Health Impacts of Aviation UFP Emissions in Europe

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Content

Summary

Emissions of particulate matter (PM) by jet engines is increasingly recognised as one of the main impacts of aviation on human health and are estimated to cause approximately 14,000 and 21,200 premature deaths each year globally. Ultrafine particles (UFPs) are a subset of particulate matter (PM) emissions, typically defined as particles with a diameter of 100 nanometre or less. They have a proven relation with various forms of cancer, heart disease, COPD and other diseases of the respiratory system.

There are no comprehensive studies of the health impact of aviation UFP emissions in Europe. Extrapolating from the best studies available, this report provides a crude first-order estimation of what the health effects caused by aviation-related UFPs in Europe could be. The analysis estimates that aviation UFPs possibly may be the cause of a total of nearly 280,000 cases of high blood pressure, 330,000 cases of diabetes and 18,000 additional cases of dementia around the 32 major airports in Europe, based on current population and UFP concentration levels. However, these values are crude first-order estimates and should be confirmed by epidemiologic studies.

Aircraft UFP and PM emissions are mainly caused by the combustion of fuel, although lubrication oils also provide a significant contribution. The composition of the fuel determines the number of particulates emitted. In particular, the amount of emitted PM critically depends on the amount of aromatics (and other cyclic structures) in the fuel, and the sulphur content of the fuel. Aromatics are the main cause for formation of non-volatile PM (nvPM). Naphthalenes cause more UFP than single ring aromatics. The sulphur content is directly related to the formation of sulphuric acids, which in turn can both form sulphuric volatile PM (vPM) and attach to non-volatile particles.

This means that aviation PM emissions, and UFP emissions specifically, can be reduced by reducing the concentration of aromatics and sulphur in jet fuel. There are two ways to achieve this, namely through hydrotreatment of fossil fuels, thus saturating the aromatics and removing sulphur, and by increasing the use of sustainable aviation fuels, which are naturally low in sulphur and very often low in aromatics as well. To achieve these goals, several regulatory alternatives can be explored. In the EU, amending the Fuel Quality Directive or the ReFuel Aviation regulation could be possible pathways, and on a global scale amending existing standards or developing new ones could be appropriate solutions.

1 Introduction

Emissions of ultrafine particles by jet engines is increasingly recognised as one of the main impacts of aviation on human health. Ultrafine particles (UFPs), typically defined as particles with a diameter of 100 nanometres or less, have a proven relation with overall morbidity, various forms of cancer, heart disease, and COPD (Moreno-Ríos et al., 2022). Whereas road transport UFPs are typically concentrated on or near major roads, aviation UFPs are typically spread out over a larger area, including residential areas (Austin et al., 2021). Studies around Amsterdam Schiphol, one of Europe's largest airports, show effects of aviation UFP emissions on the risks for dementia, high blood pressure and diabetes, as well as indications of effects on a number of other diseases.

In addition, particulate emissions of aviation at cruise altitude contribute to global warming by serving as nuclei for the formation of ice particles, forming condensation trails and causing cirrus cloudiness (EASA et al., 2020).

The amount of UFPs emitted depends crucially on the composition of aviation fuel. Especially the concentration of naphthalenes, of aromatics in general, and the sulphur content determine the number and mass of emissions.

This report aims to summarise the scientific evidence on the health impact of aviation UFP emissions and how they can be addressed by changing fuel quality. In addition, based on this evidence, it derives a crude first-order estimate of potential health impacts of aviation UFP emissions on populations living in the vicinity of the 32 largest airports in Europe, and estimates how a change in fuel quality could improve their health.

The outline of the report is as follows:
- **Chapter 2** summarises the evidence of the relation between jet fuel composition and emissions of particulates.
- **Chapter 3** presents the evidence on the relation between particulate emissions and human health.
- **Chapter 4** builds a model to provide a crude first-order estimate of potential health impacts of aviation UFP emissions on populations living in the vicinity of the 32 largest airports in Europe and estimates how a change in fuel quality could improve their health.
- **Chapter 5** discusses policy measures to address the health impacts of aviation UFP emissions.
- **Chapter 6** presents the conclusions.

2 The link between jet fuel quality and particulate emissions

Particulate matter (PM) is the term for small particles, typically smaller than 10 micrometres in diameter. When suspended in the air (also known as aerosols), these particles can be inhaled and may cause serious health problems. PM emissions are generally characterised by size. In this report, we follow the scientific literature and consider three categories: PM_{10}, particulate matter that is smaller than 10 micrometres; $PM_{2.5}$, particulate matter smaller than 2.5 micrometres; and $PM_{0.1}$, particulate matter smaller than 0.1 micrometres, also referred to as ultrafine particles (UFP). These definitions imply that UFP is a subcategory of $PM_{2.5}$, which itself is a subcategory of PM_{10}.

In this chapter, we give a qualitative overview of PM emissions as a result of aircraft fuel combustion. Although we focus on UFP, all particulate matter is considered, as well as additional non-particulate emissions such as sulphur oxides and nitrogen oxides. First, we explain how fuel combustion in aircraft results in the (direct) emission of particulate matter, and the ways in which fuel emissions interact to create additional (secondary) particulate matter. Second, we consider how the chemical composition of fuels used for aviation relates to specific emissions of PM. In particular, we are interested in the determining conditions for black carbon emissions and sulphur oxides formation.

2.1 Characterisation of emissions from aircraft engines

Although aviation contributes only a small part of the global PM emissions, a significant share (14%) of aircraft emissions are during the relatively short landing and take-off (LTO) cycle[1] (Klimont et al., 2017). These LTO emissions are a significant source of ground-level air pollution in the region around airports (Owen et al., 2022). In this section, we will distinguish and categorise different types of PM and other relevant emissions by aircraft engines.

Table 1 - Emission factors of the most notable air pollutants in kilograms per landing & take-off cycle. These are reference values, independent of aircraft type and fuel composition. Only non-CO_2 emissions are listed.

Emission species	Abbreviation	Emission factor (kg/LTO)
Particulate matter, <10 μm	PM_{10}	0.54
Particulate matter, <2.5 μm	$PM_{2.5}$	0.53
Hydrocarbons	HC	2.68
Nitrogen oxides	NO_x	16.29
Carbon monoxides	CO	9.14
Volatile organic carbons	VOC	1.95
Black carbon	BC	0.26
Sulphur dioxide	SO_2	1.40

Data obtained from Bo et al. (2019).

[1] The LTO cycle is defined by the ICAO to include all aircraft activities below an altitude of 3,000 ft. This cycle is divided in the phases: approach, idle, take-off, and climb.

In addition to particulate emissions, aircraft engines emit multiple gases that affect both climate and air quality. These include sulphur oxides and volatile organic compounds. In Table 1, air pollutants and their respective mass emitted during the LTO cycle are listed.

2.1.1 Volatile and non-volatile PM

Particulate matter from aircraft engines has a large diversity in chemical composition. It is commonly divided into two main categories: non-volatile particulate matter (nvPM) and volatile particulate matter (vPM), see Figure 1.

Non-volatile particulate matter consists primarily of soot. These particles are often very small (0.01–0.1 micrometres) and mainly result from the incomplete combustion of aromatic hydrocarbons (Duong et al., 2018) (Bockhorn et al., 2009). A part of this soot is black carbon (BC). Although these terms are often used interchangeably, soot in general contains impurities, and consists of both organic and inorganic carbon. Black carbon on the other hand refers to pure elemental carbon.

Volatile particulate matter are liquid droplets, formed by condensed combustion exhaust gases. Formation of vPM is for the large part caused by emissions of sulphur-containing particles, lubrication oil and volatile organic compounds, including polycyclic aromatic hydrocarbons (PAHs), which are carcinogenic (Tait et al., 2022) (Zhang et al., 2022) (Heeb et al., 2024). While road transport fuels sold in Europe only contain a maximum of 10 sulphur parts per million (ppm), jet fuel typically contains from 10 to 1,000 ppm, with typical averages ranging between 300 and 600 ppm (Zschocke et al., 2017). The amount of sulphur in the fuel is directly related to vPM emissions and appears to be critical in UFP formation (Wayson et al., 2009) (Stacey, 2019).

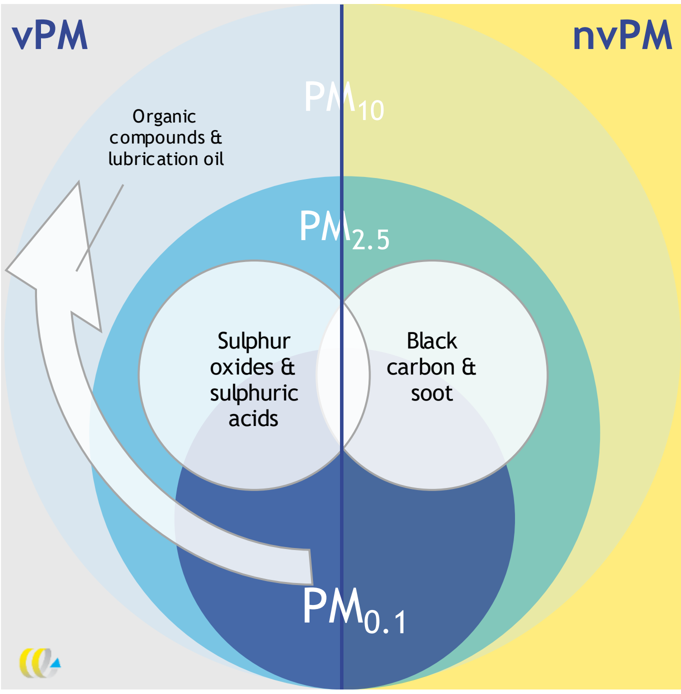
Near the exit nozzle plane of the combustor, only nvPM is emitted; the vPM forms by nucleation and condensation of the exhaust gases in the downstream, mostly sulphuric acids. Subsequently, vPM may attach to the nvPM, forming a coating around the soot particles. In particular, nvPM emitted from aircraft engines *at ground* is mainly soot which becomes coated with sulphuric acid and water. The diameter and composition of these particles change during the course of transport over the first hundreds of metres. However, their number remains more or less constant (Owen et al., 2022).

Black carbon particles as emitted by aviation mostly fall into the category of ultrafine particles. Their geometric mean diameter usually ranges from 15 to 60 nanometres (Owen et al., 2022). In contrast, vPM have typical diameters of a few nanometres after formation, but in their evolution in the aircraft exhaust plume they grow in size. Growth rates of particles are estimated in the order of 1 to 20 nanometres per hour (Kulmala et al., 2004).

The most straightforward measure for PM amounts is quantification of its mass. However, evidence accumulates that both climate and health effects are better predicted by the particle number (PN) of PM emissions instead (Zhang et al., 2022). With regards to health, smaller particles can penetrate deeper into the body. In addition, the total surface area of smaller particles is bigger than the surface area of bigger particles, assuming constant mass. As a consequence, small particles are able to carry relatively more toxicants. All in all, smaller (ultrafine) particles have an increasing detrimental effect on health, while their contribution to the total mass is relatively small due to their size. In other words, the relative quantity of most toxic particles in PM is hard to derive from the total mass alone, while absolute numbers of particles can provide this essential data. Ideally, both measures are combined and supplemented with a particles-per-size distribution.

Broad overviews and comparisons on UFP emissions are hampered by a lack of standardised methodology and robust techniques (Stacey, 2019). However, in 2019, the International Civil Aviation Organization (ICAO) agreed to the aircraft engine emission standards for nvPM to be measured both in mass and number.

Figure 1 - Qualitative overview of all types and classifications of particulate matter used in this report. PM_{10} includes $PM_{2.5}$, which itself includes UFP. Black carbon and soot mostly exist as UFP and $PM_{2.5}$. Sulphuric particles exist as vPM and condensates onto nvPM particles to form (larger) composite PM. Other vPM consists of organic compounds (VOCs) and lubrication oils, which grow over time by condensation. Areas do not represent quantity



Source: CE Delft.

2.2 Fuel composition dependency of PM emissions

In the previous section, we categorised the emissions by aircraft engines, where particulate matter emissions were distinguished from gaseous emissions. In this section, we will relate the emissions to the chemical composition of aircraft fuels. In line with the scope of this report, this is focused on particulate matter emissions.

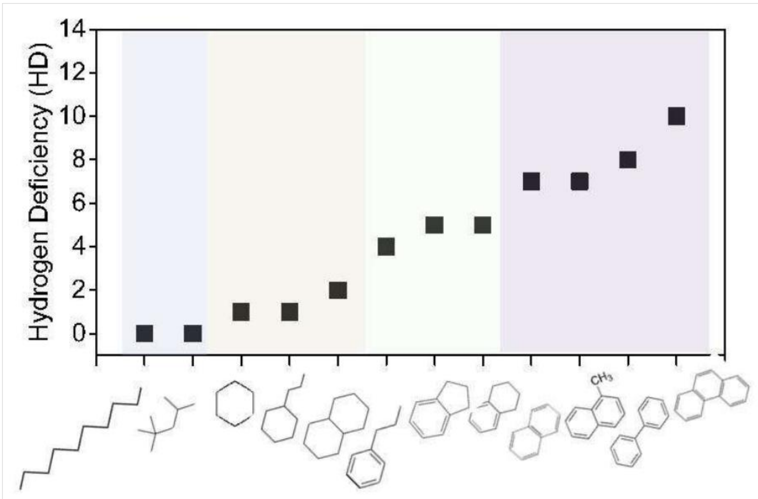
2.2.1 Black carbon

Many factors, such as engine type and combustion temperature, play an important role in the amounts and types of particulate matter emissions. Nevertheless, the chemical composition of the jet fuel used is one of the most important factors. Especially, the relative quantities of cyclic compounds in the fuel, e.g. aromatics and naphthalene, determine how much black carbon is emitted (Kathrotia & Riedel, 2020) (Zhang et al., 2022).

The average Hydrogen Deficiency (HD) number of a fuel in particular is shown to be closely related to BC emission. The HD number of a chemical component measures how unsaturated it is. In other words, this is an effective measure for how many double bonds and cyclic structures (rings) a molecule has. In Figure 2, the HD of prominent compounds of jet fuel is shown.

This relation quantifies the generally known fact that the sooting tendency of hydrocarbons increases with the amount of ring structures in the compounds (Glassman, 1989). In other words, the amount of naphthenes (cycloparaffins) and aromatics (compounds with benzene rings) is the dominant precursor of black carbon emissions. On the other hand, the amount of n-paraffins and isoparaffines (straight and branched hydrocarbons, respectively), which constitute the main part of jet fuel composition, is inversely related to soot emissions.

Figure 2 - The hydrogen deficiency (HD) number for some of the most occurring hydrocarbons in jet fuel. The HD, indicator for sooting tendency, is larger for molecules with more ring structures.



Adopted from Kathrotia and Riedel (2020).

As is depicted in Figure 3, the presence of aromatics stimulates soot formation. Although in the absence of aromatics soot can still form, this only occurs as a result of aromatic hydrocarbons that are formed due to the combustion process. In other words, conventional aromatic-rich fuels give the process of soot formation a head start (Kumal et al., 2020). Naturally, this leads to higher amounts of soot formation.

Figure 3 – Schematic overview of formation of soot in high- and low-aromatic fuels. Aromatic content in fuels allow the process a 'head start'. As a consequence, more particles are formed, and particles can grow larger.

