfiltered air and increases the overall concentration. The higher the efficiency of the filter, the higher the negative influence of the opening of windows. The addition of an indoor source increases the indoor concentration overall. But when windows are opened, the peaks are avoided and the spreading to adjacent rooms is prevented. Thus, opening a window when a PM source is present in a certain room, can be a good form of source control. However, the window should not stay open too long, as it then becomes a source of pollutants as well.

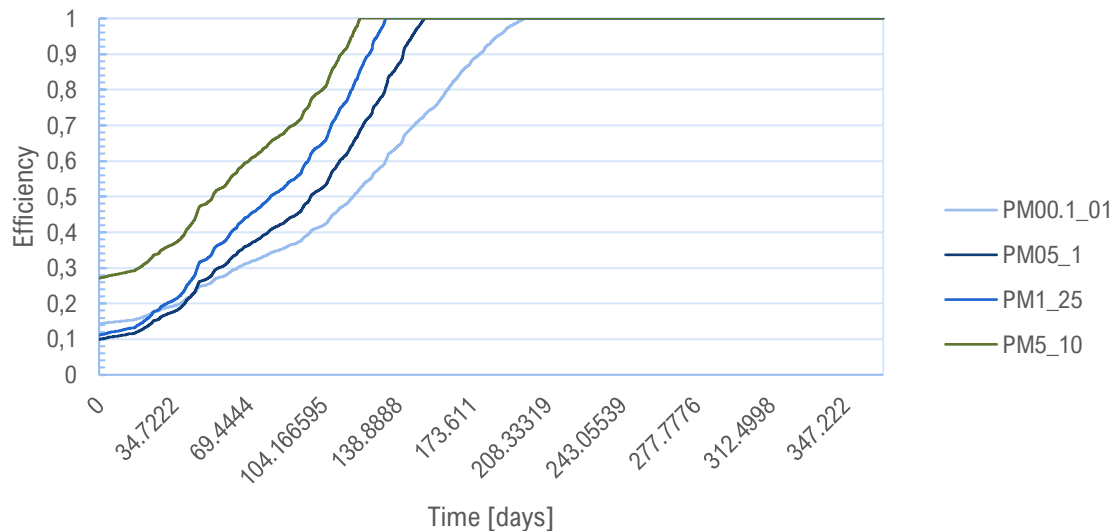
10.7. Lifetime of a filter

10.7.1. Coarse filter

When running the simulation with the coarse filter, something immediately stands out. Since this filter was tested with the real-life pollutant concentration of Brussels (close to Vilvoorde), a similar result could be expected. However, the filter in the simulation environment clogs very fast. The filter efficiencies evolve to 100% percent very quickly. This is illustrated with the particulate matter bin size 2.5-5µm in graph 28. After about 70 days, the filter efficiencies are all 1 and no PM is reaching the rooms via the ventilation system. Some filter efficiencies curve can be found in graph 29. This is of course beneficial for the indoor environment, but it is probably a bit too optimistic.



Graph 28: PM concentration in bedroom 3 between 2.5 and 5 µm when a coarse filter with varying efficiency is present.



Graph 29: Filter efficiency variation for 4 bin sizes.

This can be explained by the following. The filter was developed based on a mean concentration in the air. The variable air from Vilvoorde that was used was determined as a worst-case scenario. Where for each size fraction, the maximum possible amount of PM was present. Since the size division of PM in the air is not known and very place dependent, this was necessary to know the health risk. However, in this scenario, this assumption leads to the double counting of certain concentrations. The comparison between the mean concentration used for the calibration of the clogging filter component and the mean outdoor PM concentration of Vilvoorde for each bin size can be found in table 21. These higher input concentrations result in a faster clogging pattern. Especially, since each particle has the same contribution in the clogging process.

	Average concentration Out2In	Average concentration Vilvoorde
	$\mu\text{g}/\text{m}^3$	$\mu\text{g}/\text{m}^3$
Bin size PM0.01-0.1	0.71	8.08
Bin size PM0.1-0.2	7.81	8.08
Bin size PM0.2-0.25	7.39	8.08
Bin size PM0.25-0.3	9.01	8.08
Bin size PM0.3-0.4	4.71	8.08
Bin size PM0.4-0.5	2.00	8.08
Bin size PM0.5-1	0.91	8.08
Bin size PM1-2.5	3.93	9.6
Bin size PM2.5-5	8.46	18.5
Bin size PM5-10	7.62	18.5

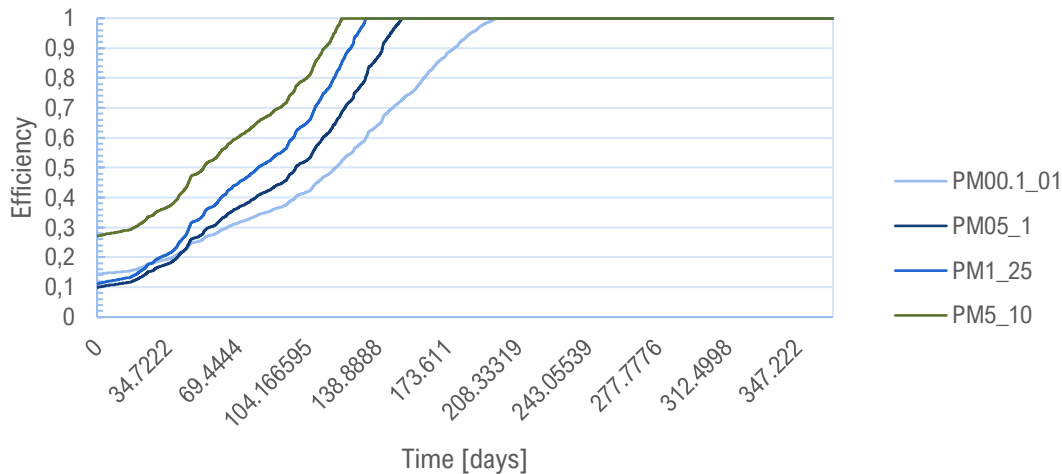
Table 21: Comparison between the average outdoor concentrations of PM for the Out2In project and Vilvoorde.

In order to get a more realistic profile, the outdoor concentration needs to be adapted. For this, the same division as in the Out2In project is adapted. In this case, each particle bin size will be multiplied by a factor to represent the share of particulate matter according to this division. The new (average) concentrations can be found in table 22. Of course, for the simulations, the varying data during the year will be used. This new division will most likely not be completely accurate. Since the location in Vilvoorde is closer to the ring road and the measuring station of Buildwise is in the centre of the city, smaller PM size fractions will most likely be more abundantly present in Vilvoorde than in the data from Buildwise.

	Average concentration Out2In	Average concentration Vilvoorde
	$\mu\text{g}/\text{m}^3$	$\mu\text{g}/\text{m}^3$
Bin size PM0.01-0.1	0.71	0.18
Bin size PM0.1-0.2	7.81	1.94
Bin size PM0.2-0.25	7.39	1.83
Bin size PM0.25-0.3	9.01	2.24
Bin size PM0.3-0.4	4.71	1.17
Bin size PM0.4-0.5	2.00	0.50
Bin size PM0.5-1	0.91	0.23
Bin size PM1-2.5	3.93	9.60
Bin size PM2.5-5	8.46	9.73
Bin size PM5-10	7.62	8.77

Table 22: Comparison between the average outdoor concentrations with adjusted size distribution.

The simulation can be repeated with this new outdoor concentration. When looking at the new efficiency curves, there is some influence. Unfortunately, it does not result in the kind of result that was expected. It takes a bit longer for all bin size efficiencies to reach 100%, but it still happens pretty fast, as can be seen in graph 30.



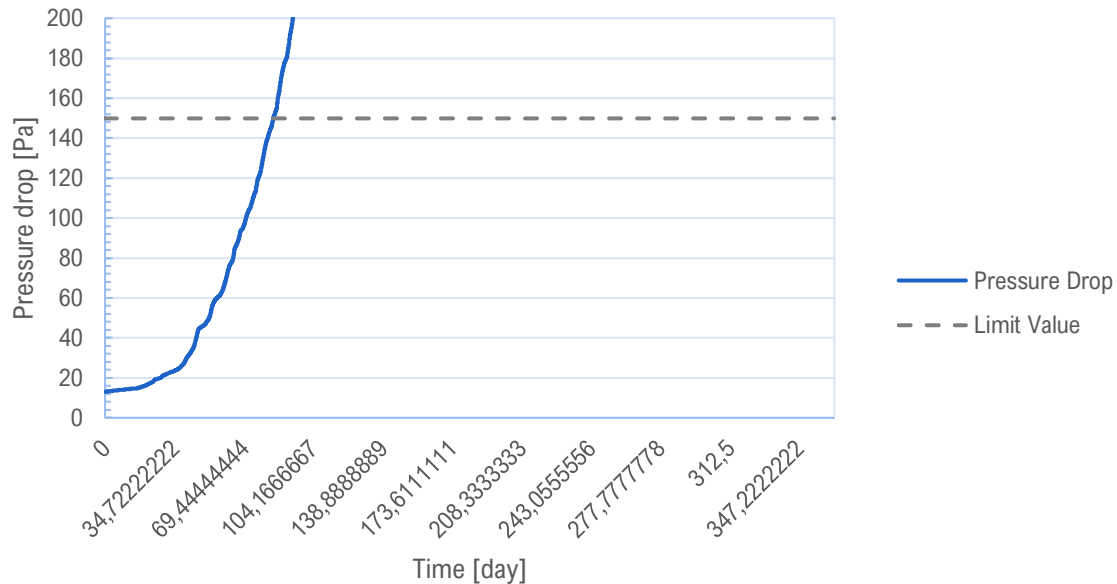
Graph 30: Filter efficiency variation for 4 bin sizes after outdoor concentration is adapted.

Obviously, this results in very low average concentrations, since after a certain date only PM that enters through the cracks of the building is measured. There are a few ways to explain this result. Firstly, the overall concentration that the filter is subjected to in this case is a lot higher than in the testing case. The variable concentration contains peaks that can be pretty high. Since the filter was tested with a mean concentration, no peaks were present. These higher concentrations could very well cause this faster clogging pattern. In the Out2In project, such peaks were most likely also present. However, this exact measured data from the research could not be retrieved, only the mean concentrations. If the filter was modelled with a varying PM concentration, the results would probably be different. This means that the filter should be retested if the real-life concentration distribution would ever become available.

Additionally, for the conversion of these values in number/l to $\mu\text{g}/\text{m}^3$ some assumptions were made that could result in a difference.

Another possibility is that the neglect of compaction and the simplification that all particles take an equal part in the clogging process, has led to this difference. A higher concentration in one of the bin sizes results in a higher concentration for all bin sizes. This is the case since the mass in the filter is distributed evenly over all bin sizes in order to determine the varying efficiencies. Thus, more research should be performed on this matter to determine the exact validity of this clogging model. Nevertheless, it possible that the filter clogs this fast. Therefore, the results will be discussed as well.

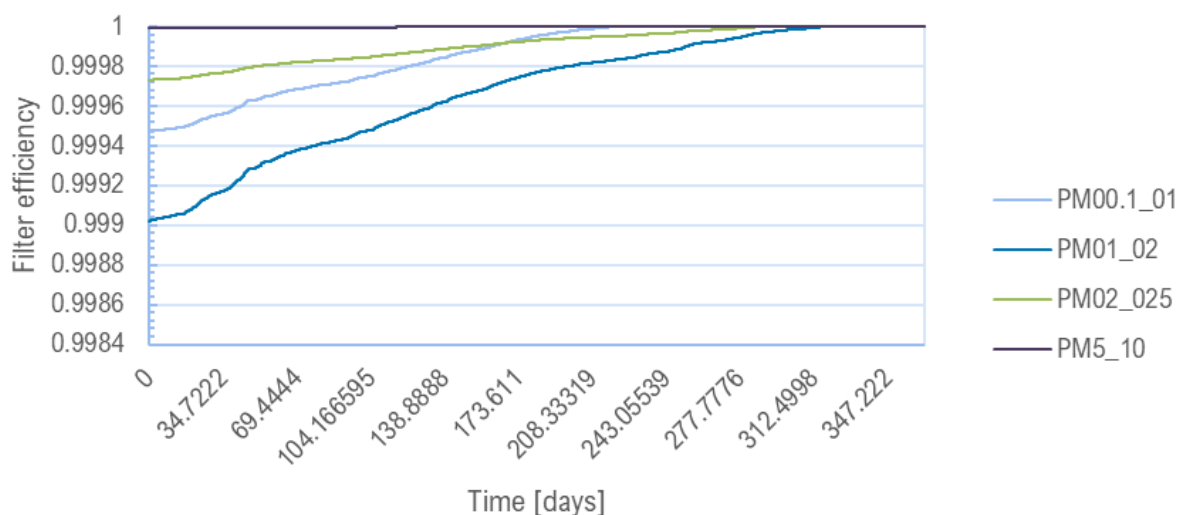
For the variable Coarse filter, the maximum pressure drop is 150 Pa, according to the Out2In project. This value is reached after about 84 days in the simulation, as can be seen in graph 31. This is very fast, as was explained above. It would mean that the determined pressure drop is not at all sufficient in a real-life environment and that filters need to be replaced faster than guidelines anticipate.



Graph 31: Pressure drop variable Coarse filter.

