Hoge
Gezondheidsraad

DE GEZONDHEIDSEFFECTEN VAN VLIEGTUIGLAWAAI EN LUCHTVERONTREINIGING IN DE OMGEVING VAN BRUSSELS AIRPORT

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Federale Overheidsdienst Volksgezondheid, Veiligheid van de Voedselketen en Leefmilieu

Hoge Gezondheidsraad
Galileelaan 5 bus 2
B-1210 Brussel

Tel: 02/524 97 97
E-mail: info.hgr-css@health.fgov.be

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Hoge Gezondheidsraad

ADVIES VAN DE HOGE GEZONDHEIDSRAAD nr. 9741

De gezondheidseffecten van vliegtuiglawaai en luchtverontreiniging in de omgeving van Brussels Airport

In this scientific advisory report, which offers guidance to public health policy-makers, the Superior Health Council of Belgium provides an overview of the potential health burden of aircraft noise and pollutant emissions on citizens living near Brussels Airport.

This report aims at providing both authorities, citizens and the airport with specific recommendations contributing to a lower burden of disease, annoyance and sleep disturbance.

Versie gevalideerd op het College van
14 April 2024[1]

SAMENVATTING

1. Inleiding, context en reikwijdte

Dit rapport van de Belgische Hoge Gezondheidsraad (HGR) analyseert de gezondheidsimpact van de luchtvaartactiviteiten van Brussels Airport (Zaventem) op de bewoners van de dichtbevolkte omliggende gebieden. Het werd opgesteld als antwoord op een adviesaanvraag van de federale minister van Sociale Zaken en Volksgezondheid in oktober 2022.

In de loop van de 20e eeuw groeide de luchthaven uit tot het internationale knooppunt dat ze vandaag is, parallel met de sterke (sub)verstedelijking van de omgeving. Volgens Brussels Airport Traffic Control (BATC) werden er in 2019 234 461 vliegtuigbewegingen uitgevoerd (vs. 178 924 in 2022)[2], waarvan 17 347 (16 916 in 2022) tussen 23u en 6u. De meeste van deze nachtvluchten tussen 23.00 en 6.00 uur zijn vrachtvluchten (45,68 % in 2019 en 53,34 % in 2022). Deze cijfers onderschatten echter sterk de werkelijke last van het luchtverkeer 's nachts, aangezien 10 029 vluchten (7804 in 2022) plaatsvinden tussen 6 en 7 uur 's ochtends. Volgens de gerapporteerde geluidscontouren rond Brussels Airport voor het jaar 2019, woonden 163 718 inwoners binnen de geluidscontouren > 45 dB(A) L_{night} (23u - 7u) (in 2022: 151 901 inwoners). Meer dan 163 518 bewoners ondervonden meer dan 10 vliegbewegingen met een L_{Amax} van meer dan 60 dB(A) per nacht (in 2022: 123 239 bewoners). Vliegtuigen stijgen voornamelijk op van baan 25R en landen op banen 25L/25R omwille van de overheersende zuidwestenwind overdag; zwakke wind 's nachts biedt meer vrijheid in de

[1] De Raad behoudt zich het recht voor om in dit document op elk moment kleine typografische verbeteringen aan te brengen. Verbeteringen die de betekenis wijzigen, worden echter automatisch in een erratum opgenomen. In dergelijk geval wordt een nieuwe versie van het advies uitgebracht.
[2] Aangezien het luchtverkeer in 2022 nog niet was teruggekeerd naar het niveau van vóór Covid, focust dit verslag op de cijfers van 2019, omdat die een betrouwbaarder beeld geven van de gebruikelijke vliegtuigbewegingen in Zaventem. De cijfers voor 2022 worden volledigheidshalve tussen haakjes vermeld.

keuze van de baan. Door de specifieke ligging van Brussels Airport dicht bij de grens van verschillende gewesten en gemeenschappen, is het adequaat omgaan met en het verdelen van de lasten van vliegtuiglawaai al decennia lang een complexe politieke kwestie.