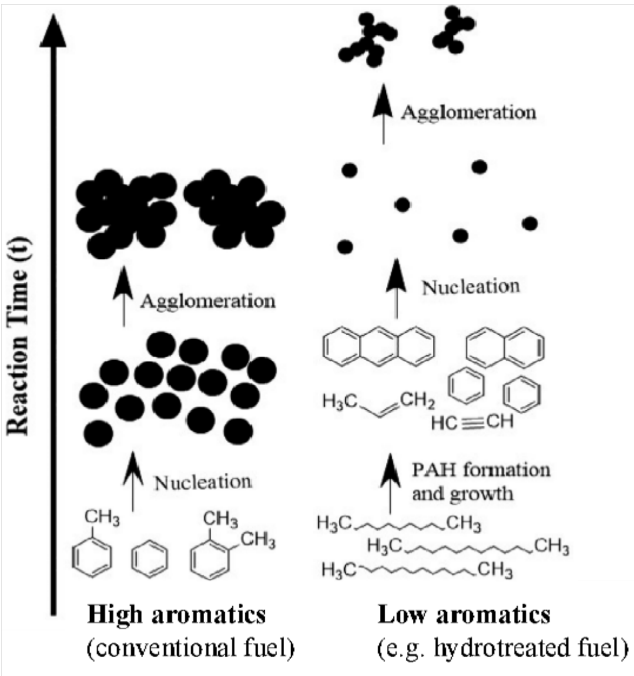


Figure adapted from Kumal et al. (2020).

2.2.2 Sulphur emissions

Sulphur is a naturally occurring component of crude oil, which is as a consequence also present in fuels, unless removed. The presence of sulphur is critical in the formation of ultrafine particles (Stacey, 2019). Sulphuric oxides form during the fuel combustion by oxidation of the sulphur that is contained in the fuel. The mass of sulphur oxides is directly proportional to the relative mass of sulphur in the jet fuel itself (Owen et al., 2022). Afterward, SO_3 and sulphuric acid (H_2SO_4) are created, which is concerning for both environment and human health (EASA, 2010).

In general, the presence of sulphuric acid promotes the nucleation process and increases the number of UFP (Kwon et al., 2020). More precisely, sulphuric acids have two distinct effects on PM formation. Firstly, sulphate particles can attach to solid particles, effectively coating nvPM soot with sulphur. Secondly, sulphate particles can condensate into vPM aerosols. The main difference between these types of particles is that the first is carbon-based, while the second is sulphur-based.

Which process is dominant depends on the ratio of nvPM to sulphate and organic particles in the exhaust plume. This in turn varies with power settings of the engine. At larger powers, more nvPM is generated and the first process is dominant. At lower powers, less nvPM is formed and sulphuric aerosols are formed, leading to a larger number of total particles (Timko et al., 2013). Some sources suggest that the total UFP (nvPM and vPM) from aviation mainly consists of condensed sulphur, as opposed to carbon-based particles (Gezondheidsraad, 2021).

Low-sulphur fuels were shown to have significantly lower UFP emissions (Beyersdorf et al., 2014). Besides reduction in SO_2, aviation fuels with reduced aromatics and close-to-zero sulphur content lowers nvPM mass and number emissions (Owen et al., 2022).
When reducing the sulphur content from 460 to 8 ppm (by mass), a 50% particle number reduction was measured, and particles where 30% bigger in size (EASA, 2010).[2] The number reduction significantly reduces PM emissions below 0.01 µm (i.e., 10 times smaller than the upper limit of $PM_{0.1}$).

2.2.3 Effect of fuel combustion on PM emissions

Although the chemical composition, number and size of PM emissions are in large part determined by the fuel used, other factors play a role, such as the chemical and physical parameters during the fuel combustion process and the type of combustion engine.
The power settings of the engine do not only have effect on sulphur emissions as explained above, but also on particulate matter in general.

The geometric mean diameter (GMD) of emitted particles is known to increase with thrust levels of the engine, from ~8 nm at idle to ~40 nm at take-off (Durdina et al., 2021). Measurements on an APU showed that increasing thrust corresponds to increasing amount of BC mass, while nvPM number peaked at 65% relative thrust (equivalent to cruise) (B.T. Brem et al., 2015). This is likely due to the coagulation of particles at higher thrust levels.
In general, reduction in nvPM (by number) by burning alternative aviation fuels is most pronounced at low engine thrust (idle phase, for example) (Zhang et al., 2022).

PM emissions also depend on the type of engine used, although quantitative comparisons are hard to make due to their technical difference and the difference in assessment methods. Nevertheless, some trends can be observed, especially for emission indices (i.e., amount of emissions per unit of burnt fuel). For turbofan-type engines maximum nvPM number emission indices are measured at lower thrust settings, such as in approach and idle phase. For turboshaft engines, this trend was similar for small engines. However, for larger engines, higher PM mass and number indices were measured with increasing power. Turboprop engines showed a decrease in PM number emissions in general, except for the increase in power between low- and high-speed ground idle (Owen et al., 2022).

2.3 Reduction of PM emissions by hydrotreatment of fuels

Hydrotreatment of petroleum fuels is a way to saturate aromatics and remove sulphur. In short, the process implies that crude oil cuts are selectively reacted with hydrogen in the presence of a catalyst. Hydrotreatment processes achieve first the removal of contaminants, such as sulphur, followed by the transformation of olefins and aromatics into stable paraffinic hydrocarbons. As a consequence, the energy density of fuels is increased and the production of harmful combustion by-products such as PM and SO_x is reduced (Ortega, 2021). Hydrotreated fossil fuels suppress both the mass and number of nvPM emitted from the turbine (Okai et al., 2019) (Schripp et al., 2022).

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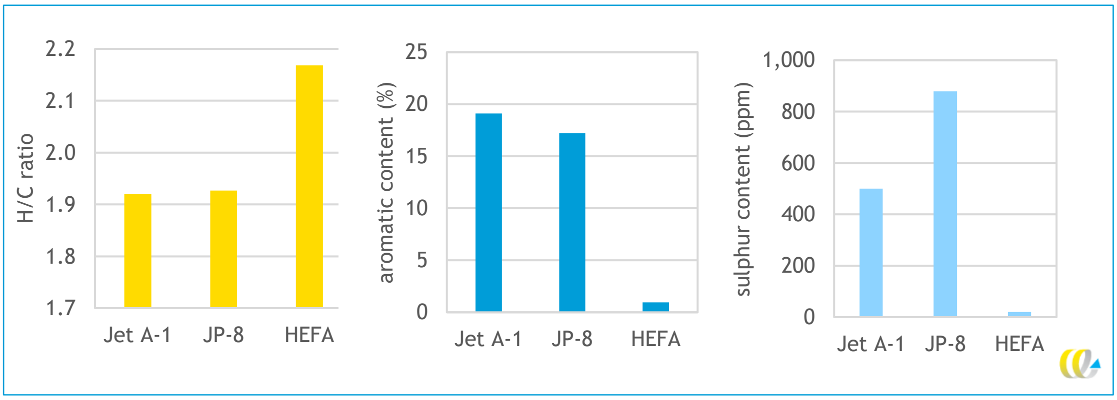
[2] This was accompanied by a mass increase. The authors suggest this can be a result of wrong assumptions in the calculation, as mass is a parameter derived from number measurements in this study. Furthermore, smaller particles that are not measurable for high-sulphur fuels may grow bigger for low-sulphur fuels. As such, they fall within measurement range and give an additional contribution to the total mass.

This process can be used both to produce fuels from biological sources, such as vegetable oil, as well as refinement of fossil fuels. Especially the latter, which is already applied in the production of petrol, diesel, and shipping fuels, is considered to be a short-term solution to reduce PM emissions from aviation. Hydrotreatment of jet fuels to reduce aromatics is possible in normal oil refineries (Kittel et al., 2022; Gunnar Quante et al., 2024).

2.3.1 Chemical composition of hydrotreated fuels

Aromatics and sulphur in jet fuel are dominant sources of PM formation in combustion. Hydrotreatment processes are capable of reducing or removing aromatics and sulphur. Since the hydrotreatment of fossil fuels is not applied to a wide level yet, little data is available on the characteristics of PM emissions of hydrotreated fossil jet fuels. Although not all sustainable aviation fuel (SAF) types contain reduced aromatics or sulphur, some SAFs such as HEFA do contain very low levels of those compounds. For that reason, HEFA SAFs, or HEFA SAF and fossil jet fuel blends may have comparable characteristics to hydrotreated fossil jet fuels with respect to sulphur content, aromatics content, and naphthalene content.[3] The hydrotreatment process used to produce HEFA SAF is also similar to the hydrotreatment of fossil fuels. Therefore, data on HEFA and blends of HEFA SAF and fossil jet fuels may be used to estimate the effects of hydrotreatment of fossil jet fuels.

Figure 4 – Comparison of common H/C ratios, aromatic and sulphur contents in Jet A-1 and JP-8 fuel, compared to hydroprocessed (HEFA) fuels



Data obtained from Zhang et al. (2022).

In Figure 4, typical values for the H/C ratio, the aromatic content, and the sulphur content are presented for Jet A-1 fuel (the most common type of fuel for commercial flights), JP-8 (common US military jet fuel) and HEFA (hydrotreated) SAF. These quantities are indicators of the quality of the fuel and how much PM is produced when they are combusted.

[3] At the time of writing, ASTM had approved seven SAF production pathways to be blended with fossil jet fuel up to different limits. Of these SAFs, one contains aromatics (Fischer-Tropsch synthesised isoparaffinic kerosene containing aromatics, or FT-SPK/A). The other approved pathways are producing blends of paraffinic and isoparaffinic kerosene (IATA, n.d.). Production pathways for other aromatic containing SAFs are under development, such as Alcohol to Jet Synthetic Kerosene with Aromatics (ATJ-SKA) and Synthesised Aromatic Kerosene (SAK) (University of Illinois at Urbana-Champaign, 2023).

The H/C ratio indicates how much hydrogen the fuel holds relative to the amount of carbon. As is explained in Section 2.2.1, a higher H/C ratio correlates to less cycloparaffins and aromatics. The H/C ratio of hydrotreated fuels in Figure 4 is 13% higher than for Jet A-1 and JP-8, on average.

The sulphur content is expressed as part per million (ppm). From the data, it is evident that hydrotreatment indeed removes a significant portion of the sulphur content in the fuel. The amount of sulphur particles is typically less than 20 ppm, which is about 3-6% of the sulphur found in non-hydrotreated fossil fuels. The low sulphur content of HEFA SAFs is consistent with that of hydrotreated fossil jet fuels analysed in G. Quante et al. (2024), whereas the H/C ratios of HEFA SAFs are slightly higher.

The aromatic content, closely linked to the sooting tendency of the fuel, ranges between 0 and 2% in the measured HEFA fuels. Compared to Jet A-1 and JP-8, with an average aromatic content of 17 and 19% respectively, this is a decrease of more than 95%.

2.3.2 Emissions of hydrotreated fuels

The difference in chemical composition of jet fuels evidently leads to a difference in emitted PM. As explained in Section 2.3.1, SAF HEFA fuels or HEFA SAF and fossil jet fuel blends may have comparable compositions to hydrotreated fossil jet fuels, so in this subsection the study uses test results from SAF HEFA fuels and blends as a proxy for hydrotreated jet fuel emissions.

As a consequence, we estimate that the use of HEFA SAF fuels can reduce the nvPM number emission from 10^{15} - 10^{17} down to 10^{14} - 10^{15} particles per kilogram fuel, measured at ground level (Zhang et al., 2022). In terms of mass, nvPM measurements for conventional jet fuels range up to 400 milligrams per kilogram fuel, while alternative aviation fuels typically produce less than 100 milligrams nvPM per kilogram fuel. This is due to the fact that for alternative fuels, such as hydrotreated fuel, generally 90% of their weight consists of n- and isoparaffins (i.e., hydrocarbons without cyclic structure). Less than 10% of the fuel is comprised of cycloparaffins, and typically these fuels do not contain any aromatics and sulphur.

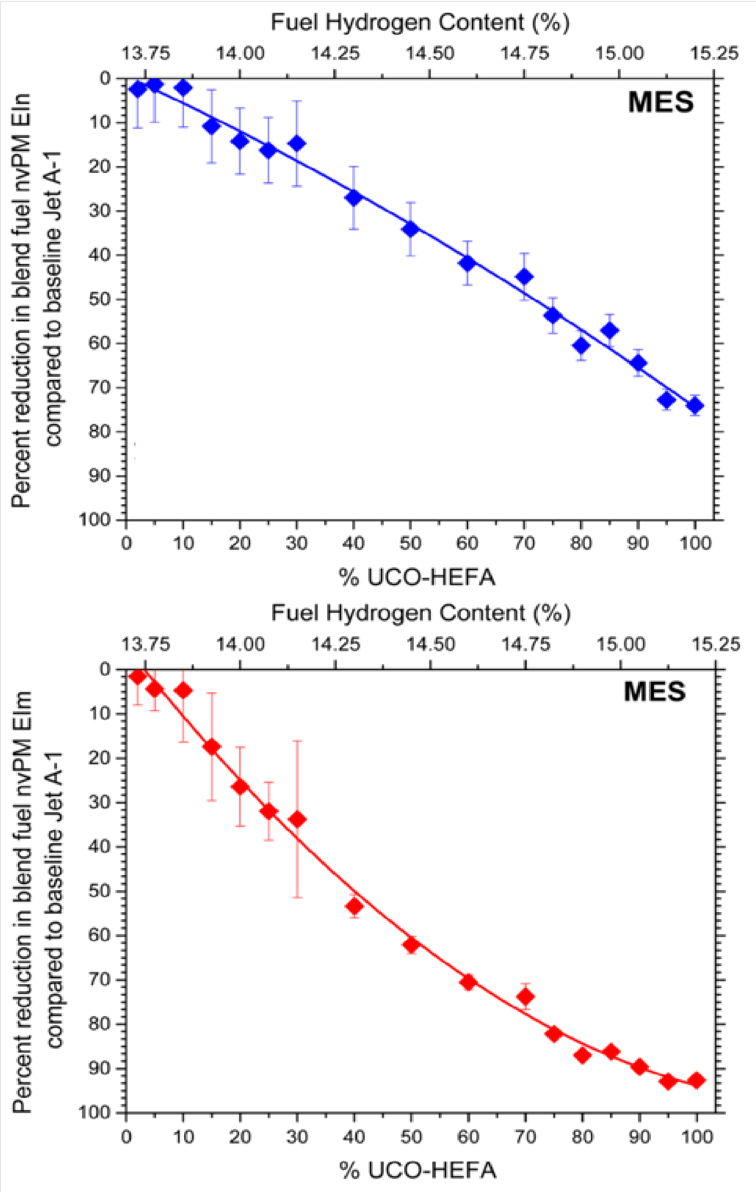
In general, the relative amount of SAF HEFA fuel in a blend with conventional jet fuel is directly linked to both nvPM emissions in number and in mass (Lobo et al., 2015).
At present, jet fuel standards allow alternative fuels such as SAF only as a blend with conventional fuel, with a maximum of 50%, depending on the type of SAF. According to Figure 5, the reduction of nvPM particle numbers in these blends is around 30%, equivalent to 60% mass. Up to ~75% of the number nvPM emissions (~90% mass) can be reduced when using 100% SAF HEFA fuel. As indicated in Figure 6, the emissions of particulates of all sizes are reduced, but more so for large particles than for smaller particles.

An increasing hydrogen content (corresponding to more hydrotreated or SAF HEFA fuel compared to conventional fuel) reduces emissions of the biggest particle sizes relatively more than the smaller particle sizes. The geometric mean particle size (that is, the peak Figure 6) is ~50 nanometres for 100% Jet A-1 fuel. Burning 100% HEFA SAF hydrotreated fuel reduces this to ~25 nanometres. Moreover, the peak number also reduces by more than half, from 8×10^{15} to roughly 3×10^{15} emitted particles per kg fuel burned. Despite some quantitative differences, the results are qualitatively similar for all engine operating conditions.