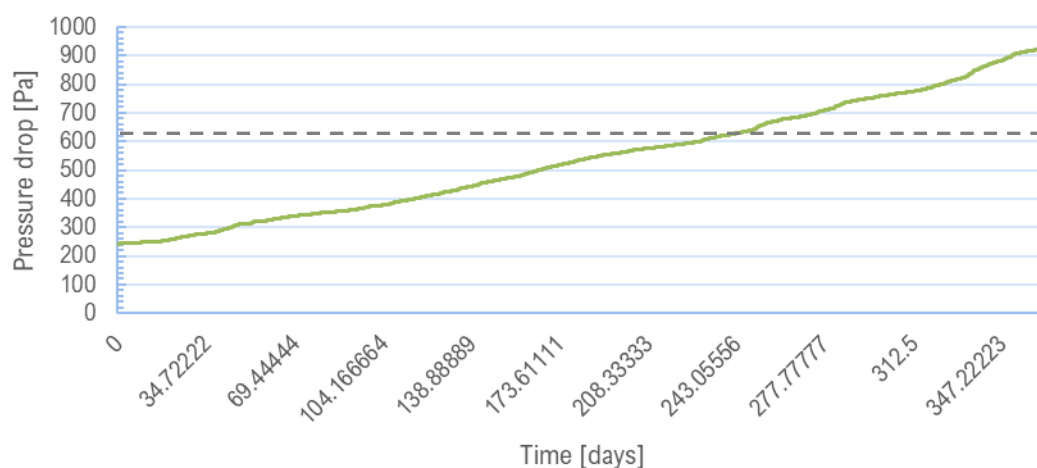
10.7.2. H13 filter

The H13 filter was tested with the adapted size division right away. Since no efficiency variation curves or pressure drop variations in time were available, this simulation already had a higher grade of uncertainty. However, when looking at the result, it does not seem unrealistic. Due to the very high efficiency of a HEPA filter, the difference in mass that remains in the filter is minimal. Even though the concentrations are different. The influence of the constant/variable outdoor concentrations as discussed for the coarse filter is a lot less important here. Close to no PM will be let through in either case. Moreover, the variation in efficiency curves is also more limited. Furthermore, due to these very high efficiencies, a fast clogging can be expected. This might explain the more fitting values in this simulation. Even so, the efficiency curves still go up to 100% before the year is over. A few efficiencies can be seen in graph 32.



Graph 32: Filter efficiency variation for H13 filter after outdoor concentration is adapted.

For the H13 filter, the pressure drop increases more gradually. A value of 622 Pa is set as the boundary value for when the filter needs to be replaced. The pressure drop evolution can be seen in graph 33. The final pressure drop is exceeded at 238 days. This lies between the 6-12 months range. But it also shows the deficiency of this guideline. Replacing the filter only after a year, will be good for the overall concentration, but bad for your energy use. And replacing the filter after six months will be good for the energy use but bad for the overall concentration.



Graph 33: Pressure drop variable H13 filter.

10.7.3. Resulting particle concentration

For a coarse filter, the effect of the high efficiencies is clearly visible in the average concentrations during its lifetime. Mean values for each bin size can be found in table 23. The values for each room specifically, are present in annex H. Especially for the lowest bin sizes, the efficiency seems to increase very fast. Since they have the same contribution as the highest bin sizes in the clogging phenomenon, and the

concentrations of the later are higher, this could be expected. Due to the initial efficiency drop, efficiencies are lower at the beginning. This initial higher concentration could not be compensated later on for the bin sizes above 1 μm . Here, the concentrations are higher than the target values.

The clogging of the H13 filter results in a significantly lower particulate matter concentration. Each concentration is lower than its target value. This can be seen in table 23.

Of course, if the filters stay in placed past their pressure drop maximum, these values will only decrease more.

Average concentration variable Coarse filter	0.01-0.1 μm	0.1-0.2 μm	0.2-0.25 μm	0.25-0.3 μm	0.3-0.4 μm	0.4-0.5 μm	0.5-1 μm	1-2.5 μm	2.5-5 μm	5-10 μm
	$\mu\text{g}/\text{m}^3$									
LIMIT VALUE	3	3	3	3	3	3	4	5	7	15
Coarse filter	4.5	4.4	4.5	4.5	4.5	4.5	4.6	4.7	9.3	7.5
Variable coarse filter	0.14	1.33	1.29	1.56	0.91	0.41	0.17	8.21	6.71	5.02
H13 filter	0.2	0.2	0.2	0.1	0.1	0.1	0.2	0.2	0.3	0.3
Variable H13 filter	0.58	0.48	0.46	0.56	0.28	0.13	0.07	2.83	3.30	2.69

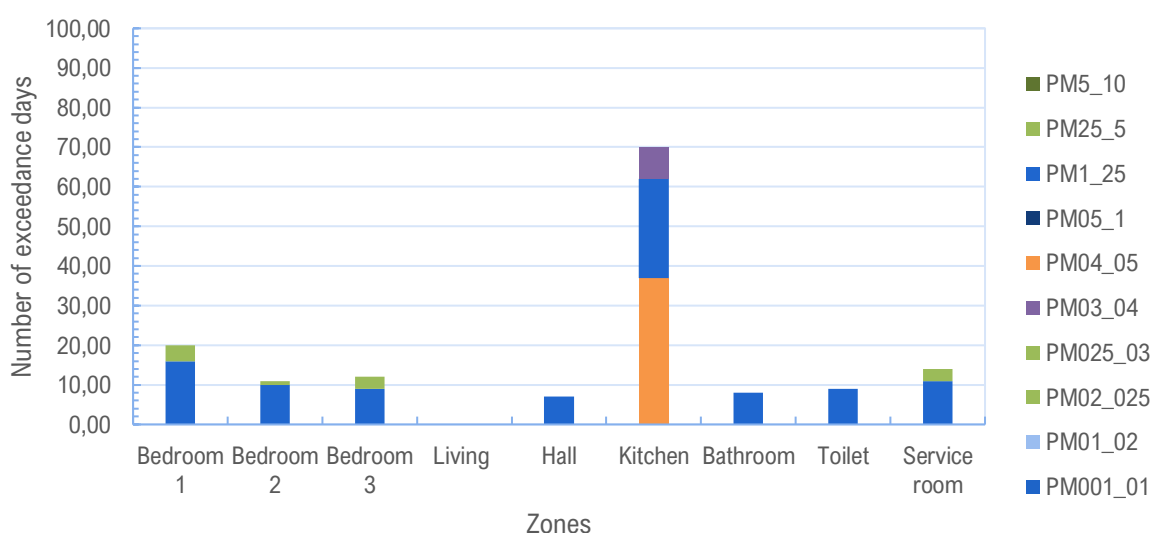
Table 23: Average annual PM concentration in each room with variable filters.

When the occupant with cooking activities and window opening is added, the influence is significant. However, in average concentration, all bin sizes stay below the boundary value. This can be seen in table 24. The difference of these concentrations compared to the ones of the base scenario, with all values exceeding the boundaries, is quite big. Only in the kitchen the boundary value is exceeded for the lowest concentration (cooking related PM). Here source control will be necessary to achieve a healthy environment.

Annual Average Variable H13 filter	0.01-0.1 μm	0.1-0.2 μm	0.2-0.25 μm	0.25-0.3 μm	0.3-0.4 μm	0.4-0.5 μm	0.5-1 μm	1-2.5 μm	2.5-5 μm	5-10 μm
	$\mu\text{g}/\text{m}^3$									
TARGET VALUE	3	3	3	3	3	3	4	5	7	15
Bedroom 1	0.06	0.54	0.54	0.65	0.32	0.14	0.07	3.34	3.97	3.18
Bedroom 2	0.06	0.48	0.48	0.57	0.29	0.13	0.07	2.82	3.18	2.59
Bedroom 3	0.05	0.47	0.45	0.56	0.28	0.13	0.07	2.85	3.19	2.55
Living	0.03	0.24	0.23	0.28	0.14	0.07	0.03	1.46	1.75	1.38
Hall	0.05	0.45	0.42	0.52	0.26	0.13	0.06	2.56	2.85	2.35
Kitchen	3.78	0.66	0.63	0.78	0.39	0.17	0.06	4.08	5.06	4.30
Bathroom	0.05	0.45	0.43	0.52	0.26	0.13	0.09	2.58	2.92	2.38
Toilet	0.05	0.46	0.43	0.54	0.27	0.13	0.07	2.65	3.07	2.50
Service room	1.10	0.53	0.49	0.61	0.31	0.14	0.07	3.07	3.68	2.96

Table 24: Average PM concentration in each room with a variable H13 filter.

As for the exceedance days, there are a few to many. The kitchen has almost 40 for the lowest bin sizes due to cooking related PM. Furthermore, bin size 1-2.5 μm also displays quite some exceedance days in all rooms except the living room. The PM2.5-5 size fraction also shows 4 exceedance days for the first bedroom. Those values can be seen in graph 34.



Graph 34 : Number of exceedance days in the scenario with occupant, window opening and variable filter.

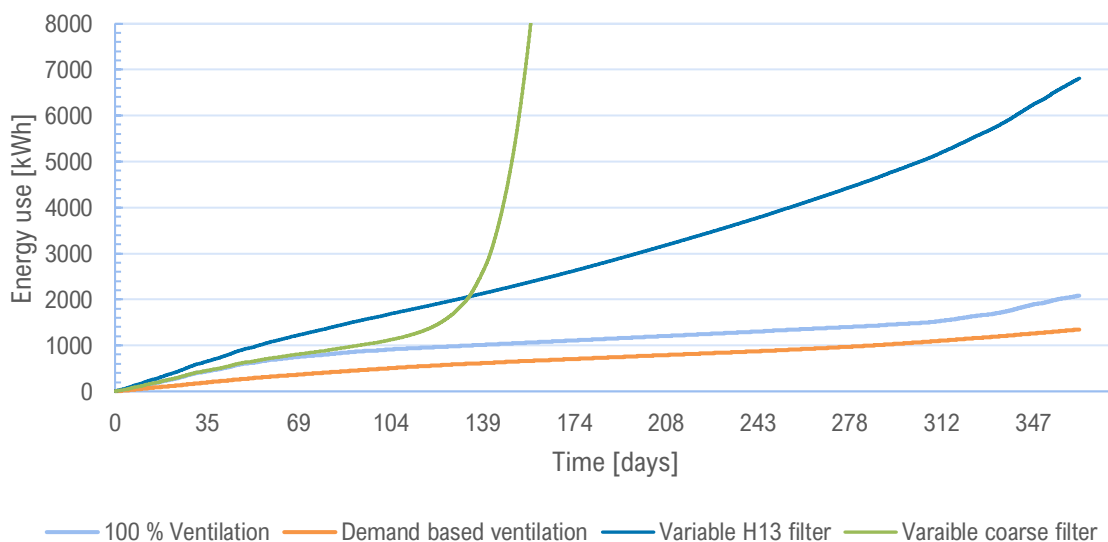
10.8. Energy Use

A ventilation system requires a lot of energy, even more so when a filter is involved. The pressure drop by the filter needs to be overcome. In order to do so, the system requires additional energy.

When no filter was present, the energy use of the building after a year was 2080.84 kWh for both fan and heating. This is when the ventilation system works 100% of the time. When a demand based ventilation system is present, the energy use goes down till 1348.53 kWh. This is a significant decrease, but the indoor quality also decreases with this system.

At the moment when the coarse filter needed replacement the total energy use was 929 kWh. If the filter is not replaced at this time, but at the end of the year, this energy use increases till unprecedented heights.

For the variable H13 filter, the total energy use on the moment when the filter is completely saturated is 3685 kWh. If the filter is not replaced and keeps filtering air, the total energy use after a year will be 6812.5 kWh. Therefore, replacing the filter is very important. So, the addition of the filter increases the energy use significantly, about three times the original energy use without filter. However, if the filter would be replaced, the energy use would be a lot lower (about twice the original energy use). Nevertheless, the influence on the overall building cost is significant. The differences can also be observed in graph 35.



Graph 35: Energy use when variable H13 is used with occupant and window opening.

10.9. DALYs

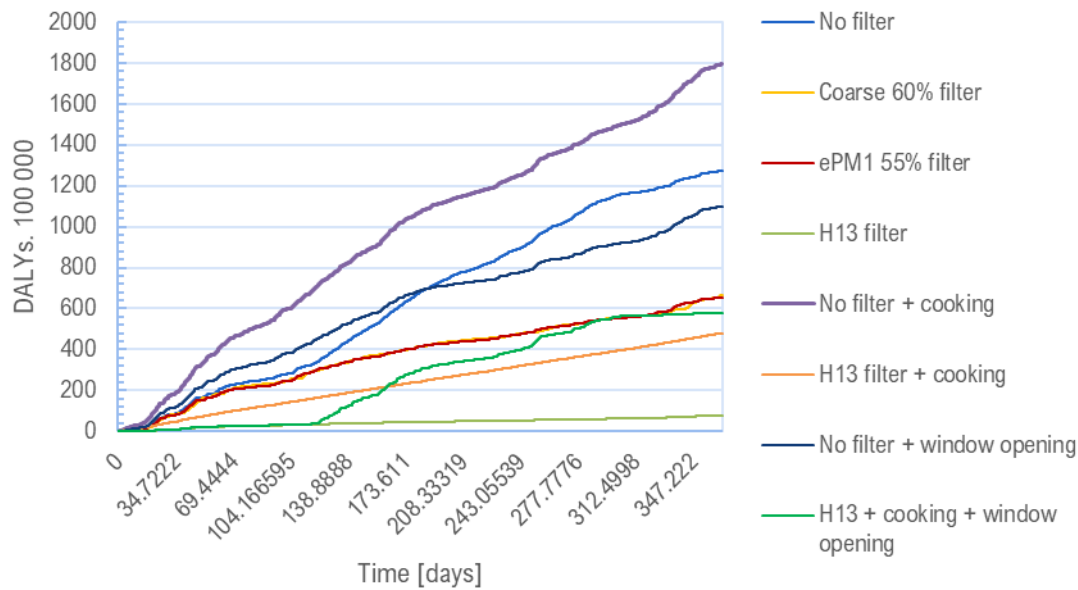
In order to further assess the health impact of particulate matter in different scenarios, the number of DALYs can be calculated for some key scenarios.