Dit rapport geeft een overzicht van de huidige gepubliceerde wetenschappelijke kennis, verkregen door het verzamelen van gegevens uit verschillende bestaande studies over Brussels Airport, maar ook door het identificeren van hiaten in de kennis en het houden van hoorzittingen met experts over studierapporten van luchthavens in Frankrijk (ACNUSA), Duitsland (NORAH studie) en Nederland (RIVM studie). Hoewel het verzoek om een advies gericht was op de effecten van vliegtuiglawaai, dat meestal de meeste aandacht krijgt in het debat over Brussels Airport, koos de HGR voor een bredere analyse door ook de effecten van luchtvervuiling gerelateerd aan vliegtuigbewegingen mee te nemen. Luchtverontreiniging is één van de grootste gekende milieurisico's voor de menselijke gezondheid en leidt tot een aanzienlijke toename van het aantal ziekten, vooral hart- en vaatziekten en verschillende soorten kanker, vroegtijdige sterfgevallen en voor meer verlies van gezonde levensjaren (uitgedrukt als *Disability Adjusted Life Years* of DALY's). Negatieve gezondheidseffecten worden niet alleen veroorzaakt door enkelvoudige factoren, maar worden vaak versterkt door de interferentie tussen meerdere factoren. Maatregelen die worden genomen om negatieve invloeden tegen te gaan, moeten daarom rekening houden met deze interferenties. Daarnaast erkent de HGR dat emissies van de luchtvaart ook bijdragen aan wereldwijde milieuproblemen zoals klimaatverandering en stikstofdepositie en roept daarom op tot een bredere reflectie over duurzame luchtvaart.

Tot slot geeft de HGR aanbevelingen voor verdere acties om de bescherming van de gezondheid van omwonenden van de luchthaven van Brussel te verbeteren. Andere belangen (bv. economische en financiële) komen niet aan bod.

2. Vragen

Deze samenvatting geeft een overzicht van de antwoorden van de HGR op de vragen die in de adviesaanvraag werden gesteld. Meer informatie en referenties zijn te vinden in het volledige rapport.

2.1 Wat zijn de directe en indirecte gevolgen voor volksgezondheid van het door vliegtuigen veroorzaakt omgevingslawaai (zowel inzake geluidsniveau als vliegfrequentie) en de luchtverontreinigende emissies van vliegtuigen in de ruime regio van de luchthaven?

Het bestuderen van schadelijke gezondheidseffecten (directe en vooral indirecte) is ingewikkeld en vereist complexe, objectieve epidemiologische studies die statistisch zoveel mogelijk gecorrigeerd zijn voor mogelijke *confounders*. **Een "associatie" tussen blootstelling en een bepaald effect impliceert niet noodzakelijkerwijs een "oorzakelijk verband".** Om een causaal verband vast te stellen, moeten enkele criteria worden gecontroleerd: (1) consistentie van de associatie in verschillende onderzoeken met verschillende methoden; (2) sterkte van de associatie; (3) specificiteit van de associatie tussen één oorzaak en het effect; (4) het bestaan van een temporele relatie; (5) coherentie van de associatie met bestaande kennis (en bij voorkeur het bestaan van een mechanistische relatie); en (6) het bestaan van een dosis-respons relatie. De door de HGR bestudeerde studies en systematische reviews hielden zoveel mogelijk rekening met deze criteria en met inclusie van *confounders*.

2.1.1. Vliegtuiglawaai

Overmatig vliegtuiglawaai is schadelijk voor de gezondheid en het welzijn. In haar "*Environmental Noise Guidelines*"[3] uit 2018 beveelt de Wereldgezondheidsorganisatie (WHO) ten zeerste aan om de door vliegtuigen geproduceerde geluidsniveaus te beperken tot minder dan 45 dB(A) L_{den} en 40 dB(A) L_{night}, aangezien vliegtuiglawaai boven deze beide grenswaarden in verband wordt gebracht met nadelige gezondheidseffecten. Dit geldt met name voor de tweede grenswaarde, gezien de negatieve effecten van nachtlawaai op de slaap. Merk op dat deze aanbevelingen betrekking hebben op geluidsniveaus buitenshuis, d.w.z. aan de meest blootgestelde gevel van de woningen.