The results in Figure 5 and Figure 6 were obtained in tests with an auxiliary power unit in MES (main engine start) operating condition. At least for 100% Jet A-1, these results closely resemble the nvPM emissions of an aircraft engine. Furthermore, ground-based measurements on jet engines showed similar results in the reduction of nvPM emissions when using HEFA or other fuels with a higher H/C ratio. For example, measurements on an A320 showed similar nvPM number distributions, where most particles are in the range of 10-100 nm, with a maximum in the range of 30-50 nm (Schripp et al., 2022). An earlier experiment on an A320 found that particle number emissions decreased up to 50% at low power settings when using synthetic jet fuel (Schripp et al., 2018). They concluded that the H/C ratio is the best indicator for both sooting tendency and particle emissions. Measurements on a turbofan engine showed a similar relation between on the one hand the aromatic concentration (or hydrogen content), and on the other hand nvPM and BC emissions (B.T. Brem et al., 2015). The relation held for both mass and particle number measurements. Another experiment at a turbofan jet engine compared the particle number distribution of Jet A-1 with that of a 32% HEFA blend (Durdina et al., 2021). They concluded that for a standardised Landing- and Take-Off cycle, the HEFA blend reduced the total nvPM number by 25%. At idle, nvPM emissions were even reduced by 70%.

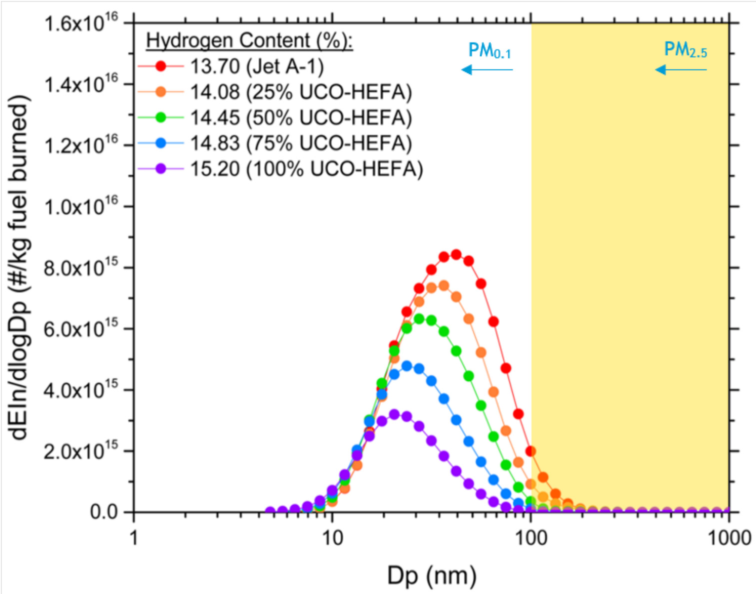
In Figure 7, the nvPM number emissions as function of particle size are presented for both conventional Jet A-1 fuel and a blend of Jet A-1 with HEFA-SPK (70 vol%/30 vol% respectively), burned in an Airbus A320 with V2527-A5 engines at ground (Schripp et al., 2022). The aromatic content in the blend was 9.5 vol%, compared to 18.6 vol% in Jet A-1. The naphthalene content was reduced from 1.17 vol% to 0.05 vol% and sulphur content from 105 ppm to less than 5 ppm. The fuel hydrogen content was 13.6% for the Jet A-1 fuel and 14.5% for the HEFA-SPK blend. At intermediate power (1,270 kg/h), the number of nvPM emissions was roughly halved, corresponding to a mass reduction of 70%. Further, the results show that for different fuel flows and different power settings, the nvPM emissions differ, although the blend always leads to less total emissions.

Figure 5 - The reduction in nvPM emissions in number (above) and mass (below), as a function of the relative amount of hydrotreated fuel blended with Jet A-1 (or equivalently, as function of the fuel hydrogen content)



Original figures from Lobo et al. (2015).

Figure 6 - Typical PNC distributions for Jet A-1 fuel blended with 0, 25,50,75 and 100% hydrotreated fuel respectively. The particle number index *EI_n* is plotted logarithmically as function of particle size *<u>Dp</u>*. With increasing hydrogen content, almost all $PM_{2.5}$ and a significant portion of $PM_{0.1}$ emissions are suppressed.



Original figure from Lobo et al. (2015).

Figure 7 - Number nvPM measurements on Jet A-1 (Ref3 in figure) and on HEFA-SPK blend (SAJF2 in figure). The left figure shows results at low power, while the right figure shows results at high power. The different colours indicate different fuel flows (FF).

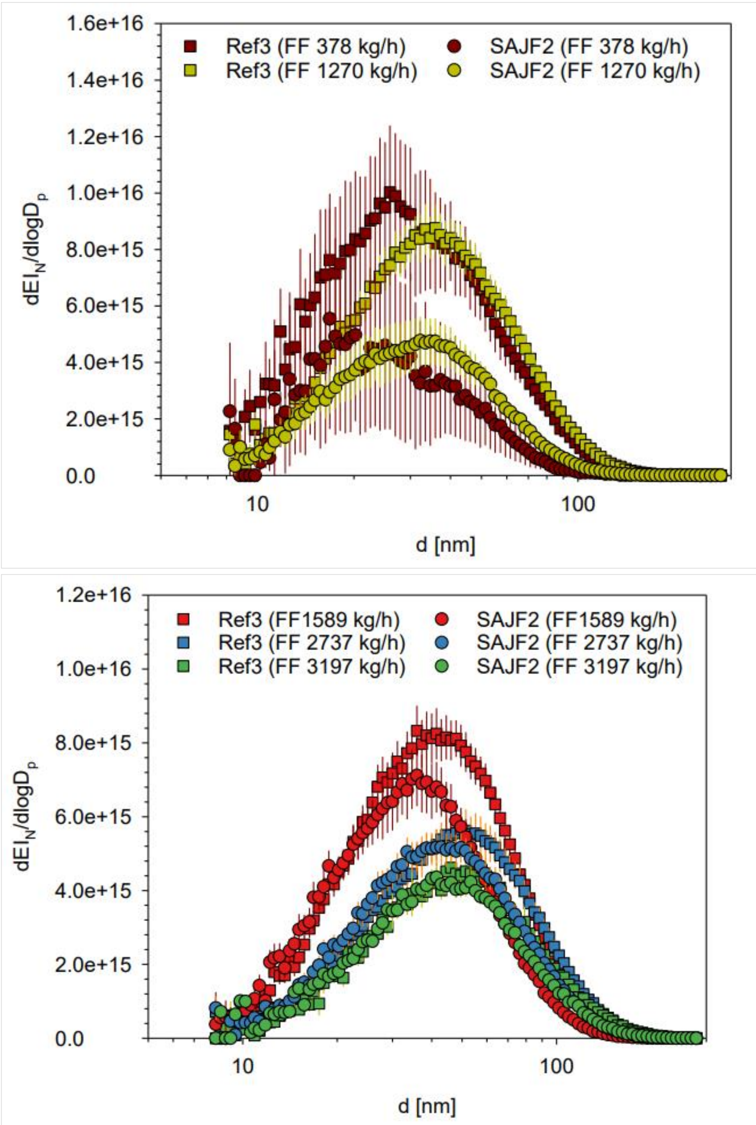


Figure adopted from Schripp et al. (2022).

Complementary to these results, B.T. Brem et al. (2015) found a linear relation between aromatic content in the fuel on the one hand, and the black carbon mass and the total nvPM particle mass on the other hand. Furthermore, it was found that naphthalene, a specific polyaromatic compound, leads to more BC and nvPM emissions than other aromatics: the amount of emitted particles increased when to total volume of aromatics was kept equal, but the relative share of naphthalene in that volume was increased. This suggests the even more noxious nature of naphthalene, compared to other aromatics. In short, reducing the aromatics by a certain volume percentage resulted in a linearly related reduction in black carbon and nvPM, and reducing the naphthalene content reduced BC and nvPM to a greater extent.

This phenomenon is illustrated in the following figure, where for different engine thrusts two aromatic solvents were injected in Jet A-1 fuel, one of which contains naphthalene (Solvesso 150; 6 vol%) and one does not (Solvesso 150ND) (Benjamin T. Brem et al., 2015). Especially at low thrust levels, the absence of naphthalene leads to significantly lower BC mass and nvPM number emissions, while the total aromatics concentration stays constant.

Figure 8 - Measurements of BC mass and nvPM number in the emissions of burning Jet A-1 fuel injected with aromatic solvent with naphthalene (Solvesso 150) and without naphthalene (Solvesso 150ND).

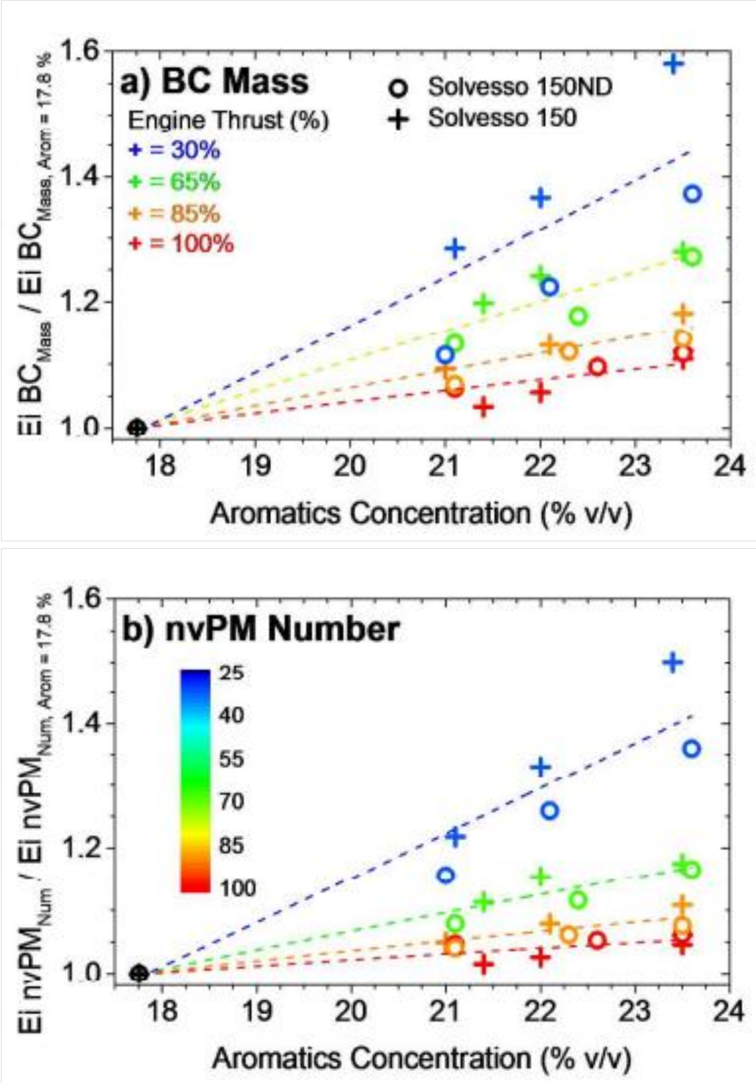


Figure adopted from Benjamin T. Brem et al. (2015).

Furthermore, the lowered sulphur content of hydrotreated fuel leads to a decrease in the formation of sulphate vPM and naturally decreases the sulphuric coating of nvPM.
In particular, the UFP emissions were shown to be drastically lower for low-sulphur fuels (Beyersdorf et al., 2014). As sulphuric emissions are directly related to the sulphur content of the fuel, Figure 4 suggests hydrotreated fuels can reduce these emissions almost entirely. Hence, low-sulphur or sulphur-free fuel will lead to lower vPM from aircraft engines (Owen et al., 2022).

In particular, the number of particles below 10 nm (0.01 µm) are reduced when the sulphur content is decreased to ~8 ppm (m) (EASA, 2010). The HEFA fuel used in Lobo et al. (2015) (the results of which are presented in Figure 5 and Figure 6) consisted of <180 ppm (m). Consequently, fuels with even less sulphur content may also reduce the number of smaller sized particles.

Since the first effects of hydrotreatment processes on jet fuel composition are reduced sulphur and naphthalene content (G. Quante et al., 2024), the associated benefits of reduced vPM and nvPM emissions may be obtained even for jet fuels produced with mild hydrotreatment processes.

In conclusion, we find that hydrotreated fuels emit less particulate matter and black carbon, mainly due to their lowered aromatic and sulphur content. This is true for all thrust settings, but it is especially significant for low thrust.

2.4 Conclusions

In this chapter, the relation between jet fuel composition and combustion emissions is specified. Aircraft engine combustion results in the emission of many types of particles. Focusing on particulate matter, this report uses the classifications PM_{10}, $PM_{2.5}$, and $PM_{0.1}$ (UFP). Furthermore, we discriminate black carbon in particular. Non-particulate emissions that are determined to be relevant to our discussion are sulphur dioxides and, to a lesser extent, VOC and nitrous oxides.

We have shown that the amount of emitted PM critically depends on the amount of aromatics (and all cyclic structures) in the fuel, and the sulphur content of the fuel. Aromatics are the main cause for formation of nvPM. The amount of aromatics can be quantified by the hydrogen deficiency number, which is proportional to BC emissions. The sulphur content is directly related to the formation of sulphuric acids, which in turn can both form sulphuric vPM and attach to non-volatile particles.

Lastly, hydrotreatment is discussed as a method to decrease the aromatic and sulphur content of the fuel, and hence to decrease the PM and sulphur emissions of aircraft engines. A 50/50 blend of SAF HEFA fuel with conventional jet fuel results in a PM number reduction of roughly 30%, equivalent to 60% mass. Up to 60% of the number PM emissions (90% mass) is reduced when 100% HEFA SAF hydrotreated fuel is combusted in aircraft engines.

3 The link between aviation particulate emissions and health

Air pollution is usually associated with short-term symptoms upon exposure, like coughing, tearing and difficulty in breathing. The public is mostly unaware of long-term effects on health, the effect of worsening already existing medical problems, and permanent damage to organs (Schraufnagel et al., 2019). However, the World Health Organization has declared that air pollution may be the greatest environmental risk to health in the world (WHO, 2016). Here, air pollution is taken to mean all substances in the air that harm humans, animals, vegetations, or materials. The most notable of these include particulate matter, sulphur and nitrogen oxides, volatile organic compounds, and carbon monoxide. All of these are produced by aircrafts, mostly due to the fuel combustion in jet engines.

In this section, we list the health impacts of aviation emissions. First, we give a broad overview of measured health impacts of aviation emissions in general. In the remaining subsections, we discuss the scientific evidence for health impacts attributed to specific components, including PM_{10}, $PM_{2.5}$, UFP, black carbon, and sulphur oxides. To the extent classifications and types of particulate matter overlap (see Figure 7), health effects of these categories also overlap. For example, health effects of UFP are in principle included in the health effects of $PM_{2.5}$, although UFP may cause specific or more severe health effects than $PM_{2.5}$ in general. In the same way, black carbon is the cause of some (but not all) health effects of $PM_{2.5}$.

3.1 General health impact of aviation emissions

Focusing on air pollution caused by aviation specifically, Yim et al. (2015) estimates 14,000 early deaths globally each year due to long-term exposure to $PM_{2.5}$ emissions caused by global aviation. These estimations are based on mass quantification of emissions, whereas Eastham et al. (2023) estimates 21,200 early deaths due to particulate matter exposure. The societal cost of the premature deaths by $PM_{2.5}$ and O_3 (ozone) emissions, which is estimated at roughly 20 billion US dollars, is in the same order as the societal costs due to climate change caused by aviation, according to Yim et al. (2015).

Aircraft emissions are associated with increased hospitalisation for asthma, respiratory, and heart conditions. This is especially true for susceptible subgroups such as children below the age of five, or elderly above 65 years and groups with lower socioeconomic status (Bendtsen et al., 2021). Airport personnel working at the apron (close to running jet engines) are exposed to the highest levels of emissions.