In each case, the kitchen is reviewed, since this was the room with the highest concentration of PM and also the room housing the indoor PM source. Since, the number of DALYs is calculated using an effect factor for PM_{2.5}, only bin sizes below this fraction can be taken into account. The higher bin sizes are less harmful and most likely lead to less serious health effects. It is also worth noting that due to the assumption of a maximal concentration in each bin size and the assumption of a worst-case scenario, some bin sizes contain less PM than is assumed in these simulations. For the DALYs calculation, an assumption like in 9.7 is more realistic. In order to calculate a more realistic number of DALYs, the concentrations of all bin sizes below 2.5 µm are combined and multiplied with their percentual share in the Out2In project. To not underestimate the influence of cooking in the smallest bin size, the difference between no cooking and cooking was not multiplied. This gives a total concentration of PM_{2.5} for all abovementioned scenarios. Ultimately, the resulting DALYs after one year of exposure per 100 000 people can be seen in table 25.

Scenario	DALYs.100 000 after one year
No filter	1270
Coarse 60 % filter	666
Fine ePM1 55% filter	655
H13 filter	76.9
No filter + cooking	1800
H13 filter + cooking	479
No filter + window opening	1100
H13 filter + cooking + window opening	580

Table 25: Number of DALYs after one year of exposure in the kitchen.

This shows not only the significant influence of cooking, but also the impact of different filters. The coarse and ePM1 filters are very similar to each other and both cut the number of DALYs in half. The difference with a H13 filter is significant. If cooking is added in this case, about 400 extra DALYs are added. This shows the importance of extra source control for cooking PM. However, the difference with the scenario without filter is enormous. The influence of window opening on the number of DALYs can also be observed in graph 36. The curves that contain both open windows and a filter increase significantly in the summer months (when the window is opened).



Graph 36: Number of DALYs for PM_{2.5} per 100 000 people in function of the time in the kitchen for different scenarios.

10.10. Other pollutants

Particulate matter is of course not the only pollutant present in the air. In the multizone model other pollutants like, NO_2 , O_3 , and various VOC's or volatile organic components were also present. However, not all pollutants are equally dangerous, if the table from the research of Belias & Licina (2024) is observed. The deposition loss rate of formaldehyde and NO_2 is even lower than the rate of PM. These pollutants will stay in the air even longer than PM_{2.5}. Therefore, it is possible to compare the concentrations of these pollutants and to determine which of those pollutants will have the highest impact. In case of O_3 a filtration efficiency of 10% was assumed. The other pollutants are not influenced by the presence of a filter.

Pollutant	Penetration coefficient [-]	Deposition loss rate [h ⁻¹]
PM _{2.5}	0.7	0.5
O ₃	0.54	2.5
NO ₂	0.72	0.27
SO ₂	1	1.72
Formaldehyde	1	0.25

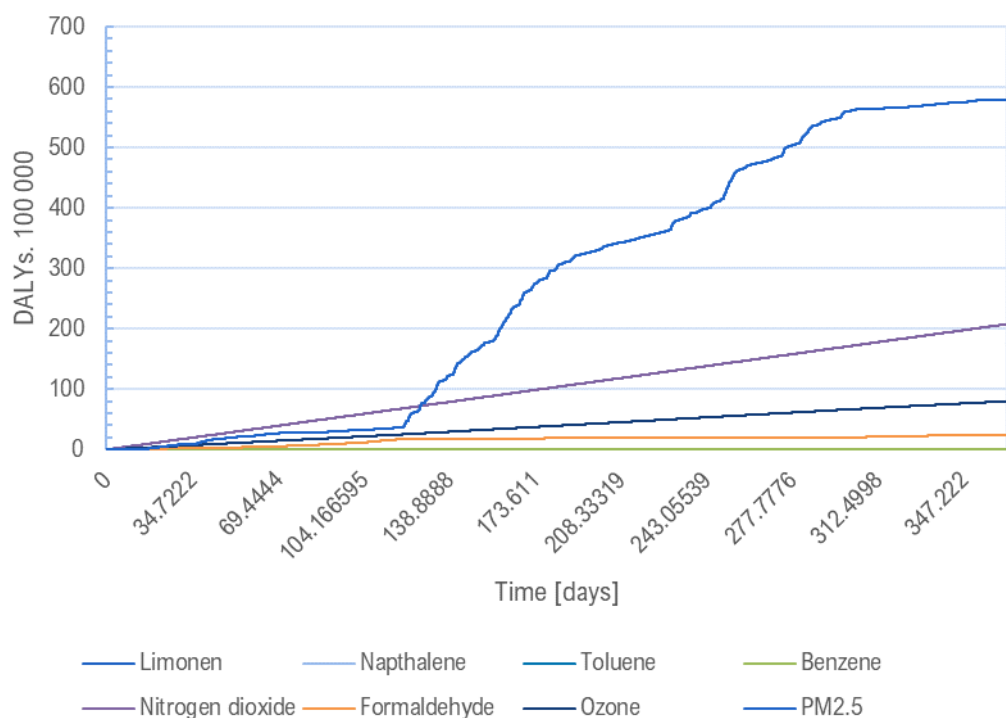
Figure 34: Penetration and deposition loss rate coefficients of some air pollutants (Source: Belias & Licina, 2024).

The number of DALYs can also be calculated for these pollutants. This makes the comparison with PM easier. For NO_2 and O_3 , a constant outdoor concentration was assumed of 27 and 50 $\mu\text{g}/\text{m}^3$ in the multizone model. The number of DALYs for the most realistic scenario (H13 filter & cooking & window opening) can be found in table 30.

Pollutant	DALYs.100 000 after one year
PM2.5	580
Formaldehyde	25
Ozone	80.1
Nitrogen dioxide	208
Benzene	0.0118
Toluene	0.0019
Naphthalene	0.284
Limonene	0.0244

Table 26: Number of DALYs for other pollutants after a year of exposure.

Graph 37 and table 30 show that PM stays the most harmful pollutant. Only if no cooking or window opening were present, this would not be the case. The most harmful pollutants when PM is excluded are nitrogen dioxide, ozone, and formaldehyde. The number of DALYs for benzene, toluene and limonene is negligible.



Graph 37: Number of DALYs for other pollutants per 100 000 people in function of the time in the kitchen.

11.

Conclusion

This dissertation focusses on particulate matter particles and its influential factors. These particles are proven to have a very significant effect on the human health and have been linked to various disease phenomena. Filters are a preferred strategy to minimize these health effects. However, since those filters are tested in labs with particle generator, its effectivity in a real-life environment might be different. This could result in a false sense of security when a filter is present. Therefore, the influence of filters on a realistic particle concentration with numerous prerequisites was investigated.

11.1. PM size distribution

Particulate matter has proven to be a dominant factor in various health effects. Especially the smallest size fractions of PM seem to have a particularly damaging effect on the human health. Therefore, it seems evident that the most effort should be allocated towards keeping the smallest particles out of the indoor environment. However, particulate matter is already able to enter the bloodstream with a size diameter of $2.5\mu\text{m}$, and larger PM can damage the respiratory tract. Therefore, there is no safe size diameter and all sizes should be considered. In short, a general rule of thumb can be formulated: the smaller the particle, the more damaging.

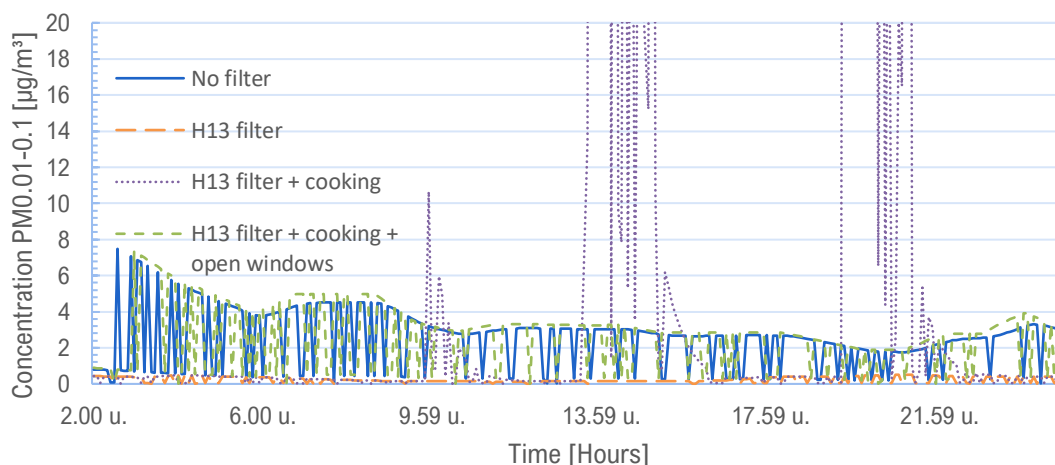
11.2. Indoor air quality

When looking at PM in a simulation environment, conclusions can be made about the use of filters and influential parameters. In Belgium, outdoor PM concentration varies slightly regarding size distribution, but overall, mean concentrations are very similar. In all cases, the outdoor PM concentration is very damaging to the overall health. Only the concentration of the largest size fraction ($5\text{-}10\mu\text{m}$) was sufficiently low. Source control for the outdoor air should be improved significantly in order to achieve a healthy outdoor PM concentration. The difference in size distribution results in a small difference between different locations. Due to their higher content of smaller particles the air is less healthy near large traffic sites than in the city center. And the air in the city center is more damaging than the air in a more rural environment.

High efficiency air filters are mainly used in clean rooms and health care to filter damaging particles from the air, before they reach the indoor environment. In residential buildings it is more common to use coarse, medium, and fine filter to fulfill this purpose. However, these filters are not able to provide a good indoor air quality. The addition of a HEPA filter has a significantly higher influence in this setting when no filter was present. From a health perspective, the use of HEPA filters in a residential environment would be preferred over no filter or a less efficient filter.

When indoor cooking is added as an indoor source, even a H13 filter can't compensate for this increase in indoor concentration in the size fraction most cooking-related PM belongs to. These particles are very small and therefore very damaging, especially to the brain and nervous system. A HEPA filter in the ventilation system and cooking hood with 70% effectivity, are not able to keep this concentration down. An extra type of source control or higher cooking hood efficiency is necessary to intercept these very damaging particles. Not using a continuous ventilation system also has a negative influence on the indoor PM concentration. Regarding PM, 100% ventilation at all times is necessary.

Additionally, when there is no filter present, the influence of window opening can be neglected or even induce a positive effect. But if a filter is present, the window opening undermines the efficiency of the filter. It can be stated that, the higher the filter efficiency, the bigger the influence of window opening. However, window opening does act as source control for cooking related PM. Opening a window significantly decreases the PM peak. However, if windows stay open to long, this positive effect is nullified by the PM that enters via the windows. Graph 38 summarizes the beforementioned findings. An overall comparison of the various annual average concentrations and exceedance days can be found in table 27. Only without additional source and with H13 filter, a sufficiently low PM concentration is obtained.



Graph 38 : Comparison of daily PM concentration in different scenarios.

	Average annual concentration [µg/m³]	Number of exceedance days
TARGET VALUE	3	3
Vilvoorde	7.24	105
Coarse 60% filter	4.66	57
ePM1	3.51	34
H13 filter	0.52	0
Cooking (no filter)	13.25	206
H13 - Cooking	6.55	49
Occupant based ventilation	28.1	338
Window opening (no filter)	7.10	97
Coarse filter – Window Opening	5.74	78
H13 - Window opening	3.19	37
H13 - Window opening - Cooking	6.99	88

Table 27: Overall comparison of annual average concentration and number of exceedance days.

Formulating definitive conclusion from the variable efficiency is difficult since results are not as could be expected. A more detailed approach and adaptation of the model should be performed in order to provide a more definitive conclusion. The addition of a variable outdoor concentration in the test model for the efficiency curves will most likely solve a lot of the uncertainties. This model, however, gives some preliminary results. In terms of efficiency, when a less performative filter is used, the initial drop in efficiency is significant, and overcoming this difference takes longer than the lifetime of the filter. Therefore, this efficiency drop should be tested and described in the standard. When no efficiency drop is present (or is assumed not to be) the overall efficiency increases. The lifetime of filters seems to be closer to 6 months than to 12 months. In any case, the replacement guidelines of a filter should be adapted. As, when a filter is replaced to early, the advantage of the increasing efficiency is negated. And when a filter is used to long, the energy use increases significantly.

If the number of DALYs per 100 000 persons is investigated, earlier findings are confirmed. Cooking shows a significant influence on the overall number of DALYS. A filter is then again able to significantly reduce the number of DALYs. The opening of windows, results in an increase of the number again, but lower than for cooking. If those numbers are compared to those of other pollutants, PM stays the most harmful particle in every scenario. Only when a H13 is used and no cooking is present, NO_2 can inflict more damage. Therefore, this is the most important pollutant except PM. This should however be

investigated with a variable outdoor concentration. Furthermore, ozone and formaldehyde give the highest DALYs values of the remaining pollutants.