Lawaai kan in verband worden gebracht met verschillende gezondheidseffecten:

a) Internationaal gestandaardiseerde scoringsmethoden laten zien dat hoge geluidsbelasting gepaard gaat met zelfgerapporteerde **ernstige hinder (*High Annoyance HA*)**, die cognitieve, emotionele en gedragsaspecten omvat en beschouwd wordt als een **vroegtijdige waarschuwing voor nadelige gezondheidseffecten.** Meer specifiek wordt gerapporteerde ernstige hinder in verband gebracht met een verhoogd risico op mentale gezondheidsproblemen, waaronder depressie en angst. Bewoners met een hoge hinderscore hebben significant hogere niveaus van fysiologische stress, wat een risicofactor is voor hypertensie. De WHO-beoordeling concludeerde dat bij blootstelling aan 40 dB(A) L_{den} 1,2 % van de populatie ernstige hinder aangeeft. Bij 50 dB(A) (L_{den}) stijgt de prevalentie van HA tot 17,9 % en bij 60 dB(A) (L_{den}) tot 36 % HA. Bij hetzelfde gemiddelde geluidsniveau (L_{den}) rapporteert een groter deel van de bevolking ernstige hinder door vliegtuiglawaai dan door wegverkeer. Dit is waarschijnlijk te wijten aan de hogere intermittentie van vliegtuiglawaai en de aard van het geluidspatroon.

b) **Slaapverstoring** is het **belangrijkste nadelige gevolg voor de gezondheid** van nachtelijk vliegtuiglawaai. Slaap vervult een essentiële fysiologische functie die niet kan worden vervangen (geheugenconsolidatie, belangrijke immunologische en endocriene processen). Het herstellend vermogen ervan wordt bepaald door de slaapduur en de slaapkwaliteit, die beide worden beïnvloed door de intensiteit van het lawaai, het aantal vluchten en de spreiding in de tijd. Slaapverstoring wordt veroorzaakt door ontwaken, opwinding en slaapfaseveranderingen. Een korte slaapduur en een slechte slaapkwaliteit hebben zowel korte termijngevolgen (slechte prestaties overdag en prikkelbaarheid) als lange termijngevolgen (bijdragen aan chronische aandoeningen zoals obesitas, diabetes type 2, hart- en vaatziekten, chronische pijn, chronisch vermoeidheidssyndroom, neurodegeneratieve ziekten, depressie en mogelijk, indirect, borstkanker). In longitudinale studies werden zelfgerapporteerde slaapstoornissen onlangs in verband gebracht met een verhoogd risico op kanker. Objectief gemeten (met actigrafie) korte slaapduur en slechte slaapkwaliteit werden ook in verband gebracht met meer cardiovasculaire ziekten en kankersterfte. Polysomnografische studies, die objectief parameters van de slaapfysiologie meten, toonden nadelige effecten aan van nachtelijk vliegtuiglawaai op de slaapduur en -kwaliteit. Dit ondersteunt de associaties tussen zelfgerapporteerde slaapstoornissen en nachtelijk vliegtuiglawaai. De WHO-beoordeling concludeerde dat 40 dB(A) L_{night} geassocieerd wordt met ernstige slaapverstoring (HSD) bij 11,3 % van de deelnemers aan het onderzoek. Bij 50 dB(A) (L_{night}) namen deze cijfers toe tot 19,7 % en bij 60 dB(A) (L_{night}) tot 32,3 % HSD. Cruciaal is echter dat recent onderzoek heeft aangetoond dat de frequentie van overvluchten van het grootste belang is voor de kwaliteit van de slaap. Kinderen vormen hierbij een meer kwetsbare groep, omdat hun

[3] https://www.who.int/europe/publications/i/item/9789289053563 (geraadpleegd op 22/1/2024)

cognitieve en fysieke ontwikkeling meer slaap vereist. Voor volwassenen is over het algemeen een kwaliteitsvolle slaap van 8 uur ideaal.