Since children breathe more air relative to their weight, they are more harmed by air pollution. Furthermore, postnatal exposures to air pollutants, including PM, O_3, and NO_2, have been associated with increased infant mortality, also in developed countries such as the United States. Air pollution has also been found to affect growth trajectories of the lung and its function during childhood, which can affect the level of respiratory health achieved in adulthood (Schraufnagel et al., 2019). Association exists between childhood leukaemia and airport emissions (Riley et al., 2021).

The National Institute for Public Health and the Environment of the Netherlands (RIVM) has investigated the short-term and long-term effects of higher exposure to ultrafine particles among residents around Schiphol Airport, Amsterdam (Janssen, Hoekstra, et al., 2022). They conclude that long-term exposure may have an effect on the cardiovascular system. A statistically significant causal relation between UFP exposure and respiratory diseases was not found. However, short-term exposure can temporarily aggravate existing respiratory diseases. In particular, children were found to suffer more from respiratory symptoms on days with high UFP concentrations. In another study, it was found that asthma patients experienced increased acute systemic inflammation and oxidative stress due to aviation-caused UFP (Habre et al., 2018).

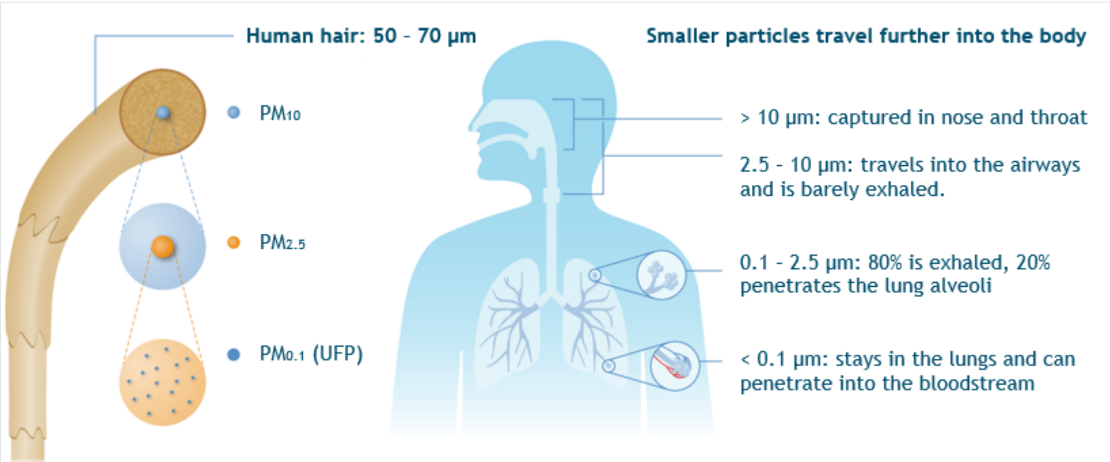
Furthermore, Janssen, Hoekstra, et al. (2022) concluded that exposure of pregnant women to ultrafine particles may possibly have a detrimental effect on the development of unborn children, although no causal relation was established. Similarly, UFP emissions from aircraft engines were associated with increased rates of pre-term birth rates in down-wind regions near the LAX airport (Wing et al., 2020).

3.1.1 Airport employees

A Danish research group investigated the exposure to UFP of airport employees (Møller et al., 2017; Møller et al., 2014). This investigation was started after three airport workers were diagnosed with bladder cancer, and others suffered from lung cancer, cardiovascular diseases, clots and chronic obstructive lung disease (Andersen, 2012). The researchers found that baggage handlers were the most exposed to UFP (37,000 particles per cubic centimetre on average), which was 7 times higher than employees working indoors. Catering drivers, cleaning staff and airside security were exposed to intermediate concentrations of 12,000-20,000 particles per cubic centimetre. Other research had already shown that respiratory symptoms are prevalent among airport employees exposed to jet stream exhaust (Yang et al., 2003) (Tunnicliffe et al., 1999).

A review paper from 2021 concludes that the mentioned papers are also the only available literature on the subject of UFP exposure to airport workers (Merzenich et al., 2021). Given that UFP concentrations are the highest at the airport, exposure to airport employees presents an unquantified but probable and serious health risk.

Figure 9 - Size comparison and health effects of particulate matter, depending on particle size



Adapted from Gezondheidsraad (2018). All rights belong to Joris Fiselier infographics.

Figure 10 - The fraction of deposited particles in the human respiratory system, as function of particle size

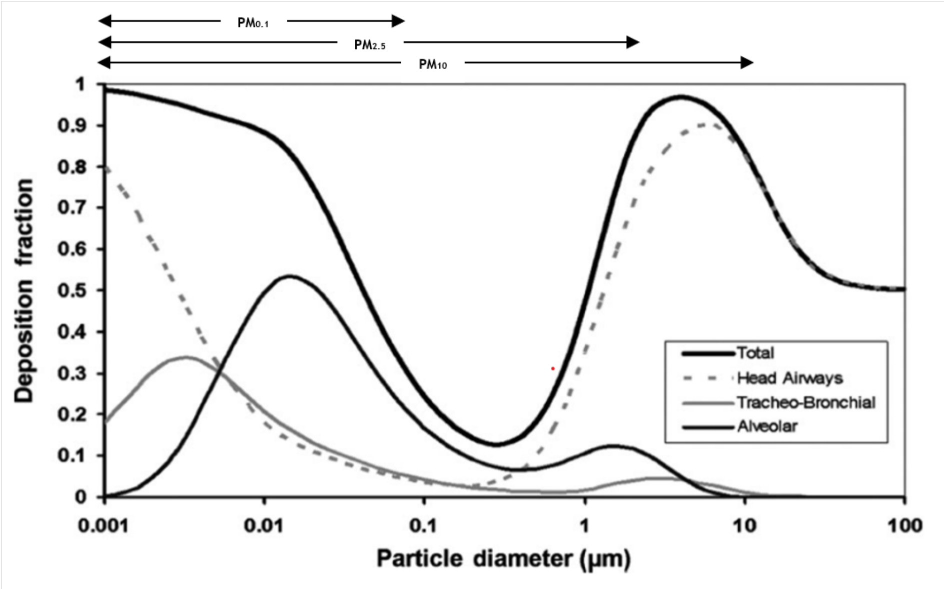


Figure adapted from Bergmans et al. (2022).

3.2 Health impact of PM_{10} and $PM_{2.5}$

Outdoor fine particulate matter exposure is the fifth leading risk factor for death in the world. It accounts for 4.2 million deaths and more than 103 million disability-adjusted life years lost (Murray et al., 2020).

Fine particulate matter ($PM_{2.5}$) due to aviation specifically is estimated to cause around 14,000 early deaths per year worldwide. In Europe, half of these early deaths caused by aviation are due to LTO emissions (Yim et al., 2015).

Fine particulate matter pollution affects the human body through tissue damage, which may result directly from pollutant toxicity because fine and ultrafine particles can gain access to organs, or indirectly through systemic inflammatory processes (Schraufnagel et al., 2019).

Inhaled particles of 5–10 micrometres usually land on the airways, as also shown in the figure above, and are normally removed by alveolar macrophages and lung lymphatics (Schraufnagel et al., 2019). Particles in the range of 1–2.5 micrometres on the other hand make their way to the terminal bronchiole, the site of greatest accumulation and tissue destruction, as commonly seen in centrilobular emphysema, a form of COPD (Schraufnagel, 2020). Complimentarily, Pope and Dockery (2006) concludes that long-term exposure to PM results in a more rapid progression of COPD.

There is a close, quantitative relationship between exposure to high concentrations of small particulates (PM_{10} and $PM_{2.5}$) and increased mortality or morbidity, both daily and over time. People with heart or lung diseases, children and older adults are most likely to be affected (EPA, 2020). When concentrations of small and fine particulates are reduced, related mortality will also go down – presuming other factors remain the same (WHO, 2021).

A review of the available scientific literature concluded that PM health effects are dependent on both exposure concentrations and length of exposure (Pope & Dockery, 2006). Short-term studies only capture a small amount of the overall health effects of PM exposure. Long-term repeated exposures have larger, more persistent cumulative effects than short-term transient exposures. Effects include lung and cardiovascular diseases.

Furthermore, the toxicity of PM is determined by its specific composition. Highly acidic particles, such as sulphuric acids, are more noxious. Other toxic elements such aromatics can also attach to particles during the combustion process. Consequently, the particulate matter takes these toxins into the lungs. This effect is enlarged for smaller particles (Schraufnagel et al., 2019).

Studies found that sleep efficiency decreases in areas with more air pollution, where increased exposure to PM is especially linked (Fang et al., 2015). PM_{10} exposure is associated with increased ischemic heart diseases among the elderly population and with higher risk of myocardial infarction (Rai, 2015).

3.3 Health impact of UFP emissions

Ultrafine particles are believed to be more toxic than $PM_{2.5}$ in general (CE Delft, 2023). As determining effects for UFP independent of bigger particles within $PM_{2.5}$ is very difficult, much health outcomes evidence for independent effects of UFP are inconclusive or insufficient. Current studies are often not adequately able to discriminate between the effects of UFP and various other pollutants, due to a lack of established methods and good models. A notable exception is the RIVM study, as mentioned in Section 3.1. In the remainder of this subsection, further proven and probable health effects and physio-chemical consequences of UFP are discussed.

Besides the physical and chemical characteristics of the type of matter, the toxic effect of particulate matter is in large part determined by the total surface area. The total surface area of a given mass strongly increases when the average particle diameter becomes smaller. UFPs are so much smaller in size, that the mass of one $PM_{2.5}$ particle is equal to the mass of 2 million UFPs of 20 nm in diameter (Oberdörster et al., 1995). Hence, when comparing equal masses, PM becomes increasingly more toxic when they get smaller in size (Schraufnagel, 2020). This is true for PM_{10} and $PM_{2.5}$, but it becomes especially of vital importance for ultrafine particles. Therefore, mass measurements may not be useful for UFP, and number concentrations should be used to better estimate health effects.

UFPs affect human physiology in a first instance by inhalation and absorption of the small particles into the lungs. Possible effects of UFP on human lung cells include asthma, COPD, pulmonary fibrosis and lung cancer (Moreno-Ríos et al., 2022). These results come from toxicological studies based on the reaction of isolated cells. However, it should be noted that Janssen, Houthuijs, et al. (2022) did not find empirical evidence for the relationship between UFP exposure and lung cancer.

Furthermore, $PM_{0.1}$ stay airborne longer and easily gain access to alveoli. Although most PM sizes can be engulfed by cells, $PM_{0.1}$ can travel across alveolar epithelial cells by diffusion through the lipid bilayer of the cell walls (Yacobi et al., 2010). In this way, particles can get into the bloodstream, and ultrafine particles may even use this mechanism to translocate to the brain and through the olfactory nerve (Calderón-Garcidueñas & Ayala, 2022) (Li et al., 2022). As a consequence, UFPs may be linked to neurodegenerative diseases and neurological disorders (Heusinkveld et al., 2016).

Ultrafine particles generate oxidative stress through inflammation of the lungs. The pro-inflammatory signals which are consequently set off may affect distant organs (Schraufnagel et al., 2019). The effects of this mechanism increase with smaller particle sizes. When ultrafine particles travel into other organs by the above-mentioned mechanism, they can be responsible for inflammations in those organs.

In a lab study, bronchial epithelial cells were exposed to UFP collected from Schiphol Airport (Amsterdam). Particle doses from 0.09 to 2.07 $\mu g/cm^2$ were tested and shown to induce cell damage and release pro-inflammatory markers (He et al., 2020). Significant evidence exists for short-term associations with cardiovascular and inflammatory changes (Ohlwein et al., 2019).

3.4 Health impact of black carbon

The main source of black carbon (BC) is combustion engines and fires. Although health effects due to BC exposure are usually associated with effects due to PM in general, the World Health Organization suggested that black carbon is a better indicator of harmful particulate substances from combustion sources, at least on short-term exposure (WHO, 2012). Black carbon was a better indicator for morbidity than PM concentrations. Studies of short-term health effects show that the associations with BC are more robust than those with $PM_{2.5}$ or PM_{10}. This suggests that BC is a better indicator of harmful particulate substances from combustion sources than undifferentiated PM mass (Janssen et al., 2012). Moreover, both the long- and short-term effects due to BC are estimated much higher than effects due to PM_{10} or $PM_{2.5}$.

There is no scientific consensus if black carbon itself is a main toxicant, or whether the toxic effects are primarily due to the adsorption of toxic substances which BC can easily translocate through the entire body, as described in Section 3.3. Regardless, black carbon concentrations are associated with lung cancer, heart attacks, low birth rates, asthma, and other respiratory diseases (EPA, 2013).

3.5 Health impact of sulphur oxides

The group of sulphur oxides mainly consists of sulphur dioxide (SO_2), which is emitted by burning fossil fuels. It is an important component for PM formation leading to similar diseases as emissions of $PM_{2.5}$. Short-term exposures to SO_2 can furthermore irritate the human respiratory system and make breathing difficult. People with asthma, particularly children, are sensitive to these effects of SO_2. Hospital admissions for cardiac disease and mortality increase on days with higher SO_2 levels. This air pollutant can also react with other compounds in the atmosphere to form PM pollution (EPA, 2021).
Orellano et al. (2021) shows that a rise in SO_2 concentrations increases the risk of all-cause and respiratory mortality in humans. Furthermore, SO_2 concentrations were a predictor for morbidity among children in a study that looked into winter air pollution in Eastern Europe (Peters et al., 1996).

3.6 Health impact of non-particulate emissions

Besides particulate matter and related species, combustion in aircraft engines also leads to other emissions that affect human health (CE Delft, 2022a). Although these are not the focus of this report, their relation to human health is mentioned here for completeness.

Volatile organic compounds (VOCs) are formed upon incomplete combustion of fuels and lubrication oil emissions in the engine (Fushimi et al., 2019) (Ungeheuer et al., 2022). They are a great contributor to the formation of ozone at ambient levels and as such indirectly responsible for adverse health effects. Ozone (O_3) may cause respiratory problems, chronic obstructive pulmonary disease (COPD), strokes, and may be related to cardiovascular diseases. Additionally, some VOCs have been classified as carcinogens and can be found in jet engine exhausts (Heeb et al., 2024).

Nitrogen oxides (NO_x) consist of nitrogen dioxide (NO_2) and other gaseous oxides containing nitrogen. Burning fuels is the main source of NO_x. NO_x emissions cause direct effects through the formation of NO_2, and indirect effects through the formation of secondary inorganic aerosols and ozone. Short exposures to elevated concentrations of NO_2 can irritate airways in the human respiratory system and can aggravate respiratory diseases (particularly asthma). Longer exposures contribute to the development of asthma and potentially increase susceptibility to respiratory infections. Reduced lung function growth is also linked to NO_2 at concentrations currently measured (or observed) in cities of Europe and North America. Nitrogen oxides react with other chemicals in the air to form both particulate matter and ozone. Both of these are also harmful when inhaled due to effects on the respiratory system.

Carbon monoxide (CO) is a colourless, non-irritant, odourless and tasteless toxic gas. Therefore, it is not detectable by humans either by sight, taste or smell. It is produced by the incomplete combustion of carbonaceous fuels such as wood, petrol, coal, natural gas and kerosene. Breathing air with a high concentration of CO reduces the amount of oxygen that can be transported in the bloodstream to critical organs like the heart and brain. Low levels of indoor CO can cause fatigue in healthy people and chest pain in people with heart disease. Moderate to higher concentrations of indoor CO are associated with symptoms such as impaired vision, reduced brain function, headaches, dizziness and nausea, and even death. Outdoor emissions of CO contribute to ozone formation.

3.7 Conclusions

In this chapter, we have reviewed and summarised the scientific evidence on the effect of aviation emissions on human health. Aircraft emissions, and UFP in particular, are responsible for worsening the symptoms of asthma and respiratory diseases. Long-term exposure to UFP has been linked to effects on the cardiovascular system.