To conclude, an architect should, in order to ensure a healthy environment, consider implementing following precautions. Considering the baseline scenario and the influence of window opening on the indoor concentration, a ventilation system D that is constantly used, is necessary to provide an indoor environment with good air quality. This system needs to be equipped with a high efficiency filter. The window opening in this case should be discouraged and avoided when possible. A cooking hood with good capturing efficiency or an additional form of source control will be necessary to minimize the effect of cooking related PM.

11.3. Comparison to the existing multi-zone model

In this dissertation certain additions were made to the existing multi-zone model by De Jonge (2023). Next to the existing 13 pollutant tracers (one of them PM_{2.5}), 10 additional PM tracers were added representing different size fractions. PM_{2.5} was already part of the model, but at a constant outdoor concentration. Firstly, these tracers, required the addition of the different PM sizes to the air. Furthermore, decay constants were added for each tracer. Additionally, the variable outdoor concentrations were added to the outdoor air. This could also allow the variability of the other traces like ozone. As an indoor PM source, cooking-related PM was added to the occupancy model for a fixed schedule based on occupancy schedule 127.

Additionally, a filter component was developed that can be added to the ventilation system to represent an indoor air filter of different efficiencies. Both a filter with continuous and varying filter efficiencies was developed. The filter with varying efficiencies allows to investigate the clogging of the filter and the pressure drop increase (= lifetime of the filter).

The resulting model allows the observation of particulate matter concentration in different scenarios as a varying pollutant with different size fraction. This allows the observation of the toxicity of the remaining particles. As opposed to a fixed PM_{2.5} concentration without any size divisions. Now, the PM concentrations can be observed in a much more detailed fashion.

11.4. Restrictions and future work

During the past year of research, it was often unavoidable to clash with the restrictions of existing data and standards. Although many of these limitations have been addressed previously in this dissertation, a summary can provide a more complete picture of the limitations that need to be conquered to allow for a more complete assessment of particulate matter related health effects in a real-life environment.

Firstly, due to a lack of research and data, the ultrafine particulate matter fraction below $0.01\mu\text{m}$ was left out of this dissertation. These fractions are very hard to measure and generate for testing and are therefore left out in many research papers. These size fraction could, however, prove to be more harmful than others. Analysis of cooking-related PM has shown that most PM emitted due to cooking belongs to this size fraction, so they are abundantly present. More research into the addition of these smaller particles is necessary for a complete health assessment of particulate matter. Additionally, since in this dissertation a fixed week menu was used, different menus could also be investigated in the future.

Furthermore, existing standards often only include PM₁₀, PM_{2.5} and sometimes PM₁ or PM_{0.1}. Many available data and papers only consider those divisions at best. These fractions always include all PM smaller than their boundary diameter. From a health perspective, this also means that some research papers attribute certain health effects to PM₁₀ that could be attributed to PM_{2.5} or smaller instead. This is something to consider when reading different papers.

Therefore, assumptions were often necessary. Making distributions based on interpolation would not be accurate. This resulted in some bin sizes having the same input for input concentrations and settling rate. The assumption of a worst-case scenario for the outdoor PM gives the worst possible outcome for each bin size. But this could also mean that some bin size boundary values would not be exceeded if a size distribution would have been used. Those conclusions would, however, be very place dependent. By using a worst-case scenario, the outcome can be generalized. However, if additional measurements were to become available for more size fractions, they could be revisited and adapted.

In order to compare resulting values with the guidelines of the world health organization, an interpolation was performed for the guideline values for the annual average concentration and the maximum daily concentration. More detailed guidelines for more size fractions could make this comparison more accurate.

Subsequently, the existing testing standards for air quality filters provided some challenges. Firstly, the standards allow manufacturers to choose the air velocity for which a filter is tested. Since this velocity

has a significant influence on the filtration efficiency, comparing different velocities is not possible. However, in order to be able to test both coarse, fine and high efficiency filters, abstraction needed to be made of this problem. The lack of unity in data with regards to airflow velocity, also resulted in the choice to leave out airflow velocity as a tested parameter. The same fate was ascribed to the relative humidity. The conditions of the testing standard are too broad to allow comparison between different humidities, hence only one fixed humidity was considered.

Both parameters: airflow velocity and relative humidity, however, have a proven effect on the indoor PM concentration. Further research into both could be very relevant.

Another factor that was not considered in this dissertation, is the height of the building. Particulate matter concentration is very variable and is different at different heights. For that reason, placing the apartment on different levels could result in different values. Nonetheless, in this dissertation the data from measuring stations were used. The height at which these stations measure data was unknown and therefore not investigated as an influential parameter.

Research into the influence of air tightness of particulate matter was already investigated in the context of natural ventilation by Belias and Licina (2024). This research gives an insight into the influence of different air tightness values on the indoor PM concentration. However, ventilation systems with mechanical exhaust and supply are only present in new buildings, where a good air tightness can be assumed. Hence, air tightness was not one of the conditions investigated in this dissertation.

In future research, the effect of an air purifier (for example in the kitchen) could be investigated as well. This dissertation concluded that additional air quality purifiers are needed in a residential environment to control PM emitted by cooking. An air purifier could possibly be used for this purpose.

Moreover, different filter efficiencies for the cooking hood could be tested as well. Since, cooking proved to have a very large influence on the indoor concentration of the most damaging bin size, a higher source removal rate for this cooker hood could have a significant influence on the indoor concentration. The influence of other indoor PM sources could also be investigated, although those are expected to be minimal compared to the emissions of cooking.

In the model used to investigate clogging and lifetime of a filter abstraction was made of compaction. Moreover, every particle, regardless of its size, was assumed to have the same contribution to the clogging phenomenon. In reality, this phenomenon is of course very present. Furthermore, the model was developed with a fixed PM concentration, since only that data was available. This could have caused the outdoor peaks to have a very high influence, which could be the cause of the unrealistic results for

the coarse filter. If the exact data would become available these curves could be revisited. Additionally, the remaining concentrations for different bin sizes vary a lot more in case of a coarse filter than with a high efficiency filter (efficiency of almost 1 in all cases). Therefore, the efficiency curves changed very quickly when another concentration was filtered. Since all the mass was added to formulate a pressure drop and then later divided again equally, the efficiency increases for all bin sizes went extremely fast. A more elaborate model will be necessary to account for these phenomena. The clogging model can be seen as a first attempt to quantify and assess lifetime of filters, but more details should be developed and included in future research.

12.

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13.

Appendix

Annex A : Bin size schemes

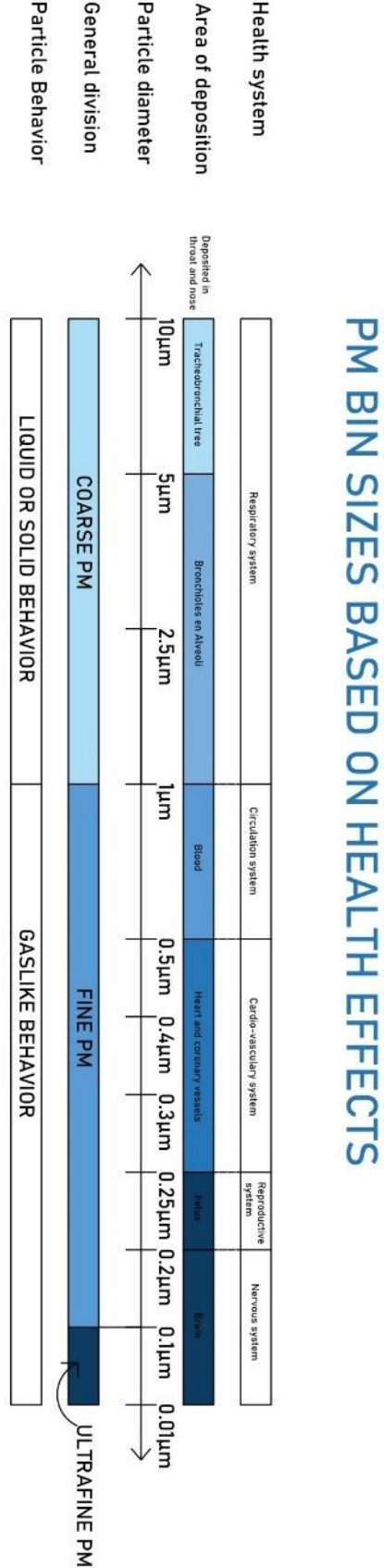


Figure A1 : Bin size scheme based on health effects.

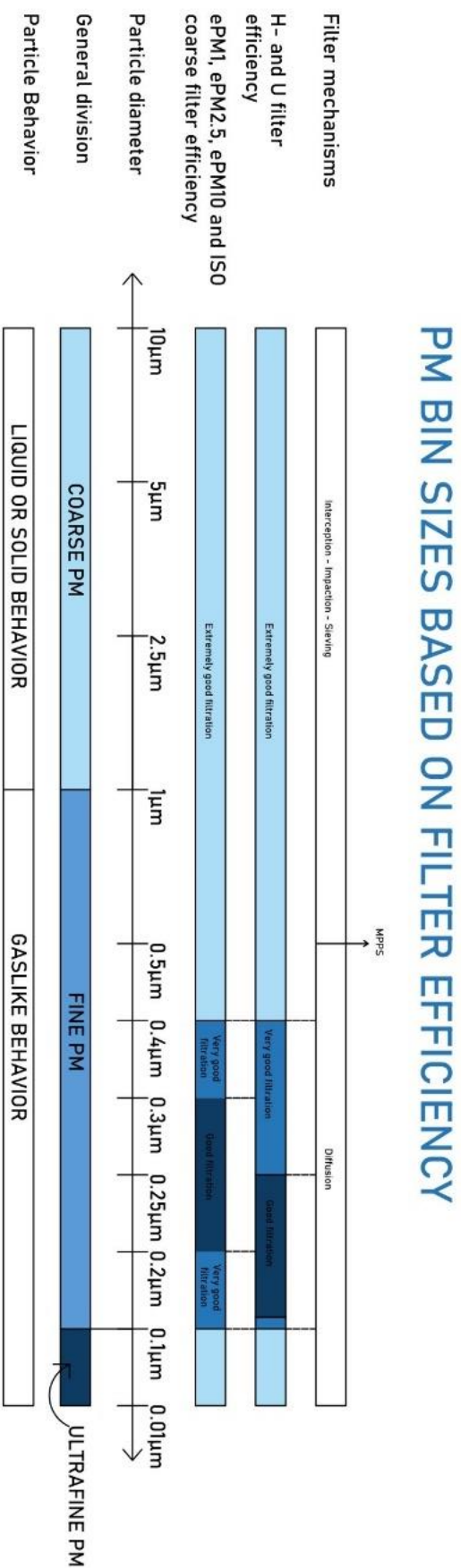


Figure A2 : Bin size scheme based on Filter Efficiency.

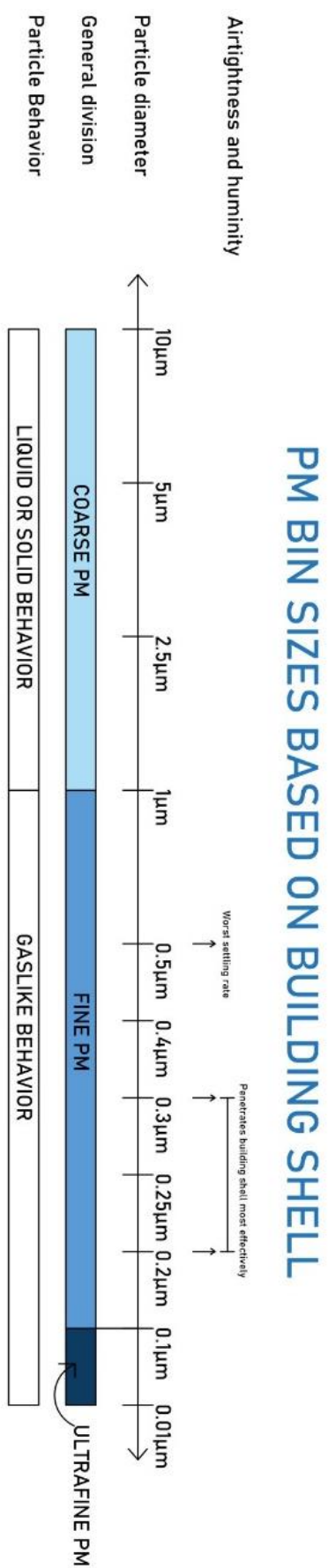


Figure A3 : Bin size scheme based on the Building Shell.