Children are especially susceptible to pollution: postnatal exposure to air pollution has been associated with increased infant mortality, and air pollution affects lung growth and function during childhood. Even associations are found between airport emissions and childhood leukaemia.

Other susceptible groups include elderly, groups of lower socioeconomic status, and airport personnel.

$PM_{2.5}$ concentrations are positively correlated to mortality and morbidity. Approximately 14,000 early deaths each year are due to $PM_{2.5}$ emissions by aviation. Particulate matter becomes increasingly more toxic when it consists of smaller particles, due to increased surface area per unit of mass, and the possibility to travel further through the human body. As a result, UFP can cause inflammation not only in the lungs, but also in more distant organs. In addition to all effects of general PM, UFP health effects include COPD, pulmonary fibrosis, and lung cancer.

The health effects of UFP are better correlated with particle number than total mass. Hence, using the number concentration instead of the mass concentration provides a better estimate into the health effects

Of all types of PM, black carbon is the best predictor for all-cause mortality. Besides the effects of UFP, BC is associated with low birth rates and heart attacks. Sulphur oxides coat nvPM and form volatile aerosols, which may further react to form sulphuric acid. The health effects are similar to the effects of $PM_{2.5}$ emissions.

Non-particulate emissions of aircraft engines include VOCs, nitrogen oxides, or carbon monoxide. The health effects associated with these emissions include respiratory problems, COPD, strokes, cardiovascular diseases, reduced lung function growth and fatigue.

4 Health impacts of aviation PM emissions around European airports

4.1 Air quality studies around airports

In a systematic literature review from 2021, 70 studies were selected that investigated near-airport air quality (Riley et al., 2021). Of these, 30 include UFP (16 in the United States, 11 in Europe, 3 other). In general, particle number concentrations are positively correlated to the flight activity. Particle number concentrations are consistently significantly higher close to and downwind from airports than far away and upwind from airports. Generally, compared to typical baseline concentrations, between 2- and 5-fold increases in particle number concentrations of UFP are measured on the flight pathways, and on locations close (0-3 kilometres) and downwind to the pathways. Further from the airports, elevated UFP concentrations are still measured. One study found a 2- and 1.33-fold concentration increase 4 and 7.3 kilometres downwind from the airport, compared to the concentrations during other wind directions (Hudda et al., 2016). Another study still measures a twofold increase in UFP 18 kilometres downwind (Hudda & Fruin, 2016).

The concentrations of $PM_{2.5}$ near airports are not in all reviewed studies incontestably correlated to flight activity or wind direction with respect to the airport location. For example, $PM_{2.5}$ concentrations in central London where similar or higher than the concentrations at London Heathrow Airport. This suggests that $PM_{2.5}$ in these situations are primarily from other sources than aircraft emissions.

Black carbon is estimated to be elevated in the vicinity of airports up to ten kilometres. In the take-off downwind at Los Angeles International Airport, a 12-fold increase of BC was measured. This was comparable or lower than observed at nearby freeways. At Boston Logan Airport, a 30% increase in BC concentrations were measured when the area was downwind of the airport and aircraft trajectories, compared to other wind directions.

Figure 11 - Measured particle concentrations around LAX

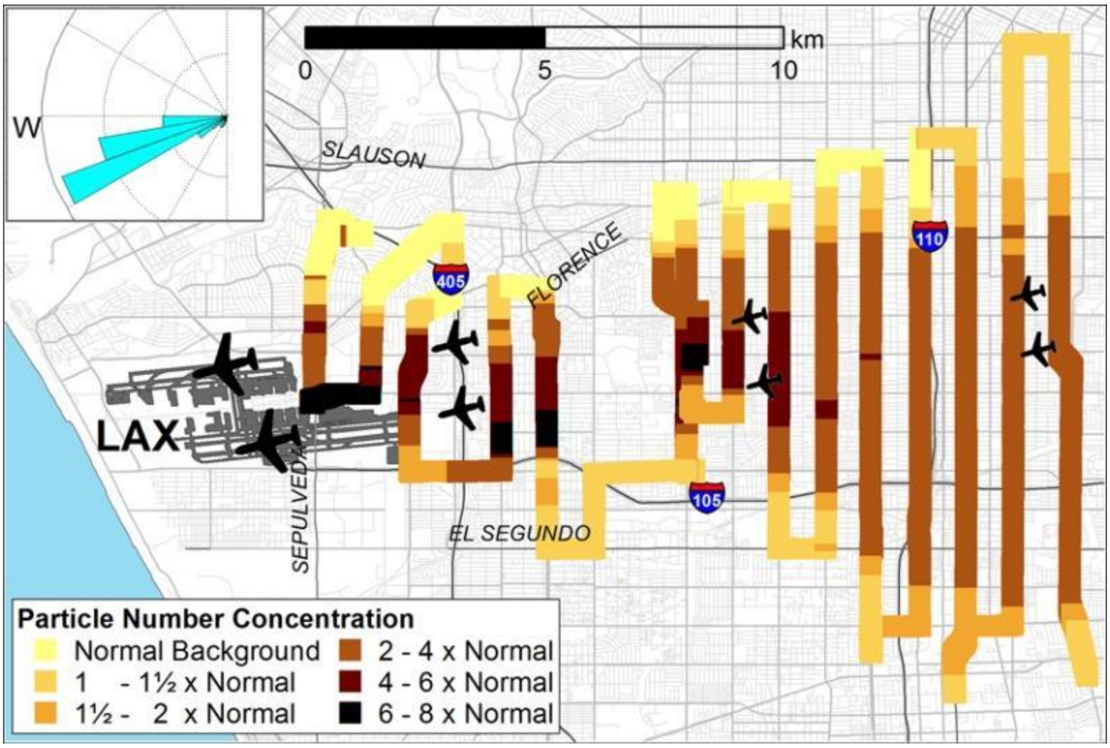


Figure adapted from Hudda et al. (2014).

One study in particular has measured particle concentrations in residential areas near Los Angeles International Airport (Hudda et al., 2014). In downwind areas up to ten kilometres, a 4- to 5-fold increase in particle number concentrations over nearby baseline concentrations were measured. At 8 kilometres downwind, 75,000 particles per cubic centimetre were measured. In residential areas closer to the airport, even 6- to 8-fold increases in particle number concentrations were measured.

Black carbon measurements showed similar spatial patterns to total particle concentrations, and had a concentration of about 1 microgram per cubic metre (Hudda et al., 2014). The corresponding number of particles was not measured. Nevertheless, these results show that at take-off, at least some aircraft produce measurable increases in black carbon concentrations in the downwind area. Another study calculated that this corresponds to a BC emission factor of roughly 0.1 g/kg fuel, compared to 0.4 g/kg fuel (mass) and 8×1,015 particles/kg fuel (number) of $PM_{2.5}$ emissions total (Shirmohammadi et al., 2017). In other words, BC emissions were measured to be a quarter of the $PM_{2.5}$ mass emissions.

Overall, these studies suggest that aviation emissions potentially cause a significant increase in the number of ultrafine particles. The mass concentration is mainly driven by other sources, such as road transport, suggesting that aviation emissions especially increase the concentrations of the smallest particles.

4.2 Epidemiological studies around airports

The National Institute for Public Health and the Environment of the Netherlands (RIVM) has investigated the short-term and long-term effects of higher exposure to ultrafine particles among residents around Schiphol Airport, Amsterdam (Janssen, Hoekstra, et al., 2022) (Janssen, Houthuijs, et al., 2022). They investigated the relation between emissions of UFP due to aviation around Schiphol airport on the one hand, and the following health effects on the other hand:

1. Respiratory effects.
2. Cardiovascular effects.
3. Pregnancy Outcomes.
4. Neurological effects.
5. Metabolic effects.
6. General health.

The study is the first published field research that correlates measured UFP emissions at airports and measured health effects in the region. This is in contrast with previous studies that either measure aviation-related UFP and infer health effects or collect data on health effects and name UFP as (one of the) causing factors.

Table 2 - Health effects of UFP on residents living around Schiphol Airport

Health effects	Long-term exposure to UFP	Short-term exposure to UFP
Respiratory effects	No associations	Demonstrated
Cardiovascular effects	Indicative evidence	Indicative evidence
Pregnancy Outcomes	Indicative evidence	-
Neurological effects	Deficient evidence	-
Metabolic effects	Deficient evidence	-
General health	No associations	-

Source: (Janssen, Houthuijs, et al., 2022).

Note: 'Demonstrated' means a highly probable link to proven effects is found. 'No association' means a relative risk close to 1. 'Indicative evidence' means a significant correlation which is not robust in sensitivity analysis, or a non-significant but mainly relation that was close to significant in sensitivity analysis. 'Deficient evidence' means there is a positive correlation that is not robust to sensitivity analysis.

Table 2 lists the results of this study. All these results are corrected for correlations with co-pollutants ($PM_{2.5}$, PM_{10}, NO_2, soot). Hence all listed effects are strictly due to UFP from aviation. Indicative evidence means associations are possible; however, too much uncertainty prevents definitive conclusions. Deficient evidence means that there is not enough information to deduce the association. Although the current research gives deficient evidence for relations between UFP exposure and diabetes and dementia, the authors propose these links may exist nevertheless and recommend further research.

In more detail, the study concluded the following:

- **Respiratory effects**: Short-term exposure was found to worsen existing respiratory problems and increase the use of asthma medication. No evidence was found for *development* of asthma, lung cancer and death due to respiratory diseases as an effect of UFP.
- **Cardiovascular effects**: A probable association was found for the start of medication use for cardiovascular diseases. On primary endpoints, no evidence was found for associations of UFP exposure with cerebrovascular diseases and strokes, coronary artery disease and high blood pressure. Based on self-reports, a strong association was found between UFP exposure and heart attack incident rate and high blood pressure.

A probable relation to death by cardiac arrhythmia was established, and possible associations with cardiovascular diseases and strokes were found. For adults, an association between UFP exposure and heart functioning was observed.

- **Pregnancy outcomes**: A possible association of UFP exposure and early birth and SGA-born children (small for gestational age). A probable relation with congenital abnormalities was found. No association between death at birth and UFP exposure was found.
- **Neurological effects**: The measured medication use for dementia was strongly associated to UFP exposure. Mortality due to Alzheimer was found as a possible association. Dementia and Parkinson's as causes of death were negatively correlated to UFP exposure. Also, no association between UFP exposure and increased medication use for Parkinson's was found.
- **Metabolic effects**: Self-reported diabetes and increase in medication use for diabetes showed a strong association to UFP exposure. In contrast, *starting* medication for diabetes and mortality by diabetes both were not associated.
- **Psychological effects**: No association was found between UFP exposure and use of anti-depression medication and medication use for psychological stress. Use of anti-depression medication in the group 6-19 years and ADHD medication in ages 16-19 years were both negatively correlated.

4.3 Estimates of potential health impact of UFP around major European airports

In this section, we will use the results obtained by Janssen, Houthuijs, et al. (2022) to find a crude first-order estimation of what the health effects caused by aviation-relation UFP in Europe could be. Note that the actual impacts are dependent on many unknown local factors, such as the atmospheric circumstances and the specifics of the population living near the airport.
Therefore, an extrapolation as in this report cannot be more than a crude first-order estimate of potential health effects. The results on health effects and UFP concentrations around Schiphol Airport are extrapolated to the 32 busiest airports in Europe (ranked on flight activity, 2019). This accounts for 42% of all annual European flights in 2019, and 68% of the total flights in the countries considered in this study. We consider health effects for the population in a 20 km radius from the airports. This area is further differentiated in proximity to the airport, to radii of 5, 10, and 20 km. The total population considered accounts for more than 10% of the entire population in the included countries (EU-28 + Switzerland + Norway, population in 2015).

4.3.1 Methods and assumptions

We first estimate the concentrations around Schiphol Airport. These estimations are based on the particle concentration model by Janssen, Houthuijs, et al. (2022), displayed in Figure 12, and UFP measurements in 2015 around Schiphol (RIVM et al., 2015).
The latter reported that the UFP concentrations attributed to Schiphol were up to 15,000 particles/cm^3 in living areas close to the airport (up to 6 km). On a distance of 15 kilometres, 3,000 particles/cm^3 were attributable to Schiphol.

The calculation values in this report are based on these results and are shown in Figure adopted from Janssen, Houthuijs, et al. (2022).

Table 3. These are the mean concentrations per area, taking into account the circular shape of the areas. For the 0-5 km area, this is lowered to 10,000 (p/cm^3) to account for the fact that the population density in this area will be shifted towards the outer regions.

Figure 12 - UFP concentrations around Schiphol Airport, averaged from 2006 until 2019. About 2.5 million residents are living in the depicted research area. The white contour is the airport terrain.

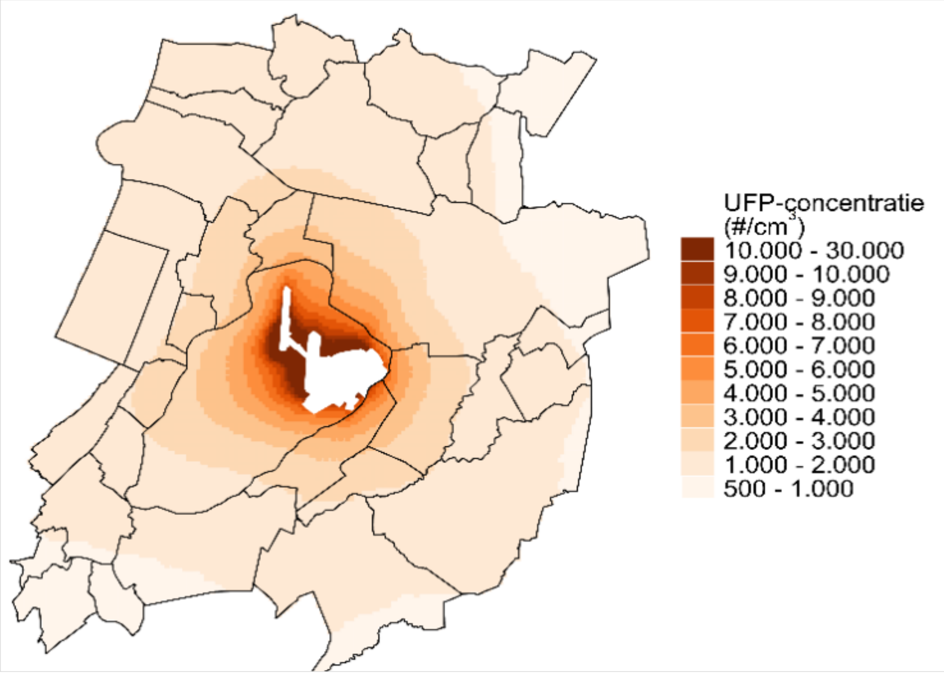


Figure adopted from Janssen, Houthuijs, et al. (2022).

Table 3 - Concentrations measured around Schiphol, Amsterdam. The calculation values are the values used for extrapolation of UFP concentrations around other airports in Europe.

Distance	Concentrations (p/cm^3)	Calculation values (p/cm^3)
0-5 km	4,000-30,000	10,000
5-10 km	3,000-6,000	4,200
10-20 km	1,000-4,000	2,200

Then we estimate the UFP concentrations around all considered airports by extrapolating the UFP concentrations around Schiphol Airport. Concentrations around all other airports are calculated by the number of flights, compared to the number of flights of Schiphol Airport.[4] In other words, we use flight activity as an indicator for the airport's total emissions and assume flight activity relates linearly to UFP concentrations. Further, we assume that UFP emissions due to taxiing are linearly related to the flight activity.

[4] Data available on www.ec.europa.eu/eurostat/databrowser/view/AVIA_TF_ACA__custom_4421344/default/table?lang=en.