PM BIN SIZES

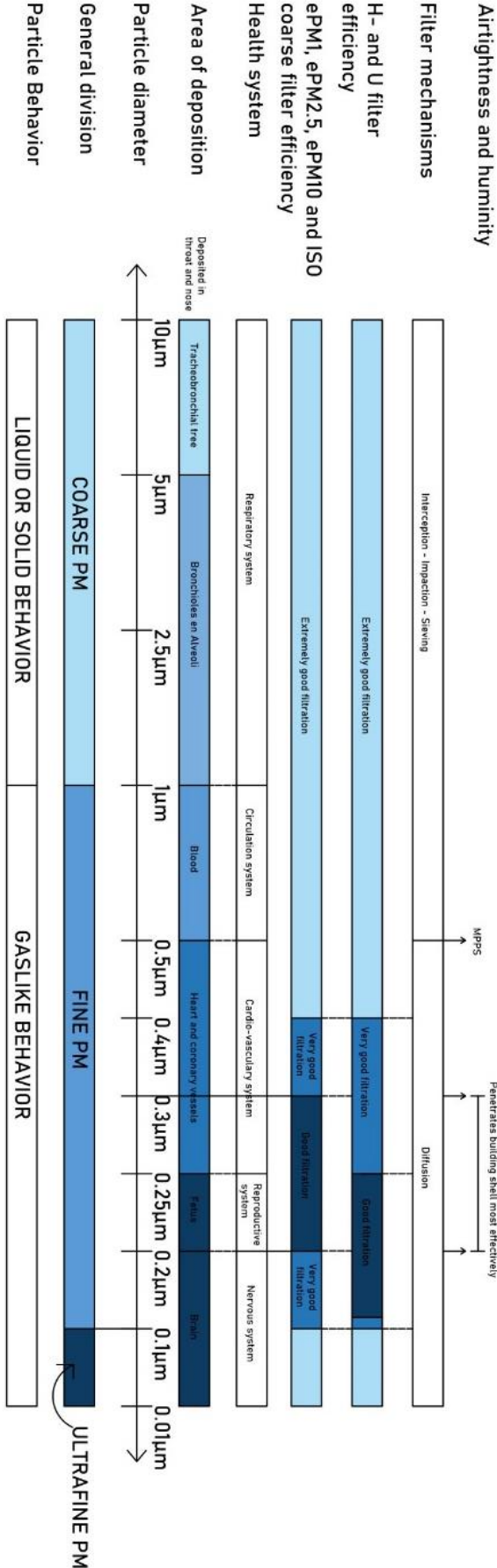


Figure A4 : Summarizing Bin size scheme.

Annex B : Base Scenario Results

Exceedance days Vilvoorde	0.01-0.1 μm	0.1-0.2 μm	0.2-0.25 μm	0.25-0.3 μm	0.3-0.4 μm	0.4-0.5 μm	0.5-1 μm	1-2.5 μm	2.5-5 μm	5-10 μm
	$\mu\text{g}/\text{m}^3$									
Bedroom 1	103	103	103	103	93	93	84	95	90	0
Bedroom 2	103	103	103	103	95	95	86	89	97	0
Bedroom 3	105	105	105	105	95	95	82	89	94	0
Living	101	101	101	101	91	91	81	80	89	0
Hall	99	99	99	99	95	95	86	91	93	0
Kitchen	105	105	105	105	95	95	86	87	91	0
Bathroom	101	101	101	101	94	94	85	87	89	0
Toilet	107	107	107	107	99	99	93	94	110	0
Service room	105	105	105	105	100	100	84	89	90	0

Table B1: Number of days exceeding the 24-hour limit opposed by the WHO in Vilvoorde.

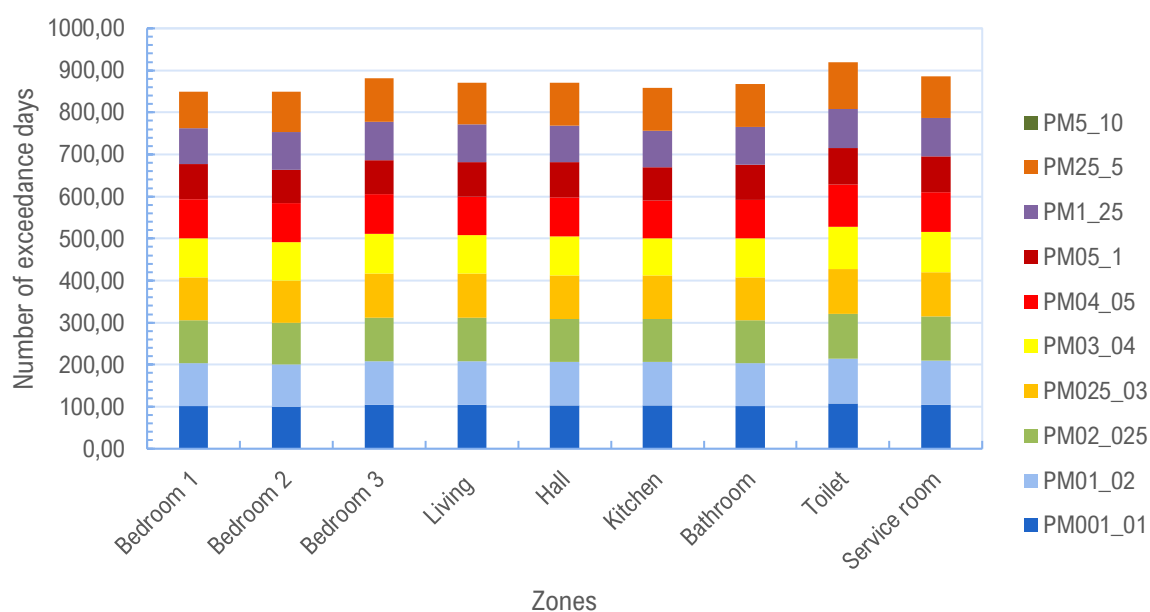
Annex C : Case study 1

Annual average Ghent	0.01-0.1 μm	0.1-0.2 μm	0.2-0.25 μm	0.25-0.3 μm	0.3-0.4 μm	0.4-0.5 μm	0.5-1 μm	1-2.5 μm	2.5-5 μm	5-10 μm
	$\mu\text{g}/\text{m}^3$									
TARGET VALUE	3	3	3	3	3	3	4	5	7	15
Bedroom 1	7.28	7.28	7.28	7.28	7.28	7.28	7.28	9.35	17.92	17.92
Bedroom 2	7.20	7.20	7.20	7.20	7.20	7.20	7.20	9.39	17.97	17.97
Bedroom 3	7.27	7.27	7.27	7.27	7.27	7.27	7.27	9.39	18.04	18.04
Living	7.15	7.15	7.15	7.15	7.15	7.15	7.15	9.18	17.89	17.89
Hall	7.26	7.26	7.26	7.26	7.26	7.26	7.26	9.34	18.02	18.02
Kitchen	7.21	7.21	7.21	7.21	7.21	7.21	7.26	9.32	18.05	18.05
Bathroom	7.20	7.20	7.20	7.20	7.20	7.20	7.21	9.27	18.08	18.08
Toilet	7.52	7.52	7.52	7.52	7.52	7.52	7.52	9.76	18.41	18.41
Service room	7.29	7.29	7.29	7.29	7.29	7.29	7.29	9.53	18.17	18.17

Table C1: Average PM concentration in each room in Ghent in 2023.

Exceedance days Ghent	0.01-0.1 μm	0.1-0.2 μm	0.2-0.25 μm	0.25-0.3 μm	0.3-0.4 μm	0.4-0.5 μm	0.5-1 μm	1-2.5 μm	2.5-5 μm	5-10 μm
	$\mu\text{g}/\text{m}^3$									
Bedroom 1	102	102	102	102	93	93	83	85	87	0
Bedroom 2	100	100	100	100	92	92	80	89	97	0
Bedroom 3	104	104	104	104	95	95	80	92	103	0
Living	104	104	104	104	92	92	82	90	99	0
Hall	103	103	103	103	93	93	83	88	101	0
Kitchen	103	103	103	103	89	89	83	87	102	0
Bathroom	102	102	102	102	92	92	80	90	101	0
Toilet	107	107	107	107	100	100	87	93	112	0
Service room	105	105	105	105	95	95	85	92	99	0

Table C2: Number of days exceeding the 24-hour limit opposed by the WHO in 2023 in Ghent.



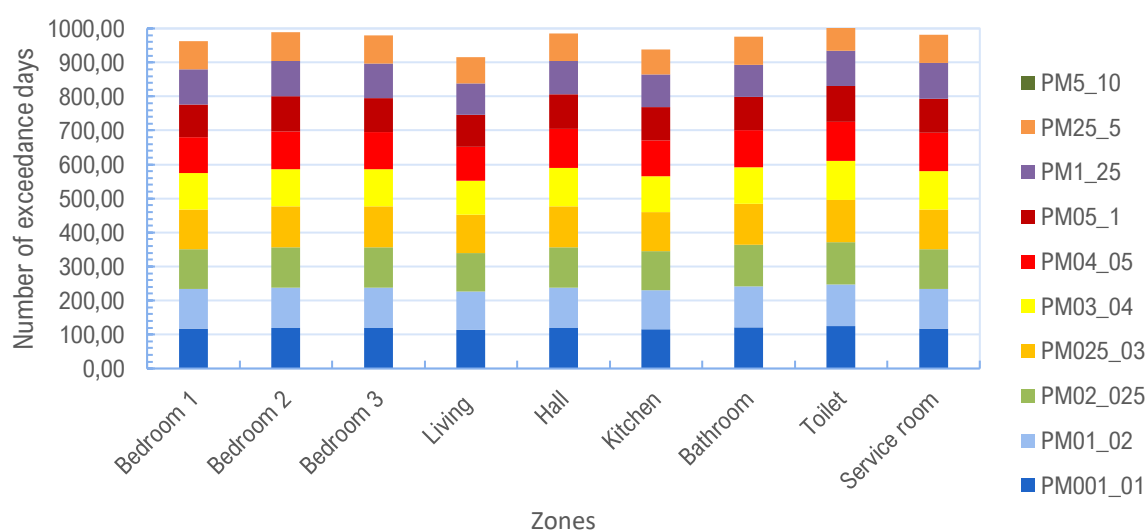
Graph C1: Number of days exceeding the 24-hour limit opposed by the WHO in 2023 in Ghent.

Annual average Antwerp	0.01-0.1 μm	0.1-0.2 μm	0.2-0.25 μm	0.25-0.3 μm	0.3-0.4 μm	0.4-0.5 μm	0.5-1 μm	1-2.5 μm	2.5-5 μm	5-10 μm
	$\mu\text{g}/\text{m}^3$									
TARGET VALUE	3	3	3	3	3	3	4	5	7	15
Bedroom 1	7.89	7.89	7.89	7.89	7.89	7.89	7.89	9.88	17.52	17.52
Bedroom 2	7.96	7.96	7.96	7.96	7.96	7.96	7.96	9.85	17.60	17.60
Bedroom 3	7.91	7.91	7.91	7.91	7.91	7.91	7.91	9.88	17.59	17.59
Living	7.73	7.73	7.73	7.73	7.73	7.73	7.73	9.74	17.32	17.32
Hall	7.93	7.93	7.93	7.93	7.93	7.93	7.93	9.88	17.56	17.56
Kitchen	7.90	7.90	7.90	7.90	7.90	7.90	7.93	9.85	17.43	17.43
Bathroom	7.88	7.88	7.88	7.88	7.88	7.88	7.90	9.80	17.58	17.58
Toilet	8.12	8.12	8.12	8.12	8.12	8.12	8.12	10.15	18.22	18.22
Service room	7.97	7.97	7.97	7.97	7.97	7.97	7.97	9.93	17.68	17.68

Table C3: Average PM concentration in each room in Antwerp in 2023.

Exceedance days Antwerp	0.01-0.1 μm	0.1-0.2 μm	0.2-0.25 μm	0.25-0.3 μm	0.3-0.4 μm	0.4-0.5 μm	0.5-1 μm	1-2.5 μm	2.5-5 μm	5-10 μm
	$\mu\text{g}/\text{m}^3$									
Bedroom 1	117	117	117	117	106	106	97	103	82	0
Bedroom 2	119	119	119	119	111	111	102	104	85	0
Bedroom 3	119	119	119	119	110	110	100	100	83	0
Living	113	113	113	113	100	100	95	91	77	0
Hall	119	119	119	119	114	114	103	98	81	0
Kitchen	115	115	115	115	105	105	103	97	73	0
Bathroom	121	121	121	121	108	108	98	94	83	0
Toilet	124	124	124	124	115	115	104	104	99	0
Service room	117	117	117	117	113	113	100	104	83	0

Table C4: Number of days exceeding the 24-hour limit opposed by the WHO in 2023 in Antwerp.



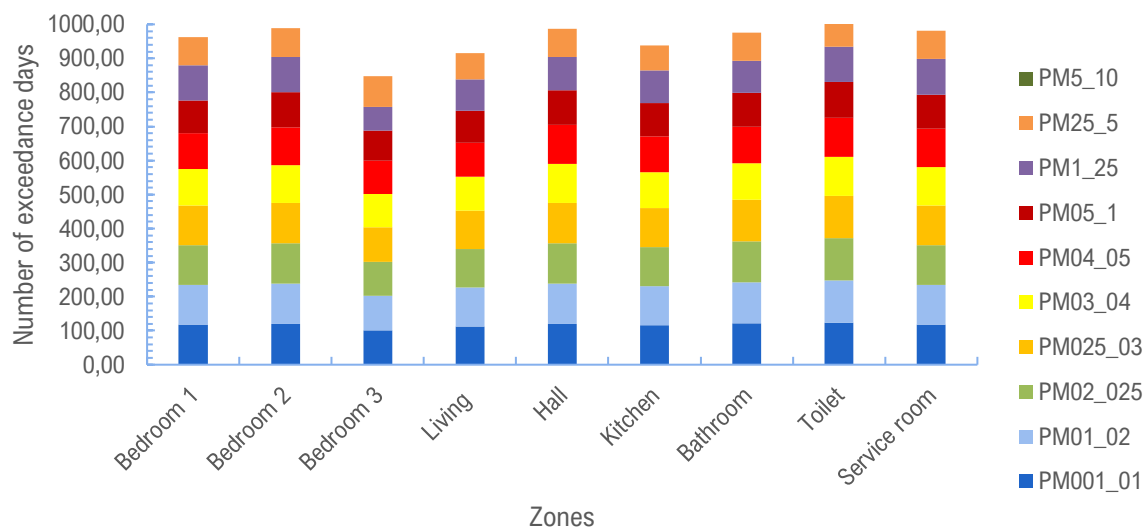
Graph C2: Number of days exceeding the 24-hour limit opposed by the WHO in 2023 in Antwerp.