Next, the population in circular areas with corresponding radii of 5 km, 10 km, and 20 km around the airports in 2015 are calculated, using Schiavina (2019).[5] A slight overcount due to overlap areas Charles de Gaulle and Orly is corrected (5 km overlap of the 20 km radius).[6] The centres chosen correspond to the estimated geographical centre of the airports.[7]

In order to estimate the health effects of UFP exposure, we use the epidemiological data obtained by Janssen, Houthuijs, et al. (2022) around Schiphol airport. This study obtained relative risks for the health effects listed in Table 4. These relative risks are given for UFP concentrations of 3,500 particles/cm^3. We have assumed that these risks scales linearly with concentration, without lower or upper limit (RIVM, ongoing).

In summary, using the relative health risks combined with the population data and UFP concentrations around the considered European airports, we are able to calculate a crude first-order estimate of the potential health effects on the population near those airports.

The health effects listed in Table 4 are the ones for which Janssen, Houthuijs, et al. (2022) found a strong association. In addition to these effects, indicative evidence for cardiovascular diseases and birth outcomes were also found. In this report, we only calculate the effect of health outcomes with a strong association. It should be noted that some of these health effects do not have sufficient evidence of being correlated to UFP exposure on all measured endpoints. For example, for diabetes, the self-reports and medication use from the Gezondheidsmonitor (GGD Health Monitor) show a clear correlation to aviation-related UFP exposure, while no relation was found for diabetes medication use in a cohort study (Janssen, Houthuijs, et al., 2022). Therefore, the authors conclude that there is insufficient evidence for the relation of long-term exposure and diabetes.

The above results strongly suggest that further research is necessary. The strong correlations on some endpoints nevertheless suggest that the link to these health effects exists. In this research, we extrapolate the results that *do* show a clear association to long-term aviation-related UFP exposure, that is, for example, the self-reports on diabetes. This report serves as a first-order estimation of the possible health effects and is by no means a robust epidemiological study. Hence, one must be careful to draw too strong conclusions from the results, as the relation between diabetes and aviation-related UFP exposure is not clear from all endpoints.

Similarly for dementia, the study around Schiphol airport shows a clear association between medication use (from the cohort study) and long-term aviation-related UFP exposure.

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[5] The population data used in this study is from 2015 and was declared obsolete (due to the availability of more recent population data) after the current research project was finished. Given the small changes in total population in the European Union from 2015, the overall results are not significantly affected. However, airport specific results may deviate in from the current situation in this respect, due to possible significant regional population developments.

[6] Assuming homogenous distribution of the population around the airports, this overlap can be accounted for by subtraction of 16% reduction in residents 10-20 km for both airports. Note that the UFP concentrations are not overcounted; each airport contributes separately and independently.

[7] The estimated population is very sensitive to the chosen centre of the circles. Especially in cases where the airport is close to a densely populated area, such as Madrid, Rome, and Paris, a shift in the order of 100 meters can cause a difference in the order of 100 thousand in population count. In interpreting the results, one should be cautious of this sensitivity. Instead of taking the radius from a single point, in more advanced methods one could take the range from each point of the contour of the airport. This will always result in a higher population count. In this regard, our population estimations are an underestimate.

However, long-term exposure to aviation-related UFP actually correlates negatively with death due to dementia, but positively to Alzheimer's disease specifically (Janssen, Houthuijs, et al., 2022). This shows in the first place that more scientific inquiry in the effects of UFP exposure on neurodegenerative diseases is necessary. In this report, we only extrapolate the relation between UFP exposure and medication use for dementia, as this was shown to be a clear association. Again, as the correlation from different endpoints in the used literature are not consistent, the results must not be overstated, and all data must be interpreted very carefully as first-order estimations.

Table 4 - Relative risks as a consequence of long-term UFP exposure of 3,500 p/cm^3. These risks are assumed to scale linearly with concentrations.

Health effect	Age group	Relative Risk (3,500 p/cm^3)	Error value (95% confidence intervals)
High blood pressure (medication use)	19+	1.05	(1.00-1.11)
Diabetes (medication use)	19+	1.08	(1.00-1.17)
Diabetes (self-reported)	19+	1.16	(1.02-1.33)
Medication use for dementia	40+	1.14	(1.013-1.286)

Source: (Janssen, Houthuijs, et al., 2022).

Text box 1 - Relative risks

What is a relative risk?

A Relative Risk or Risk Ratio (RR) is a number that characterises the risk for a given health effect. To find the effect of exposure to a risk (i.e., UFP in this report), it compares the risk of a health effect in the exposed group compared to the risk in a non-exposed group. The RR is calculated as:

$$RR = \frac{risk\ in\ exposed\ group}{risk\ in\ comparison\ group}$$

The risk in a group is simply the proportion of the attacked (affected) people to the total (affected + healthy) people in the group. The increased risk for the exposed group (in percentage) is calculated as:

$$Increased\ risk\ = (RR - 1) \times 100\%$$

For example, if the RR is 1.15, the increased risk for the exposed group is 15%.

How can a RR be used to make predictions?

The RR is a number that can be used to predict the number of people that will be affected when a group is exposed to the subject risk. The estimated number of people affected in an exposed group is:

$$people\ affected\ in\ exposed\ group\ = risk\ in\ exposed\ group\ \times\ number\ of\ people\ in\ exposed\ group$$

If a RR is given, and one wants to calculate the number of people affected in an exposure group, the risk in the comparison group must be known:

$$people\ affected\ in\ exposed\ group\ = RR\ \times risk\ in\ comparison\ group\ \times\ number\ of\ people\ in\ exposed\ group$$

In other words, the 'general' risk for health effect must be known in order to estimate the effect of the exposure.

How should a RR be interpreted?
In this study, we use a RR to indicate the effect of exposure to UFP. However, UFP exposure for people living around airports is not a single incident, but a continuous exposure for the time they are living there. Therefore, the relative risk we use in this study must be interpreted as a number that assigns a *static* risk to the population living there at the moment of study.

In other words, the actual risk for diseases is probably dependent on the entire exposure history, which varies from person to person. The UFP concentrations around airports are likely to have increased in the past decades since air traffic has been increasing strongly. Therefore, the actual risk for diseases for the people living there now does not only depend on their historic exposure, but also on the future UFP concentrations, which in turn depend on the development of air traffic, aircraft technology, and fuel use.

In short, the relative risk in this study should be treated with care and should be understood as a general estimate to the actual risks due to UFP exposure. This articulates the need for more scientific investigation in the exposure-effect relation for UFP.

The calculated number of affected people is as a consequence also a *total* number of people that would be affected, based on the current population and UFP concentrations. This number is based on relative risks, which are calculated for a given period of time and given an unknown exposure history. There are many more factors that are unknown in the exposure-effect relation, which would require more scientific investigation. Our calculations provide a crude first-order estimate of the possible impact of UFP exposure.

All associated health effects by Janssen, Houthuijs, et al. (2022) are associations that persist after correction for other pollution, such as $PM_{2.5}$, soot, and NO_x, and also for noise (for long-term health effects). Since UFP are believed to be the most toxic particles within $PM_{2.5}$, it is expected that many health effects of $PM_{2.5}$ are at least in part attributable to UFP. Table 5 shows all health effects (CE Delft, 2023) considered in calculating environmental prices for $PM_{2.5}$. It is not known to what extent UFP is responsible for the health effects of $PM_{2.5}$. Therefore, it is not possible to quantify these health effects in terms of UFP exposure. Additionally, the different quantification in exposure (mass versus particle number concentration for PM and UFP, respectively) prevents direct translation of the $PM_{2.5}$ health risks to UFP health risks.

Table 5 - Relative risks of health effects as a consequence of exposure to $PM_{2.5}$

Endpoint (incidents)	Age group	RR per 10 µg
Mortality, all natural causes	30+	1,080
Hospitalisations, cardiovascular diseases	All	1,009
Hospitalisations, respiratory diseases	All	1,019
Restricted activity days (RAD)	All	1,047
Work loss days (WLD)	20-65	1,046
Days with asthma symptoms for children suffering from asthma disease	5-19	1,028

Source: CE Delft (2023) based on WHO (2013).

In calculating the health effects, we assume a circular symmetric distribution of UFP around the airport. In other words, wind directions are not considered. Health effects may be modelled too low for populations that are located in an area that is a large part of the time downwind of the airport (this may be the case for airports near sea like Barcelona, Lisbon, etc.). On the other hand, health effects may be modelled too high for airports where the prevailing wind direction is opposite to densely populated areas.

To find the absolute reduction in number of affected people, additional information on the occurrence of these health effects is necessary (see the box above). For high blood pressure and diabetes, data from Eurostat on self-reported chronic morbidity has been used[8]. This data is from 2019. For dementia medication, data from (Ju et al., 2021) has been used, which gives daily doses per inhabitant by country. This data is from 2018.

For dementia, the type and amount of medication used varies strongly by country (Ju et al., 2021). Our calculation only assumes that the increase in general medication use due to UFP exposure will be similar in relative terms as in the Netherlands. The link between medication use and prevalence of dementia is not further quantified and is likely to vary amongst countries.

Summarising, the following assumptions have been made:
- By extrapolation of the UFP measurements from Schiphol Airport to all other European airports in the scope of the study, we assume geographical independence of UFP formation and spatial distribution.
- We assume the UFP distribution is radially symmetric around the airport and (on large time scales) independent of wind direction.
- We assume UFP concentrations scale linearly with the number of flights at an airport. The population living near airports is calculated as the population in a radius of 5, 10, and 20 kilometres from the estimated geographical centre of the airport.
- By extrapolation of epidemiological data from Schiphol Airport to all other airports in the scope of the study, we assume health effects of UFP are not dependent on regional, social, or environmental factors.
- We assume health effects of UFP scale linearly with concentrations, without lower or upper bound.
- Besides mortality, we only take health effects into account that are only attributable to UFP and are not correlated or attributable to $PM_{2.5}$, soot, or any other air pollution. The health effects calculated are shown to have a strong association to UFP (Janssen, Houthuijs, et al., 2022).
- The population data used is from 2015. The flight data is from 2019. The UFP measurements were executed in 2017 and 2018. The health effects data is from 2018 and 2019. We assume insignificant deviation on our results from the slight spread in years of measurement of different quantities.

The absolute numbers of affected people should be interpreted with care. Besides all assumptions listed above, one must note that the absolute numbers are based on current populations and current UFP concentrations. The absolute numbers hence refer to the total number of people which are likely to be affected by a disease, relative to the current population. Possible future change in population as well as UFP emission are *not* incorporated into this number. Different historic growths of the emissions at airport level and local circumstances are also not considered.

[8] Available on www.ec.europa.eu/eurostat/databrowser/view/HLTH_EHIS_CD1E__custom_1301246/bookmark/table?lang=en&bookmarkId=46974e2c-13e5-4ff4-a1de-4b68019a87bf

It should be noted that this report serves the goal of a first-order estimate in the largely unknown and unresearched area of concrete health effects on population due to aviation-induced UFP concentrations around major European airports. In more advanced methods, some of the above assumptions would have to be revised.

4.3.2 Results – UFP concentrations and population around airports

By assuming UFP concentrations scale linearly with the flight activity on airports, measurements from Janssen, Houthuijs, et al. (2022) are extrapolated to major airports in Europe. The average concentrations have been calculated for three circular areas around the airports, with radii of 5 km, 10 km, and 20 km. The results are shown in Figure 14. Naturally, higher concentrations are found closer to the airports.

Considering the populations in the same areas around the airports, Figure 13 shows the relation between population and aviation-related UFP exposure. In other words, this figure shows how many people are exposed to certain degrees of UFP concentrations due to aviation, on top of the background UFP concentrations.

Figure 13 - Total people exposed (in millions) by aviation-related UFP for the included European airports, broken down by concentration levels and distance to the airport

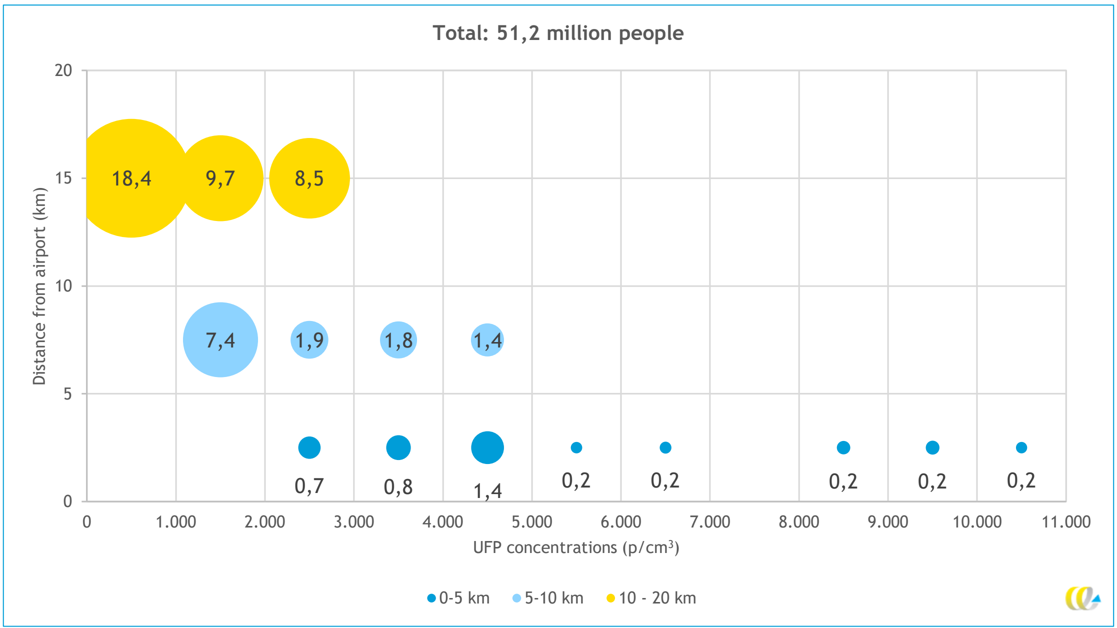
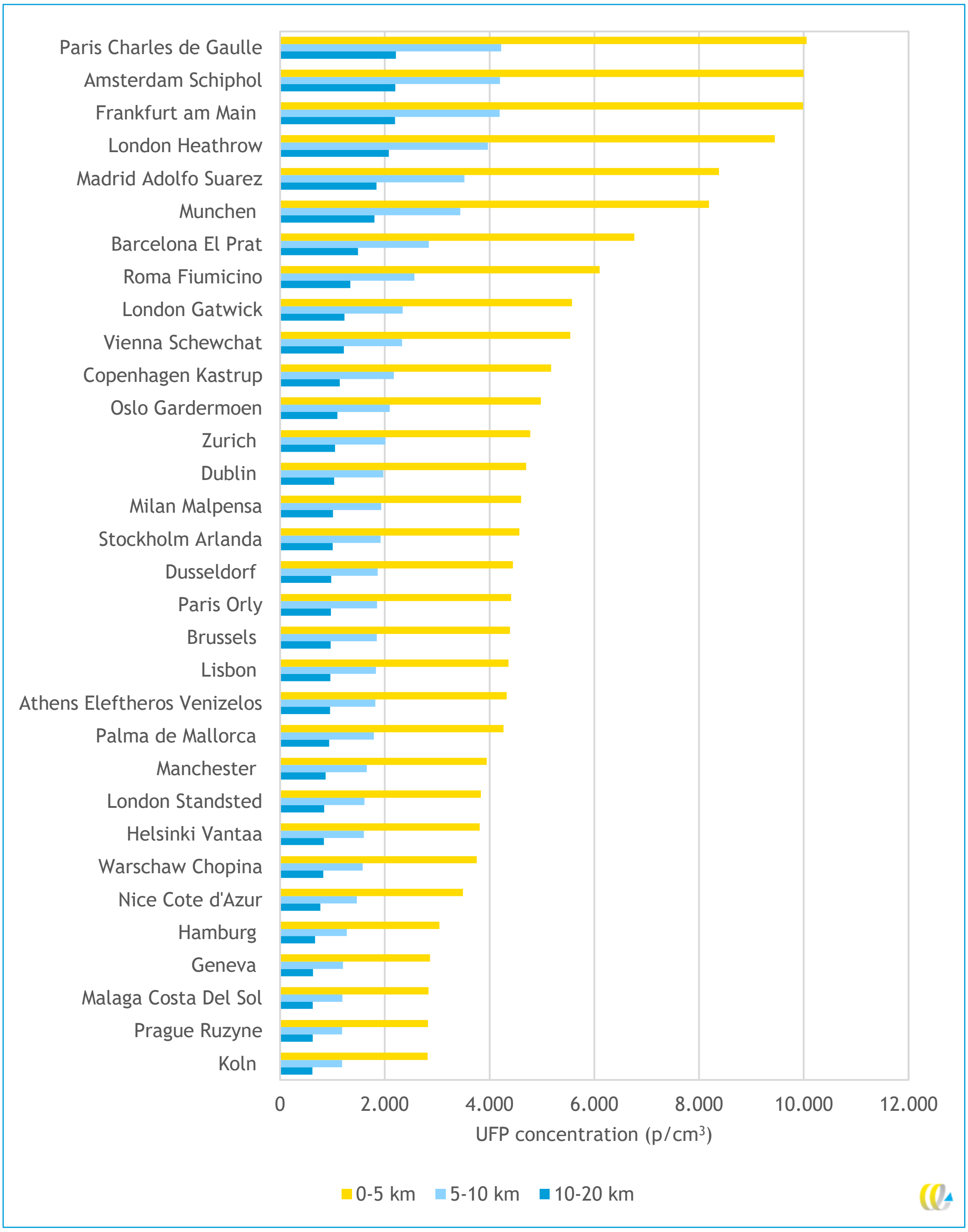


Figure 14 - Estimated UFP concentrations due to aviation activity around major airports in Europe. The figure shows concentrations in a radii of 5, 10 and 20 km from the airport.



Note: Estimations are based on extrapolations from Janssen, Houthuijs, et al. (2022) on the basis of the number of flights per airport.

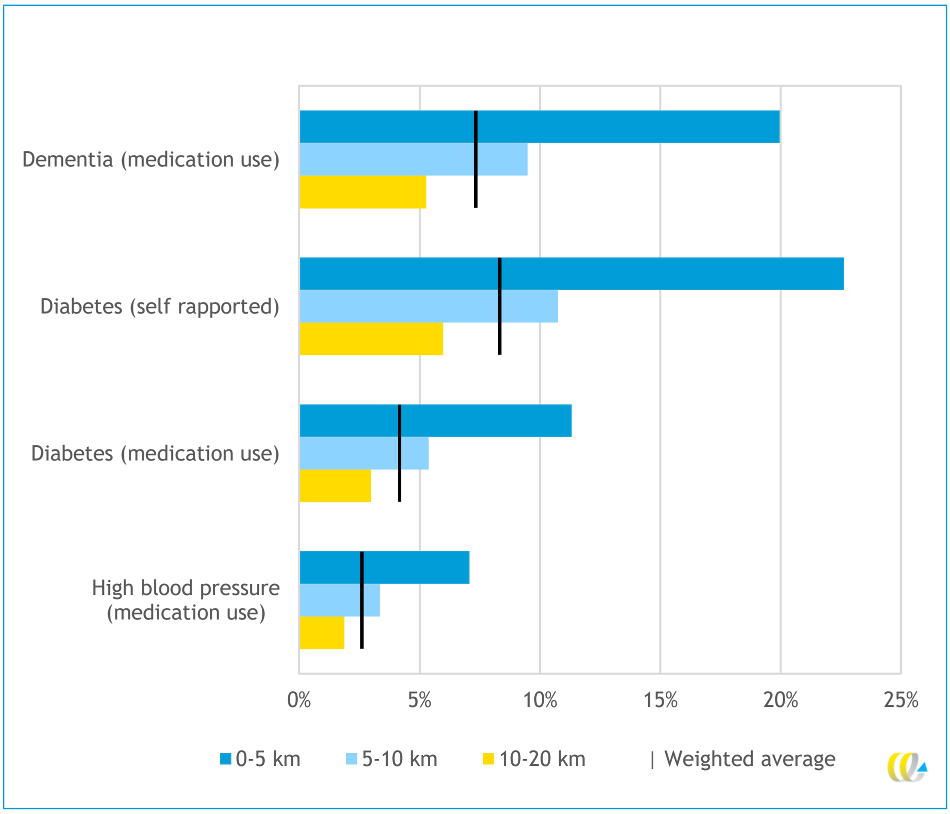
4.3.3 Results – Health effects of UFP

The UFP concentrations around major European airports, as calculated in the previous paragraph, are used to infer the health effects on the population living near these airports. First, the relative risks for dementia, diabetes and high blood pressure as reported by Janssen, Houthuijs, et al. (2022) are used to determine the relative risks around each included airport, per radius. This is calculated by assuming linearity between UFP concentration and relative risks. The average relative risks over all airports for these effects is found by taking the weighted average of the risks in all areas, relative to the number of people in those areas. The results are displayed in Figure 15.

According to this calculation, on average, population around major European airports (included in this model) have a 7.3% higher risk for dementia (judging from medication use) than the general population, a 4.2-8.3% higher risk for diabetes (judging from medication use or self-report, respectively), and a 2.6% higher risk for high blood pressure (based on medication use), see Figure 15. These risks represent the increased risk due to aviation-induced UFP exposure, and is a relative increase compared to a similar population not exposed to these UFP concentrations.

As UFP concentrations near airports are higher, populations living closer to the airport have an even higher risk for these health effects: the increased risk for high blood pressure for populations in a radius of 5 km to the airport is 7%. The risks for dementia and diabetes are as high as 20 and 23%, based on medication and self-report, respectively.

Figure 15 - Relative risks for UFP-associated health effects. These risks are the weighted average for all populations around airports included in this report.



Using the baseline occurrence of the diseases in European countries[9], an estimate can be made on the number of people that likely suffer from these diseases due to aviation-induced UFP. The results are given per country in Annex A, and are used in the next paragraph. One must note that the absolute number of people affected is relative to the current population. It should be treated cautiously, and it should not be interpreted as an incidence rate.

4.4 Reduction of health impact by hydrotreatment

In this report the scientific evidence has been summarised for the relation between jet fuel composition and human health, in particular regarding particulate matter emissions.
In Chapter 2, the relation between jet fuel composition and PM emissions was demonstrated. Next, in Chapter 3, the effects of PM emissions on human health were reviewed.

Section 2.3.2 showed how HEFA SAFs and hydrotreatment of fossil fuels leads to a reduction of PM emissions due to the lowered naphthalene, aromatics, and sulphur content. Therefore, usage of HEFA SAFS and hydrotreated fossil fuels instead of traditional fossil fuel in aviation will lead to a reduction in health impact. In this section, we give a crude first-order estimate of the possible health effects of hydrotreatment of aviation fuels.

4.4.1 Model and assumptions

The model developed in Section 4.3 can be used to quantify the estimated health effect of UFP reduction by the implementation of hydrotreatment of fossil fuels on European scale. To this end, some assumptions have to be made regarding HTFF and PM emissions.

First, it is assumed that all UFP emissions in the model (and accordingly, all UFP measured by Janssen, Houthuijs, et al. (2022) around Schiphol Airport) are due to either the combustion of aviation fuels or caused by lubrication oil. This is a stronger assumption than linearity of emissions with aircraft activity, since for example Ground Support Equipment (GSE) are also in part responsible for PM emissions of airports. However, the calculated UFP emissions around Schiphol Airport by Janssen, Houthuijs, et al. (2022) are UFP emissions exclusively attributable to aircraft traffic, where other sources are eliminated (Voogt et al., 2019).

Second, we assume that currently all aircrafts use Jet A-1 fuel, such that data from Figure 6 for 100% Jet A-1 fuel represents the baseline scenario where no HTFF or SAF is used. This assumption is based on the fact that in 2018, only 0,01% of all aviation fuel came from alternative sources (Pavlenko, 2021).

Third, it is assumed that the scientific measurements as shown in Figure 5 and Figure 6 on the particle number reduction by burning HEFA SAF fuels in an APU is representative for the particle number reduction in burning HTFF in an aircraft engine. This can be understood by the similarity in chemical composition, especially their similar low naphthalene and sulphur content. Concretely, the assumption on reduction in particle number concentration follows the trendline of Figure 5, which is the measurement of the MES (main engine start) operating condition of the APU. According to the authors, these measurements are representative for nvPM emissions in working aircraft engines (Lobo et al., 2015).
This is supported by data from other studies, see for example Schripp et al. (2022).

[9] Calculable from www.ec.europa.eu/eurostat/databrowser/view/HLTH_EHIS_CD1E__custom_1301246/bookmark/table?lang=en&bookmarkId=46974e2c-13e5-4ff4-a1de-4b68019a87bf

However, a part of the UFP emissions is not caused by the burned fuel itself, but by the lubrication oils in the engine. One study concluded that for the smallest particles (10-18 nm), lubrication oils are the source of 10-32 mass-percent of the ambient UFP due to aircraft engines (Ungeheuer et al., 2022). For 18-32 nm particles, this was 2-8 mass-percent and for 32-56 nm particles 9 mass-percent. This suggests that especially a significant portion of the very small particles are due to lubrication oils. Another study found that for particles smaller than 30 nm, approximately half of the organic compounds are due to lubrication oils (Fushimi et al., 2019). Note that, as UFP consists of more than organic compounds only, this does not mean that half of the UFP mass is due to lubrication oils.

According to Figure 5, the UFP number concentration is roughly symmetrically distributed between 10 and 100 nm for Jet A-1 fuel, and most particles are in the 30-60 nm range. By lack of more accurate data, we will therefore assume that the 9 mass-percent share in the 32-56 nm range is the best estimate for the overall share of lubrication oils in the total number of UFPs due to aircraft engines. Although this relation between mass and number concentration is not one-to-one, we assume this is also an estimate for the share in number concentration. This is backed by the fact that the majority of particles are in the 30-60 nm range. Therefore, we can assume the mass per particle of the particles in the 32-56 nm range is close to the average mass per particle of the total UFP.

The UFPs due to lubrication oils are not affected by the type of fuel burned (e.g. conventional fossil fuel or HTFF). We assume interactions between the fuel burn emissions and the lubrication oil emissions in the exhaust plume are not dependent on the type fuel either. Table 6 shows the assumed reduction in UFP concentrations for a given share of HTFF in the jet fuel.

Table 6 - The assumed relation between the relative share of HTFF and reduction in particle number concentration, based on the data of Lobo et al. (2015) for the MES operating condition using HEFA SAF blends.

Relative share HTFF	Assumed share of UFP due to fuel burning	Assumed share of UFP due to lubrication oils	Reduction in fuel-related UFP number concentration	Net reduction in UFP number concentration
0%	91%	9%	0.0%	0.0%
20%	91%	9%	13.0%	11.8%
40%	91%	9%	27.3%	24.8%
60%	91%	9%	42.9%	39.0%
80%	91%	9%	59.9%	54.5%
100%	91%	9%	78.2%	71.2%

Fourth, it is assumed that total reduction in nvPM is representative for the UFP number reduction. From Figure 6 it becomes clear that the biggest part of nvPM is in fact UFP. Insofar aviation-induced UFP is volatile, it is assumed to show similar reductive behaviour as nvPM. This is supported by the fact that vPM formation is predominantly driven by the nucleation of sulphuric acids (Owen et al., 2022). In other words, a reduction in sulphur content will prevent the main part of vPM formation. Although no useful quantitative data on the reduction in vPM formation by SAF HEFA fuels is available, Figure 4 shows that sulphur content reduction in HEFA fuels is typically of the same order of magnitude as the decrease in aromatic content. Hence, we estimate that the reduction of vPM is proportional to the reduction of nvPM.

Last, the reduction in emissions by blending HTFF with Jet A-1 in an engine is assumed to be representative for the blending of HTFF with Jet A-1 on a large-scale fuel supply.

While this is a natural assumption, it should be noted that the reduction in particle numbers is *not* linearly dependent on the share of HTFF. In fact, since this relation is modelled to be quadratic, the extrapolation to large scale is sensitive to the distribution of HTFF, and the estimation is most accurate when HTFF is blended equally in all aviation fuel supply systems. This is in fact the scenario of smallest reduction: when HTFF is distributed *unequally* (as illustrative example, half of the aircrafts use pure Jet A-1, while the rest uses a 50/50 blend), this will lead to more net PM reduction than for equal distribution (in this example: all aircrafts use 25% HTFF). This can be seen from the concave shape of the reduction trendline in Figure 5: the average of multiple points on that line will always lie lower than trendline itself. From the small quadratic contribution, it is also clear that the linear extrapolation approximation will only result in a small deviation.

From the reduction in UFP concentrations, reduction in health risks can straightforwardly be calculated given its linear dependence. By the same method as in the previous chapter, these risks can be translated to concrete numbers of affected people, and in this case, to the reduction of affected people by implantation of HTFF.

4.4.2 Results – Health impact

Given the reduction in UFP particle number concentration by blending HTFF into jet fuel, the reduction in UFP concentrations around the airports in Europe as included in the previous chapter can be estimated. From the reduced UFP concentrations, the reduction in health risks as a function of the relative share of HTFF in total jet fuel can be estimated. The results are displayed in Figure 16.

As can be seen from this figure, the risks on UFP-associated health effects decrease for an increasing share of HTFF. Moreover, this decrease progressively advances for higher shares of HTFF. Current fuel standards dictate a maximum blend of 50/50 for Sustainable Aviation Fuels (representative of hydrotreated fuels). The red dotted line in the figure indicates the maximum reduction in health risks that can be achieved by these standards.

Using the data of occurrence of health effects in the country as baseline risk, the health impact of reduction in emissions can be calculated. These results are displayed in Figure 17.

These calculations involve the relative risks, for which a clear exposure-effect relation remains to be determined scientifically. Although it is certain that the risk for the diseases is increased due to exposure to UFP, the precise dependence on historical concentration levels and exposure time is unknown. Therefore, the estimated total number of affected people represents an estimation, based on the current population and current concentration levels, and does not include future changes in any of those aspects.

Figure 16 - Risks on UFP-associated health effects as a function of the share of hydrotreated fossil fuel (HTFF). As fuel standards dictate a maximum blend of 50% SAF (representative of HTFF), the red line indicates the current theoretical maximum of reducing health risks through SAF use.