Annual average Moerkerke	0.01-0.1 μm	0.1-0.2 μm	0.2-0.25 μm	0.25-0.3 μm	0.3-0.4 μm	0.4-0.5 μm	0.5-1 μm	1-2.5 μm	2.5-5 μm	5-10 μm
	$\mu\text{g}/\text{m}^3$									
TARGET VALUE	3	3	3	3	3	3	4	5	7	15
Bedroom 1	7.27	7.27	7.27	7.27	7.27	7.27	7.27	7.95	18.46	18.46
Bedroom 2	7.20	7.20	7.20	7.20	7.20	7.20	7.20	7.84	18.51	18.51
Bedroom 3	7.24	7.24	7.24	7.24	7.24	7.24	7.24	7.89	18.58	18.58
Living	7.17	7.17	7.17	7.17	7.17	7.17	7.17	7.75	18.19	18.19
Hall	7.28	7.28	7.28	7.28	7.28	7.28	7.28	7.93	18.62	18.62
Kitchen	7.18	7.18	7.18	7.18	7.18	7.18	7.28	7.86	18.48	18.48
Bathroom	7.18	7.18	7.18	7.18	7.18	7.18	7.18	7.84	18.46	18.46
Toilet	7.48	7.48	7.48	7.48	7.48	7.48	7.48	8.24	19.02	19.02
Service room	7.26	7.26	7.26	7.26	7.26	7.26	7.26	7.92	18.69	18.69

Table C5: Average PM concentration in each room in Moerkerke in 2023.

Exceedance days Moerkerke	0.01-0.1 μm	0.1-0.2 μm	0.2-0.25 μm	0.25-0.3 μm	0.3-0.4 μm	0.4-0.5 μm	0.5-1 μm	1-2.5 μm	2.5-5 μm	5-10 μm
	$\mu\text{g}/\text{m}^3$									
Bedroom 1	99	99	99	99	93	93	84	72	93	0
Bedroom 2	99	99	99	99	93	93	86	66	96	0
Bedroom 3	101	101	101	101	98	98	87	70	91	0
Living	102	102	102	102	93	93	82	65	87	0
Hall	103	103	103	103	92	92	85	64	102	0
Kitchen	100	100	100	100	94	94	85	69	102	0
Bathroom	101	101	101	101	93	93	87	65	95	0
Toilet	105	105	105	105	97	97	93	75	109	0
Service room	105	105	105	105	91	91	82	73	102	0

Table C6: Number of days exceeding the 24-hour limit opposed by the WHO in 2023 in Moerkerke.



Graph C3: Number of days exceeding the 24-hour limit opposed by the WHO in 2023 in Moerkerke.

Annex D : Case study 2

Annual average Coarse 60 % filter	0.01-0.1 μm	0.1-0.2 μm	0.2-0.25 μm	0.25-0.3 μm	0.3-0.4 μm	0.4-0.5 μm	0.5-1 μm	1-2.5 μm	2.5-5 μm	5-10 μm
	$\mu\text{g}/\text{m}^3$									
TARGET VALUE	3	3	3	3	3	3	4	5	7	15
Bedroom 1	4.33	4.37	4.37	4.36	4.40	4.40	4.45	4.59	8.93	7.00
Bedroom 2	4.39	4.39	4.39	4.41	4.42	4.48	4.47	4.63	9.22	7.36
Bedroom 3	4.34	4.39	4.35	4.35	4.34	4.39	4.49	4.60	9.03	7.16
Living	4.35	4.36	4.42	4.40	4.39	4.42	4.45	4.55	8.95	7.09
Hall	4.43	4.36	4.45	4.48	4.49	4.46	4.50	4.62	9.17	7.27
Kitchen	4.66	4.66	4.71	4.74	4.72	4.70	4.50	4.94	9.93	8.40
Bathroom	4.38	4.36	4.39	4.33	4.42	4.42	4.81	4.62	9.16	7.27
Toilet	4.60	4.54	4.54	4.58	4.65	4.66	4.69	4.86	9.79	7.92
Service room	4.58	4.55	4.55	4.58	4.61	4.59	4.67	4.83	9.60	7.98

Table D1: Average PM concentration in each room with coarse 60% filter.

Exceedance days coarse 60% filter	0.01-0.1 μm	0.1-0.2 μm	0.2-0.25 μm	0.25-0.3 μm	0.3-0.4 μm	0.4-0.5 μm	0.5-1 μm	1-2.5 μm	2.5-5 μm	5-10 μm
	$\mu\text{g}/\text{m}^3$									
Bedroom 1	50	50	50	50	45	45	43	21	19	0
Bedroom 2	52	51	51	51	49	51	40	21	25	0
Bedroom 3	51	51	52	52	45	50	45	19	19	0
Living	52	51	50	48	47	47	40	19	19	0
Hall	52	53	52	51	46	49	41	20	25	0
Kitchen	57	56	56	57	52	53	41	23	26	0
Bathroom	50	52	51	47	47	46	48	22	21	0
Toilet	55	56	52	53	49	53	45	23	27	0
Service room	52	55	55	52	52	50	46	25	28	0

Table D2: Number of days exceeding the 24-hour limit imposed by the WHO with coarse filter.

Annual average ePM1 55% filter	0.01-0.1 μm	0.1-0.2 μm	0.2-0.25 μm	0.25-0.3 μm	0.3-0.4 μm	0.4-0.5 μm	0.5-1 μm	1-2.5 μm	2.5-5 μm	5-10 μm
	$\mu\text{g}/\text{m}^3$									
TARGET VALUE	3	3	3	3	3	3	4	5	7	15
Bedroom 1	3.11	3.16	3.16	3.51	3.74	3.97	4.67	5.40	10.85	9.00
Bedroom 2	3.13	3.28	3.28	3.55	3.75	3.98	4.67	5.42	10.96	9.05
Bedroom 3	3.10	3.20	3.37	3.49	3.69	3.97	4.70	5.40	11.03	8.95
Living	3.12	3.23	3.40	3.53	3.71	4.00	4.66	5.42	10.86	8.92
Hall	3.15	3.25	3.48	3.60	3.70	4.05	4.72	5.52	11.00	9.07
Kitchen	3.51	3.56	3.82	4.00	4.06	4.34	4.72	5.60	11.51	9.74
Bathroom	3.09	3.21	3.41	3.49	3.68	3.96	4.94	5.44	11.02	9.02
Toilet	3.26	3.39	3.61	3.74	3.94	4.21	4.87	5.73	11.57	9.54
Service room	3.31	3.43	3.60	3.73	4.01	4.16	4.85	5.65	11.40	9.52

Table D3: Average PM concentration in each room with a fine filter.

Exceedance days ePM1 55% filter	0.01-0.1 μm	0.1-0.2 μm	0.2-0.25 μm	0.25-0.3 μm	0.3-0.4 μm	0.4-0.5 μm	0.5-1 μm	1-2.5 μm	2.5-5 μm	5-10 μm
	$\mu\text{g}/\text{m}^3$									
Bedroom 1	24	23	23	33	32	36	43	32	27	0
Bedroom 2	24	27	27	33	32	35	45	33	25	0
Bedroom 3	24	25	28	33	28	37	46	34	31	0
Living	25	27	33	35	33	36	44	32	23	0
Hall	26	28	33	32	32	37	49	38	27	0
Kitchen	34	34	37	38	37	44	49	36	35	0
Bathroom	27	29	30	31	30	36	56	33	24	0
Toilet	27	30	34	37	33	39	49	42	36	0
Service room	30	34	36	38	37	39	49	40	27	0

Table D4: Number of days exceeding the 24-hour limit opposed by the WHO with fine filter.

Annual average H13 filter	0.01-0.1 μm	0.1-0.2 μm	0.2-0.25 μm	0.25-0.3 μm	0.3-0.4 μm	0.4-0.5 μm	0.5-1 μm	1-2.5 μm	2.5-5 μm	5-10 μm
	$\mu\text{g}/\text{m}^3$									
TARGET VALUE	3	3	3	3	3	3	4	5	7	15
Bedroom 1	0.03	0.03	0.03	0.03	0.03	0.03	0.03	0.03	0.06	0.06
Bedroom 2	0.10	0.10	0.10	0.09	0.09	0.09	0.09	0.11	0.22	0.21
Bedroom 3	0.04	0.04	0.03	0.03	0.03	0.03	0.03	0.04	0.07	0.07
Living	0.16	0.16	0.16	0.16	0.16	0.16	0.16	0.18	0.32	0.32
Hall	0.09	0.09	0.09	0.09	0.09	0.09	0.09	0.10	0.18	0.18
Kitchen	0.52	0.52	0.51	0.51	0.51	0.51	0.09	0.60	1.20	1.19
Bathroom	0.08	0.09	0.08	0.08	0.08	0.08	0.51	0.10	0.17	0.17
Toilet	0.09	0.09	0.09	0.09	0.09	0.09	0.09	0.10	0.19	0.19
Service room	0.26	0.26	0.26	0.26	0.26	0.26	0.26	0.30	0.52	0.53

Table D5: Average PM concentration in each room with a H13 filter.

Exceedance days H13 filter	0.01-0.1 μm	0.1-0.2 μm	0.2-0.25 μm	0.25-0.3 μm	0.3-0.4 μm	0.4-0.5 μm	0.5-1 μm	1-2.5 μm	2.5-5 μm	5-10 μm
	$\mu\text{g}/\text{m}^3$									
Bedroom 1	0	0	0	0	0	0	0	0	0	0
Bedroom 2	0	0	0	0	0	0	0	0	0	0
Bedroom 3	0	0	0	0	0	0	0	0	0	0
Living	0	0	0	0	0	0	0	0	0	0
Hall	0	0	0	0	0	0	0	0	0	0
Kitchen	0	0	0	0	0	0	0	0	0	0
Bathroom	0	0	0	0	0	0	0	0	0	0
Toilet	0	0	0	0	0	0	0	0	0	0
Service room	0	0	0	0	0	0	0	0	0	0

Table D6: Number of days exceeding the 24-hour limit opposed by the WHO with H13 filter.

Annex E : Case study 3

Annual average Occupant	0.01-0.1 μm	0.1-0.2 μm	0.2-0.25 μm	0.25-0.3 μm	0.3-0.4 μm	0.4-0.5 μm	0.5-1 μm	1-2.5 μm	2.5-5 μm	5-10 μm
	$\mu\text{g}/\text{m}^3$									
LIMIT VALUE	3	3	3	3	3	3	4	5	7	15
Bedroom 1	7.22	7.22	7.22	7.22	7.22	7.22	7.22	8.88	17.46	17.57
Bedroom 2	7.23	7.23	7.23	7.23	7.23	7.23	7.23	8.97	17.52	17.58
Bedroom 3	7.19	7.19	7.19	7.19	7.19	7.19	7.19	9.06	17.55	17.64
Living	7.20	7.20	7.20	7.20	7.20	7.20	7.20	8.87	17.32	17.31
Hall	7.28	7.28	7.28	7.28	7.28	7.28	7.28	8.99	17.54	17.65
Kitchen	13.25	7.19	7.19	7.19	7.19	7.19	7.28	8.95	17.37	17.49
Bathroom	7.21	7.21	7.21	7.21	7.21	7.21	7.19	8.95	17.55	17.63
Toilet	7.46	7.46	7.46	7.46	7.46	7.46	7.46	9.35	18.03	18.16
Service room	9.07	7.32	7.32	7.32	7.32	7.32	7.32	9.13	17.75	17.70

Table E1: Average annual PM concentration in each room with occupant.

Number of exceedance days Occupant	0.01-0.1 μm	0.1-0.2 μm	0.2-0.25 μm	0.25-0.3 μm	0.3-0.4 μm	0.4-0.5 μm	0.5-1 μm	1-2.5 μm	2.5-5 μm	5-10 μm
	$\mu\text{g}/\text{m}^3$									
Bedroom 1	101	101	101	101	90	90	80	85	88	0
Bedroom 2	98	98	98	98	93	93	83	90	99	0
Bedroom 3	105	105	105	105	92	92	81	93	92	0
Living	98	98	98	98	94	94	87	89	90	0
Hall	103	103	103	103	91	91	85	90	84	0
Kitchen	206	107	107	107	95	95	85	85	87	0
Bathroom	103	103	103	103	94	94	88	90	96	0
Toilet	112	112	112	112	103	103	89	99	92	0
Service room	124	103	103	103	96	96	88	92	94	0

Table E2: Number of days exceeding the 24-hour limit imposed by the WHO with occupant.