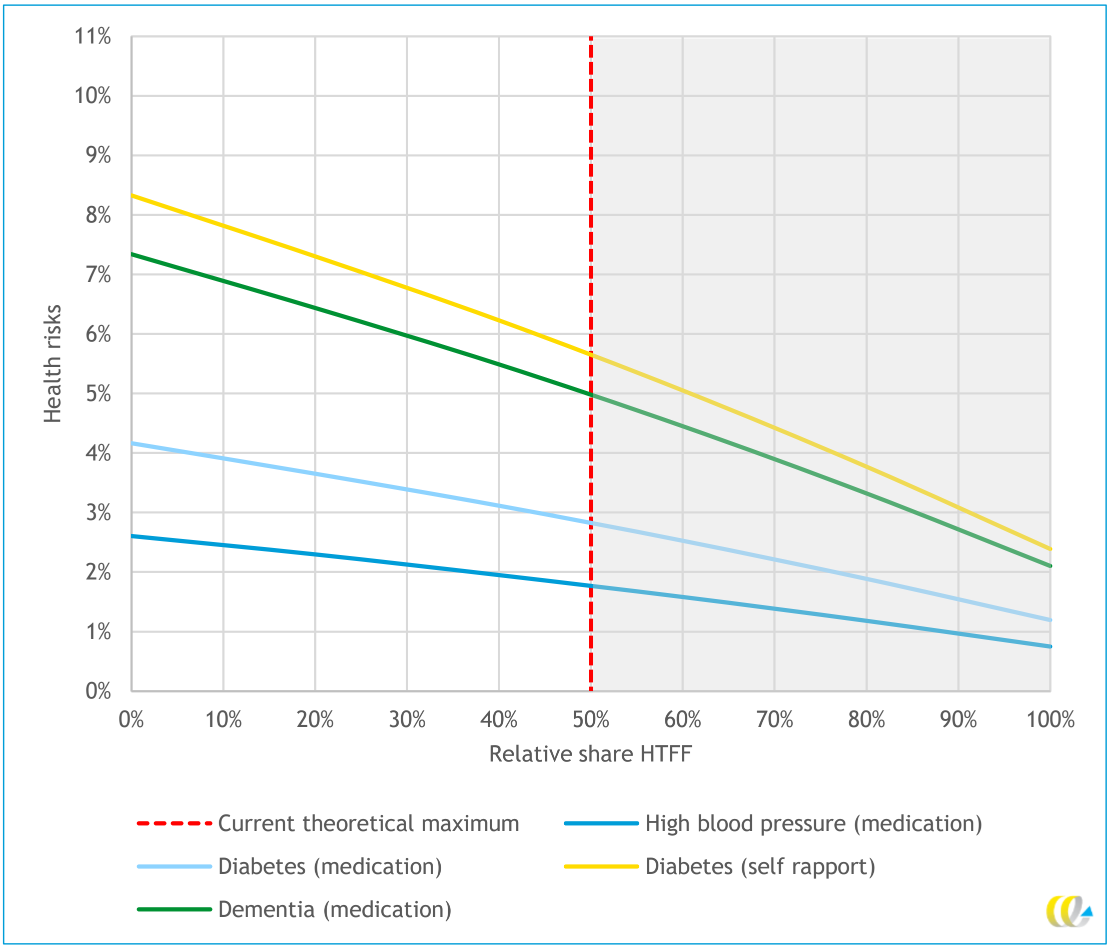


Figure 17 - Estimated reduction of number of people affected by the health effects of aviation-induced UFP. The red dotted line represents the current theoretical maximum of SAF by current jet fuel standards.

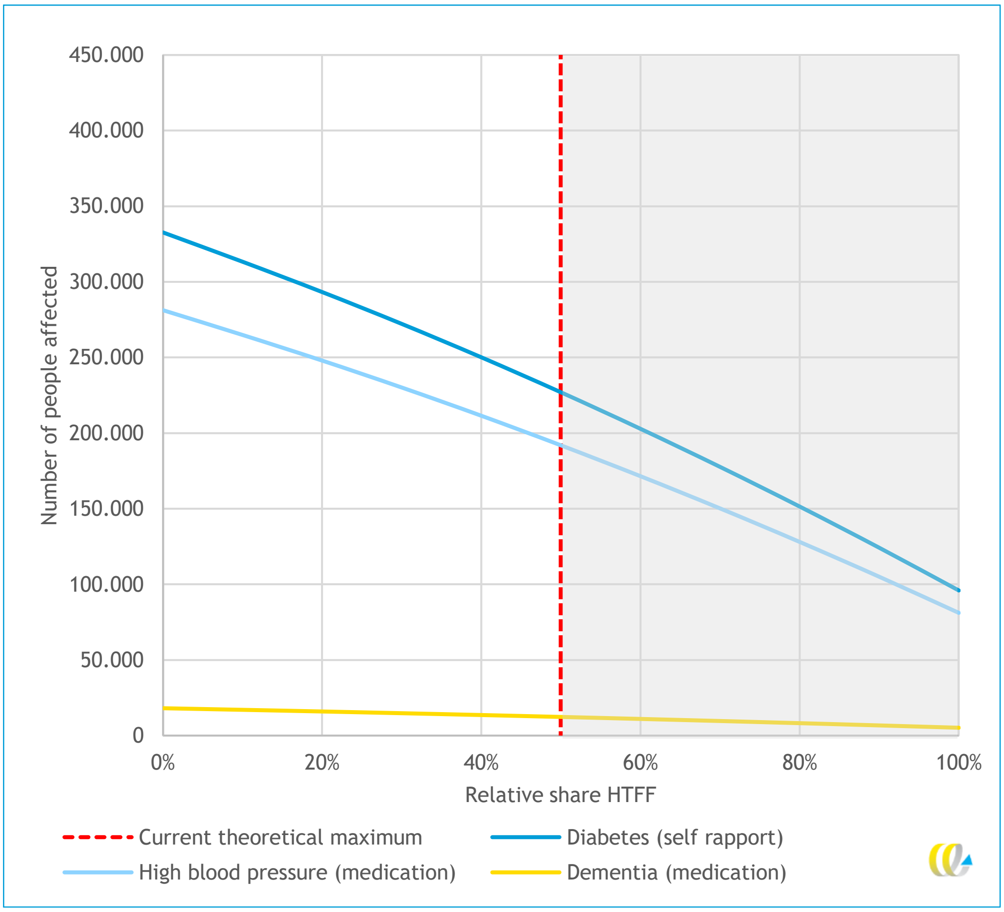


Table 7 - Number of prevented cases of health impact through the use of HTFF. These numbers are calculated for the total population 51 million that live around the airports included in this model.

Relative share HTFF	Prevented cases, per health effect		
	High blood pressure	Diabetes	Dementia
0%	0	0	0
10%	16,000	19,000	1,000
20%	33,000	39,000	2,000
30%	51,000	60,000	3,000
40%	70,000	83,000	5,000
50% (current maximum)	89,000	106,000	6,000
60%	110,000	130,000	7,000
70%	131,000	155,000	8,000
80%	153,000	181,000	10,000
90%	176,000	208,000	11,000
100%	200,000	237,000	13,000

5 Policy measures to address the health impacts of aviation PM emissions

Ultra fine particulate emissions from aviation have a negative impact on health, as shown in Chapters 3 and 4. Reducing the emissions would improve the health of people, especially those living near or working at airports. There are also wider benefits of reducing UFP emissions, notably to reduce the non-CO_2 climate impacts of aviation (EASA et al., 2020).

UFP emissions result from incomplete combustion of aromatic compounds, especially double ringed aromatic compounds (naphthalenes), and from combustion of sulphur present in jet fuel. Hence, lowering the concentration of these compounds will have benefits (see also Chapter 2).

There are generally three options to reduce the concentration of aromatics and sulphur in jet fuel (CE Delft et al., 2022):
1. Produce jet fuel from crudes with a naturally lower concentration of single-and double ringed aromatics and sulphur.
2. Hydrotreat fossil fuels so that aromatic compounds are saturated, and sulphur can be removed in a claus process.
3. Blend fossil jet fuels with sustainable aviation fuels (SAFs) with zero or very low concentration of aromatics and sulphur (note, however, that some types of SAF contain aromatics).

The first option would be hard, if not impossible, to regulate.

Technically, it is possible to produce jet fuels without aromatics (CE Delft, 2022b). These fuels would meet the existing fuel standards which only set a maximum for the aromatics concentration (25% by volume), unless they are blended with sustainable aviation fuels, in which case the standard specifies a minimum aromatics concentration of 8% by volume (CE Delft et al., 2022).

Although the costs of producing fuels with a low concentration of aromatics and sulphur would be higher than the cost of producing conventional jet fuels, the health and climate benefits outweigh these and other additional costs so that economic welfare would increase when these fuels would be used (CE Delft, 2022b). Hence, there are good reasons to mandate or incentivise their use.

CE Delft et al. (2022) have analysed which policies would effectively increase the use of low-aromatics, low-sulphur jet fuels.
- A change of existing jet fuel standards (notably ASTM 1655 (for fuels produced from crude oil), ASTM 7566 (for fuels containing synthesised hydrocarbons) and DefStan 91-091) would have a global effect on fuels used and have the benefit that these standards are accepted and used by all major industrial stakeholder groups and regulators alike. However, the outcome of an initiative to change these standards would be uncertain and outside of the control of the EU, as the technical committees typically

operate by consensus and anybody can apply for membership (ASTM International, 2021).
- A provision in EU legislation that limits the aromatics or sulphur content of aviation fuels. Such a limit could perhaps be introduced in the Fuel Quality Directive (Directive 2009/30/EC), thus extending its scope to aviation fuels. In that case, the limit would apply to fuels sold in the EU. This is the same scope as the revised Renewable Energy Directive, which could be amended with a provision that the renewable fuels supplied to aviation should have a low or zero concentration of aromatics and sulphur. Another possibility would be to adopt a similar provision in ReFuelEU aviation.

A part of UFP emissions is not due to the fuel burning, but to the exhaust of lubrication oil particles. These emissions could be reduced through the development of superior technologies for controlling oil emissions (Fushimi et al., 2019). In this study, we have assumed that there is no interaction between the UFP emissions due to lubrication oil and those due to jet fuel burning.

Lastly, to establish an estimate of the societal impact of UFP concentrations, it is insightful to determine the societal financial impact of UFP emissions. Environmental prices for UFP have been calculated in terms of costs per emitted mass UFP (CE Delft, 2023).
However, the actual impact depends not only on the emitted mass, but primarily on the concentration levels. The societal impact of UFP concentrations is furthermore dependent on the type of area where these concentrations are present (for example, the population in these areas). This is out of scope of this study. Further research into the societal costs of UFP would be necessary, and this will likely increase the awareness of the impact of UFP emissions.

6 Conclusions

Particulate emissions from aviation, and especially emissions of ultra fine particles (UFP) have detrimental effects on human health:

- Many studies have found causal relations between aircraft emissions, and UFP in particular, and worsening symptoms of asthma and respiratory diseases.
- UFP has been found to cause COPD, pulmonary fibrosis (scarring of the lungs), and lung cancer.
- Long-term exposure to UFP has been linked to effects on the cardiovascular system like hypertension.
- Aircraft UFP emissions have been found to potentially cause or worsen diabetes and dementia.
- $PM_{2.5}$ emissions positively correlated to mortality and morbidity. Globally, approximately between 14,000 and 21,200 early deaths each year are due to $PM_{2.5}$ emissions by aviation.

The reason that the impacts of UFP emissions are worse than the impacts of larger particulates is that they can travel further through the human body, and that their surface area relative to their mass is larger so that they can transport relatively more toxins. Children and elderly are more at risk than the average population.

In addition to health risks, PM emissions from aviation also cause the formation of contrails, which contribute to global warming. Reducing PM emissions would have both health and climate benefits.

People living around airports or under busy flight paths are more exposed to aviation-related UFP (and aviation-related PM in general) than the general population, and so are airport workers. Their exposure depends on many factors, such as atmospheric circumstances, distance to runways and flightpaths, and fuel composition. There has been no comprehensive study of the health impacts of aviation UFP emissions in Europe (although the impacts around Schiphol have been estimated). This report presents a crude first-order estimation of what the health effects caused by aviation-relation UFP around major European airports could be. It finds that aviation UFP may possibly cause a total of nearly 280,000 cases of high blood pressure, 330,000 cases of diabetes and 18,000 cases of dementia around the 32 airports in the scope of the study, based on current population and UFP concentration levels. However, these values are a crude first-order estimate and should be confirmed by epidemiologic studies.

Aircraft UFP and PM emissions are mainly caused by the combustion of fuel, and to a smaller extent by the use of lubrication oils. For the fuel burning related emissions, the composition of the fuel impact the number of particulates emitted. In particular, the amount of emitted PM critically depends on the amount of aromatics (and all cyclic structures) in the fuel, and the sulphur content of the fuel. Aromatics are the main cause for formation of non-volatile PM (nvPM). Naphthalenes cause more UFP than single ring aromatics. The sulphur content is directly related to the formation of sulphuric acids, which in turn can both form sulphuric volatile PM (vPM) and attach to non-volatile particles. For the lubrication oil related emissions, these emissions could be reduced through the development of superior technologies for controlling oil emissions.

This means that aviation UFP and PM emissions can be reduced by reducing the concentration of aromatics and sulphur in jet fuel. There are two ways to achieve this,

namely through hydrotreatment of fossil fuels, thus saturating the aromatics and removing sulphur, and by blending fossil jet fuel with non-aromatic sustainable aviation fuels.
To ensure the widespread deployment of hydrotreated fossil jet fuel in the EU, amending the Fuel Quality Directive or the ReFuel Aviation regulation could be appropriate regulatory pathways, while on a global scale amending existing standards or developing new ones could help achieve the same goal.

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A Estimated health effects per country

Table 8 – Considered airports in this study, per country

Country	Considered airports
Belgium	Brussels
Czechia	Prague Ruzyne
Denmark	Copenhagen Kastrup
Germany	München; Frankfurt am Main; Köln; Hamburg; Düsseldorf
Ireland	Dublin
Greece	Athens Eleftherios Venizelos
Spain	Palma de Mallorca; Malaga Costa Del Sol; Barcelona El Prat; Madrid Adolfo Suarez
France	Nice Cote d'Azur; Paris Charles de Gaulle; Paris Orly
Italy	Roma Fiumicino; Milan Malpensa
Netherlands	Amsterdam Schiphol
Austria	Vienna Schwechat
Poland	Warsaw Chopin
Portugal	Lisbon
Finland	Helsinki-Vantaa
Sweden	Stockholm-Arlanda
Norway	Oslo Gardermoen
Switzerland	Geneva; Zurich
United Kingdom	London Stansted; London Gatwick; London Heathrow; Manchester

Table 9 - The estimated health effects through UFP emission, per country. These results only include the considered airports in this study (see Table 8). The share of flight activity attributed to the considered airports per country is presented as well.

Country	Total flights per country (×1000)	Considered flights in this study (×1000)	Share of flights considered	Population around airports (×1000)				Health effects (number of cases)		
				0-5 km	5-10 km	10-20 km	Total (0-20 km)	High blood pressure	Diabetes cases	Dementia
Belgium	324	223	69%	112	579	1.322	2.013	7.055	7.526	492
Czechia	151	144	95%	86	264	1.037	1.388	4.507	4.825	203
Denmark	313	263	84%	76	223	827	1.126	4.920	4.415	410
Germany	1.854	1.449	78%	635	1.870	5.283	7.788	49.587	52.691	1.975
Ireland	272	239	88%	127	373	780	1.280	3.541	7.814	594
Greece	428	220	51%	18	93	1.507	1.617	4.705	6.145	934
Spain	1.584	1.132	71%	740	2.635	4.675	8.050	52.205	64.918	5.339
France	1.322	914	69%	483	1.977	8.794	11.255	46.836	66.309	1.441
Italy	1.203	545	45%	66	315	1.236	1.617	7.140	7.280	276
Netherlands	564	509	90%	88	487	1.198	1.773	12.786	14.740	246
Austria	306	282	92%	14	49	944	1.006	4.181	3.682	270
Poland	379	191	50%	247	730	1.163	2.140	11.504	11.252	673
Portugal	401	222	55%	414	619	1.181	2.215	15.473	18.615	1.837
Finland	205	194	95%	81	272	786	1.139	5.475	6.097	928
Sweden	361	232	64%	14	21	131	166	611	677	39
Norway	432	253	59%	16	24	67	107	436	415	24
Switzerland	454	389	86%	327	720	1.031	2.078	9.428	11.122	266
United Kingdom	1.976	1.160	59%	313	1.219	4.623	6.155	40.846	44.165	2.209
Total	**12.529**	**8.560**		**3.858**	**12.468**	**36.588**	**52.914**	**281.234**	**332.687**	**18.157**