Annual average Occupant Coarse 60% filter	0.01-0.1 μm	0.1-0.2 μm	0.2-0.25 μm	0.25-0.3 μm	0.3-0.4 μm	0.4-0.5 μm	0.5-1 μm	1-2.5 μm	2.5-5 μm	5-10 μm
	$\mu\text{g}/\text{m}^3$									
LIMIT VALUE	3	3	3	3	3	3	4	5	7	15
Bedroom 1	4.24	4.18	4.18	4.25	4.26	4.31	4.26	4.49	8.73	6.85
Bedroom 2	4.23	4.22	4.22	4.33	4.30	4.33	4.43	4.51	8.92	7.26
Bedroom 3	4.28	4.22	4.23	4.23	4.28	4.31	4.39	4.51	8.89	6.98
Living	4.25	4.21	4.25	4.23	4.20	4.31	4.31	4.52	8.72	6.95
Hall	4.30	4.31	4.30	4.26	4.36	4.35	4.39	4.55	9.04	7.18
Kitchen	10.56	4.49	4.57	4.60	4.57	4.64	4.39	4.86	9.78	8.32
Bathroom	4.23	4.27	4.22	4.26	4.28	4.31	4.65	4.51	8.91	7.07
Toilet	4.43	4.51	4.46	4.50	4.45	4.55	4.53	4.76	9.46	7.72
Service room	6.15	4.43	4.46	4.46	4.48	4.45	4.58	4.75	9.50	7.82

Table E3: Average annual PM concentration in each room with coarse filter and occupant.

Number of exceedance days Occupant coarse 60 % filter	0.01-0.1 μm	0.1-0.2 μm	0.2-0.25 μm	0.25-0.3 μm	0.3-0.4 μm	0.4-0.5 μm	0.5-1 μm	1-2.5 μm	2.5-5 μm	5-10 μm
	$\mu\text{g}/\text{m}^3$									
Bedroom 1	45	43	43	44	41	40	38	18	14	0
Bedroom 2	46	46	46	48	44	44	41	18	19	0
Bedroom 3	45	47	47	48	42	43	39	20	21	0
Living	44	44	43	46	39	43	36	21	16	0
Hall	48	46	47	48	43	46	42	22	20	0
Kitchen	138	53	54	55	50	51	42	22	20	0
Bathroom	47	48	43	49	39	43	44	19	20	0
Toilet	50	51	52	53	43	47	41	25	17	0
Service room	61	52	48	50	45	47	49	24	20	0

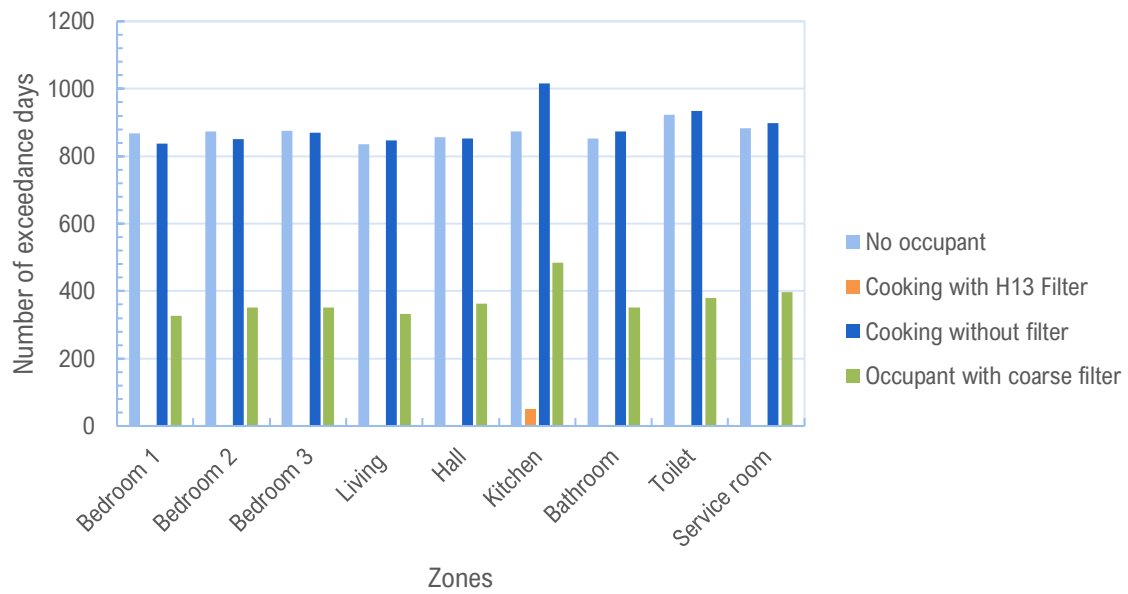
Table E4: Average annual PM concentration in each room with coarse filter with occupant.

Annual average Occupant H13 filter	0.01-0.1 μm	0.1-0.2 μm	0.2-0.25 μm	0.25-0.3 μm	0.3-0.4 μm	0.4-0.5 μm	0.5-1 μm	1-2.5 μm	2.5-5 μm	5-10 μm
	$\mu\text{g}/\text{m}^3$									
LIMIT VALUE	3	3	3	3	3	3	4	5	7	15
Bedroom 1	0.03	0.03	0.03	0.03	0.03	0.03	0.03	0.03	0.06	0.06
Bedroom 2	0.10	0.10	0.10	0.09	0.09	0.09	0.09	0.11	0.22	0.22
Bedroom 3	0.04	0.04	0.03	0.03	0.03	0.03	0.03	0.04	0.07	0.07
Living	0.16	0.16	0.16	0.16	0.16	0.16	0.16	0.18	0.32	0.32
Hall	0.09	0.09	0.09	0.09	0.09	0.09	0.09	0.10	0.18	0.18
Kitchen	6.55	0.52	0.52	0.52	0.52	0.52	0.09	0.60	1.19	1.21
Bathroom	0.08	0.09	0.08	0.08	0.08	0.08	0.52	0.10	0.17	0.18
Toilet	0.09	0.09	0.09	0.09	0.09	0.09	0.09	0.10	0.19	0.19
Service room	2.24	0.26	0.26	0.26	0.26	0.26	0.26	0.30	0.53	0.52

Table E5: Average annual PM concentration in each room with H13 filter and occupant.

Number of exceedance days Occupant H13 filter	0.01-0.1 μm	0.1-0.2 μm	0.2-0.25 μm	0.25-0.3 μm	0.3-0.4 μm	0.4-0.5 μm	0.5-1 μm	1-2.5 μm	2.5-5 μm	5-10 μm
	$\mu\text{g}/\text{m}^3$									
Bedroom 1	0	0	0	0	0	0	0	0	0	0
Bedroom 2	0	0	0	0	0	0	0	0	0	0
Bedroom 3	0	0	0	0	0	0	0	0	0	0
Living	0	0	0	0	0	0	0	0	0	0
Hall	0	0	0	0	0	0	0	0	0	0
Kitchen	49	0	0	0	0	0	0	0	0	0
Bathroom	0	0	0	0	0	0	0	0	0	0
Toilet	0	0	0	0	0	0	0	0	0	0
Service room	0	0	0	0	0	0	0	0	0	0

Table E6: Average annual PM concentration in each room with H13 filter with occupant.



Graph E1: Number of days exceeding the 24-hour limit opposed by the WHO for different scenarios.

Annex F : Case study 4

Annual average demand based	0.01-0.1 μm	0.1-0.2 μm	0.2-0.25 μm	0.25-0.3 μm	0.3-0.4 μm	0.4-0.5 μm	0.5-1 μm	1-2.5 μm	2.5-5 μm	5-10 μm
	$\mu\text{g}/\text{m}^3$									
LIMIT VALUE	3	3	3	3	3	3	4	5	7	15
Bedroom 1	7.57	7.58	7.58	7.58	7.58	7.58	7.58	9.49	18.58	18.59
Bedroom 2	7.63	7.67	7.67	7.67	7.67	7.67	7.67	9.54	18.84	18.72
Bedroom 3	7.59	7.60	7.60	7.60	7.60	7.60	7.60	9.54	18.68	18.66
Living	7.22	7.19	7.19	7.19	7.19	7.19	7.19	9.05	17.98	17.92
Hall	7.81	7.74	7.74	7.74	7.74	7.74	7.74	9.53	18.69	18.86
Kitchen	28.10	7.53	7.53	7.53	7.53	7.53	7.74	9.38	18.46	18.50
Bathroom	8.00	7.93	7.93	7.93	7.93	7.93	7.53	10.04	19.29	19.27
Toilet	10.09	10.01	10.01	10.01	10.01	10.01	10.01	12.53	24.77	24.54
Service room	17.11	7.92	7.92	7.92	7.92	7.92	7.92	9.92	19.06	19.10

Table F1: Average annual PM concentration in each room with demand-based ventilation.

Exceedance days demand-based ventilation	0.01-0.1 μm	0.1-0.2 μm	0.2-0.25 μm	0.25-0.3 μm	0.3-0.4 μm	0.4-0.5 μm	0.5-1 μm	1-2.5 μm	2.5-5 μm	5-10 μm
	$\mu\text{g}/\text{m}^3$									
Bedroom 1	106	108	108	108	97	97	94	105	112	0
Bedroom 2	113	112	112	112	105	105	98	104	117	0
Bedroom 3	110	110	110	110	101	101	95	107	110	0
Living	98	99	99	99	90	90	87	93	108	0
Hall	111	111	111	111	100	100	98	107	112	0
Kitchen	338	109	109	109	99	99	98	102	114	0
Bathroom	118	114	114	114	106	106	93	106	120	0
Toilet	155	154	154	154	150	150	141	162	217	0
Service room	290	117	117	117	106	106	99	108	115	0

Table F2: Number of days exceeding the 24-hour limit opposed by the WHO with demand based ventilation.

Annex G : Case study 5

Annual average Window opening	0.01-0.1 μm	0.1-0.2 μm	0.2-0.25 μm	0.25-0.3 μm	0.3-0.4 μm	0.4-0.5 μm	0.5-1 μm	1-2.5 μm	2.5-5 μm	5-10 μm
	$\mu\text{g}/\text{m}^3$									
LIMIT VALUE	3	3	3	3	3	3	4	5	7	15
Bedroom 1	7.12	7.12	7.12	7.12	7.12	7.12	7.12	8.89	17.35	17.59
Bedroom 2	7.18	7.18	7.18	7.18	7.18	7.18	7.18	8.95	17.56	17.44
Bedroom 3	7.15	7.15	7.15	7.15	7.15	7.15	7.15	8.86	17.40	17.38
Living	7.10	7.10	7.10	7.10	7.10	7.10	7.10	8.79	17.40	17.37
Hall	7.22	7.22	7.22	7.22	7.22	7.22	7.22	9.02	17.48	17.56
Kitchen	7.10	7.10	7.10	7.10	7.10	7.10	7.22	8.82	17.30	17.39
Bathroom	7.15	7.15	7.15	7.15	7.15	7.15	7.10	8.79	17.49	17.50
Toilet	7.39	7.39	7.39	7.39	7.39	7.39	7.39	9.30	17.90	17.96
Service room	7.20	7.20	7.20	7.20	7.20	7.20	7.20	9.02	17.52	17.57

Table G1: Average annual PM concentration in each room with window opening.

Number of exceedance days windows	0.01-0.1 μm	0.1-0.2 μm	0.2-0.25 μm	0.25-0.3 μm	0.3-0.4 μm	0.4-0.5 μm	0.5-1 μm	1-2.5 μm	2.5-5 μm	5-10 μm
	$\mu\text{g}/\text{m}^3$									
Bedroom 1	96	96	96	96	92	92	79	84	84	0
Bedroom 2	101	101	101	101	91	91	82	85	91	0
Bedroom 3	100	100	100	100	89	89	87	85	88	0
Living	99	99	99	99	94	94	79	82	89	0
Hall	100	100	100	100	90	90	84	83	90	0
Kitchen	97	97	97	97	88	88	84	86	87	0
Bathroom	103	104	104	104	93	93	79	83	86	0
Toilet	108	108	108	108	100	100	90	95	90	0
Service room	101	101	101	101	96	96	85	87	93	0

Table G2: Number of days exceeding the 24-hour limit opposed by the WHO with window opening.

Annual average Coarse 60% filter with window opening	0.01-0.1 μm	0.1-0.2 μm	0.2-0.25 μm	0.25-0.3 μm	0.3-0.4 μm	0.4-0.5 μm	0.5-1 μm	1-2.5 μm	2.5-5 μm	5-10 μm
	$\mu\text{g}/\text{m}^3$									
TARGET VALUE	3	3	3	3	3	3	4	5	7	15
Bedroom 1	5.56	5.46	5.46	5.55	5.58	5.61	5.58	6.38	12.97	12.07
Bedroom 2	5.31	5.33	5.33	5.43	5.37	5.39	5.45	6.18	12.56	11.69
Bedroom 3	5.37	5.36	5.40	5.41	5.40	5.41	5.49	6.21	12.52	11.76
Living	4.72	4.71	4.74	4.77	4.76	4.72	4.84	5.24	10.34	9.03
Hall	5.25	5.24	5.38	5.28	5.30	5.31	5.39	6.09	12.36	11.37
Kitchen	5.74	5.81	5.79	5.80	5.81	5.81	5.39	6.78	13.71	13.04
Bathroom	5.16	5.26	5.23	5.24	5.32	5.26	5.83	5.96	12.40	11.44
Toilet	5.46	5.44	5.45	5.52	5.56	5.51	5.61	6.34	12.88	11.92
Service room	5.48	5.63	5.58	5.59	5.61	5.59	5.57	6.39	13.02	12.16

Table G3: Average annual PM concentration in each room with coarse filter and window opening.

Number of exceedance days Coarse 60% filter with window opening	0.01-0.1 μm	0.1-0.2 μm	0.2-0.25 μm	0.25-0.3 μm	0.3-0.4 μm	0.4-0.5 μm	0.5-1 μm	1-2.5 μm	2.5-5 μm	5-10 μm
	$\mu\text{g}/\text{m}^3$									
Bedroom 1	73	70	70	72	63	63	59	41	45	0
Bedroom 2	62	62	62	66	58	62	55	42	35	0
Bedroom 3	62	63	62	65	58	61	57	41	33	0
Living	48	51	50	53	43	43	44	27	19	0
Hall	60	61	61	60	57	54	49	37	35	0
Kitchen	78	78	77	77	68	70	49	49	54	0
Bathroom	60	63	62	63	59	54	64	40	32	0
Toilet	65	64	68	69	63	60	59	45	31	0
Service room	69	72	73	70	65	64	57	46	46	0

Table G4: Number of days exceeding the 24-hour limit opposed by the WHO with coarse filter and open windows.

Annual average H13 filter with window opening	0.01-0.1 μm	0.1-0.2 μm	0.2-0.25 μm	0.25-0.3 μm	0.3-0.4 μm	0.4-0.5 μm	0.5-1 μm	1-2.5 μm	2.5-5 μm	5-10 μm
	$\mu\text{g}/\text{m}^3$									
TARGET VALUE	3	3	3	3	3	3	4	5	7	15
Bedroom 1	2.59	2.65	2.65	2.63	2.63	2.63	2.63	3.33	8.31	8.27
Bedroom 2	2.32	2.28	2.28	2.31	2.31	2.31	2.31	2.92	7.40	7.48
Bedroom 3	2.28	2.27	2.27	2.28	2.28	2.28	2.28	2.91	7.45	7.48
Living	1.13	1.12	1.14	1.12	1.12	1.12	1.12	1.46	3.38	3.43
Hall	2.11	2.16	2.08	2.11	2.11	2.11	2.11	2.67	7.02	7.04
Kitchen	3.19	3.15	3.17	3.11	3.11	3.11	2.11	4.16	9.28	9.35
Bathroom	2.11	2.08	2.08	2.11	2.11	2.11	3.11	2.60	7.06	7.03
Toilet	2.19	2.21	2.18	2.21	2.21	2.21	2.21	2.73	7.24	7.13
Service room	2.46	2.44	2.45	2.48	2.48	2.48	2.48	3.13	7.75	7.76

Table G5: Average annual PM concentration in each room with H13 filter and window opening.

Number of exceedance days H13 filter with window opening	0.01-0.1 μm	0.1-0.2 μm	0.2-0.25 μm	0.25-0.3 μm	0.3-0.4 μm	0.4-0.5 μm	0.5-1 μm	1-2.5 μm	2.5-5 μm	5-10 μm
	$\mu\text{g}/\text{m}^3$									
Bedroom 1	22	22	22	20	19	19	17	15	24	0
Bedroom 2	17	20	20	18	15	15	13	13	14	0
Bedroom 3	14	16	20	18	16	16	13	7	13	0
Living	0	0	1	0	0	0	0	0	0	0
Hall	13	16	14	11	10	10	10	7	19	0
Kitchen	37	39	36	33	27	27	10	29	38	0
Bathroom	14	16	13	13	11	11	22	8	18	0
Toilet	12	15	14	13	11	11	10	8	14	0
Service room	19	17	16	16	14	14	13	11	14	0

Table G6: Number of days exceeding the 24-hour limit imposed by the WHO with H13 filter and open windows.

Annual Average H13 with window opening and cooking	0.01-0.1 μm	0.1-0.2 μm	0.2-0.25 μm	0.25-0.3 μm	0.3-0.4 μm	0.4-0.5 μm	0.5-1 μm	1-2.5 μm	2.5-5 μm	5-10 μm
	$\mu\text{g}/\text{m}^3$									
LIMIT VALUE	3	3	3	3	3	3	4	5	7	15
Bedroom 1	2.65	2.63	2.63	2.61	2.61	2.61	2.61	3.31	8.21	8.26
Bedroom 2	2.26	2.29	2.29	2.29	2.29	2.29	2.29	2.86	7.43	7.36
Bedroom 3	2.27	2.28	2.23	2.24	2.24	2.24	2.24	2.89	7.38	7.33
Living	1.11	1.11	1.13	1.13	1.13	1.13	1.13	1.45	3.40	3.41
Hall	2.10	2.10	2.09	2.09	2.09	2.09	2.09	2.54	6.93	6.89
Kitchen	6.99	3.11	3.17	3.17	3.17	3.17	2.09	4.07	9.30	9.33
Bathroom	2.11	2.10	2.07	2.08	2.08	2.08	3.17	2.59	7.02	7.05
Toilet	2.15	2.17	2.18	2.15	2.15	2.15	2.15	2.67	7.12	7.27
Service room	3.50	2.43	2.44	2.42	2.42	2.42	2.42	3.07	7.74	7.73

Table G7: Average annual PM concentration in each room with H13 filter, window opening and occupant.

Number of exceedance days H13 with window opening and cooking	0.01-0.1 μm	0.1-0.2 μm	0.2-0.25 μm	0.25-0.3 μm	0.3-0.4 μm	0.4-0.5 μm	0.5-1 μm	1-2.5 μm	2.5-5 μm	5-10 μm
	$\mu\text{g}/\text{m}^3$									
Bedroom 1	20	19	19	20	18	18	17	17	25	0
Bedroom 2	17	15	15	19	16	16	12	10	15	0
Bedroom 3	17	18	18	17	13	13	12	10	16	0
Living	3	1	1	0	0	0	0	0	0	0
Hall	12	13	14	14	12	12	12	7	17	0
Kitchen	88	35	39	37	31	31	12	23	35	0
Bathroom	13	15	14	13	11	11	27	8	17	0
Toilet	13	15	14	12	10	10	10	8	15	0
Service room	19	16	18	16	12	12	10	13	14	0

Table G8: Number of days exceeding the 24-hour limit imposed by the WHO with H13 filter, window opening and occupant.

Annex H : Case study 6

VISKON-AIRE[®] Air Filter Products

SuperCel 24 V Max HEPA Filter

Viskon-Aire[®] SuperCel 24 V Max HEPA filter is designed to withstand the toughest operating environments. The SuperCel 24 V HEPA filter is specifically designed for high flow applications that require HEPA filtration at the lowest possible pressure drop while maintaining their specified penetration efficiency. The V pleat formation has a penetration efficiency of 99.99% at 0.3µm and can withstand high air flows exceeding 600 FPM and a pressure drop of 1.0" w.g. while maintaining the integrity of the filter.

The Visikon-Aire SuperCel 24 V Max HEPA filter is manufactured with "Perform" wet laid micro fiberglass media. The SuperCel 24 V Max filter delivers HEPA efficiency for the life of the installation. The filter is designed with optimized pleat geometry and surface area to minimize energy usage and maximize particle loading, providing extended life and the lowest pressure drop possible. Each pack is potted within a non-corrosive plastic frame to eliminate bypass in the filter. The SuperCel 24 V Max HEPA filter incorporates recessed carry-all handles for ease of handling and installation.

The media area is optimized, and the pleat geometry allows for the lowest pressure drop possible, in some cases the SuperCel 24 V Max may have less media area than competitive HEPA filters, yet will still have an overall lower ΔP.

The Visikon-Aire[®] SuperCel 24 V Max HEPA is tested to ISO EN1822 Test Standard.

The Visikon-Aire[®] SuperCel 24 Filter Benefits:

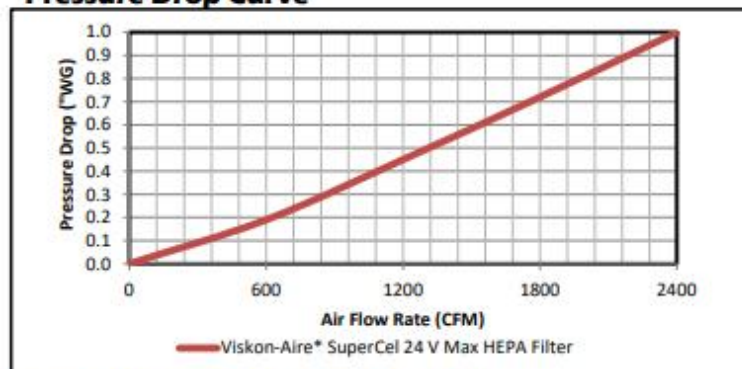
- Longer service life due to optimized pleat geometry and large surface area - 430 sq. ft.
- Low pressure drop (ΔP) at 600 FPM 1.00" w.g.
- Two component continuous polyurethane gasket
- All plastic construction with built-in handles
- Chemical resistant ABS plastic construction
- 100% RH resistance
- Maximum Temperature 175°F (79.4°C)



Performance	Results
Pressure Drop @ 2,400 CFM:	1.00" w.g. (249Pa)
Efficiency:	0.3µm 99.99% (H13)
Final Pressure Drop:	2.5" w.g. (622 Pa)
Media:	Perform high permeability glass
Media Area:	430 sq. ft (39.95)



Pressure Drop Curve



“When Clean Air is Critical”

VISKON-AIRE[®]
Air Filter Products

84 day Average concentration variable Coarse filter	0.01-0.1 μm	0.1-0.2 μm	0.2-0.25 μm	0.25-0.3 μm	0.3-0.4 μm	0.4-0.5 μm	0.5-1 μm	1-2.5 μm	2.5-5 μm	5-10 μm
	$\mu\text{g}/\text{m}^3$									
<i>LIMIT VALUE</i>	3	3	3	3	3	3	4	5	7	15
Bedroom 1	0.14	1.32	1.32	1.54	0.90	0.41	0.17	8.06	6.57	4.85
Bedroom 2	0.14	1.34	1.34	1.56	0.89	0.41	0.17	8.15	6.62	5.00
Bedroom 3	0.14	1.32	1.27	1.53	0.91	0.41	0.17	8.18	6.69	4.95
Living	0.14	1.31	1.26	1.50	0.90	0.41	0.17	8.11	6.46	4.88
Hall	0.14	1.34	1.27	1.55	0.92	0.41	0.17	8.14	6.75	5.07
Kitchen	0.14	1.34	1.31	1.59	0.92	0.42	0.17	8.39	6.97	5.25
Bathroom	0.14	1.32	1.27	1.55	0.90	0.41	0.18	8.18	6.64	4.79
Toilet	0.15	1.36	1.31	1.64	0.93	0.42	0.18	8.51	7.02	5.20
Service room	0.14	1.35	1.31	1.59	0.91	0.42	0.18	8.19	6.72	5.18

Table H1: Average annual PM concentration in each room with variable coarse filter during its lifetime.

238 day Average Variable H13 filter	0.01-0.1 μm	0.1-0.2 μm	0.2-0.25 μm	0.25-0.3 μm	0.3-0.4 μm	0.4-0.5 μm	0.5-1 μm	1-2.5 μm	2.5-5 μm	5-10 μm
	$\mu\text{g}/\text{m}^3$									
<i>TARGET VALUE</i>	3	3	3	3	3	3	4	5	7	15
Bedroom 1	0.06	0.54	0.54	0.65	0.32	0.14	0.07	3.34	3.97	3.18
Bedroom 2	0.06	0.48	0.48	0.57	0.29	0.13	0.07	2.82	3.18	2.59
Bedroom 3	0.06	0.49	0.46	0.57	0.29	0.13	0.07	2.92	3.23	2.54
Living	0.03	0.24	0.23	0.28	0.14	0.07	0.03	1.46	1.75	1.38
Hall	0.05	0.45	0.42	0.52	0.26	0.13	0.06	2.56	2.85	2.35
Kitchen	3.78	0.66	0.63	0.78	0.39	0.17	0.06	4.08	5.06	4.30
Bathroom	0.05	0.45	0.43	0.52	0.26	0.13	0.09	2.58	2.92	2.38
Toilet	0.05	0.46	0.43	0.54	0.27	0.13	0.07	2.65	3.07	2.50
Service room	1.10	0.53	0.49	0.61	0.31	0.14	0.07	3.07	3.68	2.96

Table H2: Average annual PM concentration in each room with variable H13 filter during its lifetime.

Number of exceedance days H13 with window opening and cooking	0.01-0.1 μm	0.1-0.2 μm	0.2-0.25 μm	0.25-0.3 μm	0.3-0.4 μm	0.4-0.5 μm	0.5-1 μm	1-2.5 μm	2.5-5 μm	5-10 μm
	$\mu\text{g}/\text{m}^3$									
Bedroom 1	0	0	0	0	0	0	0	16	4	0
Bedroom 2	0	0	0	0	0	0	0	10	1	0
Bedroom 3	0	0	0	0	0	0	0	9	3	0
Living	0	0	0	0	0	0	0	0	0	0
Hall	0	0	0	0	0	0	0	7	0	0
Kitchen	37	0	0	0	0	0	0	25	8	0
Bathroom	0	0	0	0	0	0	0	8	0	0
Toilet	0	0	0	0	0	0	0	9	0	0
Service room	0	0	0	0	0	0	0	11	3	0

Table H3: Number of days exceeding the 24-hour limit imposed by the WHO with variable H13 filter, window opening and occupant.