Als gevolg van deze stressfactoren kunnen **indirect** de volgende negatieve effecten optreden:

a) **Cognitieve stoornissen**. Er werd aangetoond dat vliegtuiglawaai negatief samenhangt met de cognitieve ontwikkeling van **kinderen**. Een cross-sectionele studie (NORAH) onder >1000 leerlingen uit het tweede leerjaar van 29 basisscholen rond de luchthaven Frankfurt/Main toonde aan dat er een lineair verband was tussen toenemende geluidsblootstelling en drie verschillende indicatoren: beoordeling van levenskwaliteit, geluidshinder en leesprestaties. Terwijl de blootstelling aan lawaai minder dan 60 dB(A) ($L_{Aeq,\ 8\text{-}14}$) was in alle onderzochte scholen, werd een toename van 10 dB(A) en 20 dB(A) in vergelijking met de scholen met de laagste blootstelling (39 tot 46 dB(A) $L_{Aeq,\ 8\text{-}14}$) in verband gebracht met respectievelijk een vertraging van 1 en 2 maanden in de leesvaardigheid.

b) **Hypertensie**. Meerdere studies, zoals de HYENA-cohortstudie rond zes Europese luchthavens, vonden dat hypertensie vaker optrad bij hogere blootstelling aan vliegtuiglawaai. Terwijl volgens de WHO-evaluatie van 2018 de incidentie van hypertensie per toename met 10 dB(A) L_{den} niet significant verhoogd bleek te zijn, heeft een meer recent longitudinaal onderzoek (DEBATS in Frankrijk) het verband bevestigd tussen vliegtuiglawaai en meer voorkomende gevallen van hypertensie evenals nieuwe gevallen van hypertensie.

c) **Cardiovasculaire aandoeningen** werden in meerdere onderzoeken in verband gebracht met vliegtuiglawaai via stress en slaaptekort. Volgens de WHO is de verhoogde risicoratio voor de incidentie van ischemische hartziekten (IHC) 9 % (95 % CI: 4-15 %) per toename met 10 dB(A) (L_{den}). Op basis van verzekeringsclaims van meer dan een miljoen inwoners ouder dan 40 jaar associeerde de NORAH-studie geluidsniveaus van vliegtuigen boven 60 dB(A) ($L_{Aeq,\ 24\ uur}$) met een hoger risico op myocardinfarct. Het totale risico van weglawaai en spoorweglawaai werd echter hoger geschat. Daarnaast bleek uit het NORAH-onderzoek dat verzekerden die blootgesteld waren aan laag achtergrondgeluid ($L_{Aeq,24uur}$ < 40 dB(A)) maar L_{Amax} > 50 dB(A) een verhoogd risico van 7 % voor beroerte en 6 % voor hartfalen vertoonden. Onderzoek heeft ook een verband aangetoond tussen vliegtuiglawaai en een hoger sterfterisico door myocardinfarct dat onafhankelijk is van luchtvervuiling en sociaaleconomische factoren.

d) **Geestelijke gezondheid en depressie**. Geluidshinder door vliegtuigen wordt in verband gebracht met geestelijke gezondheidsproblemen. In het NORAH-onderzoek werd een omgekeerd U-vormig verband gevonden tussen vliegtuiglawaai en depressie. Het risico op depressie nam toe met 8,9% bij een stijging van 10 dB(A) $L_{Aeq,24\ uur}$, maar er werd een afname genoteerd bij hogere geluidsniveaus, waarschijnlijk door een soort gewenning. Een alternatieve verklaring zou kunnen zijn dat er over het algemeen hogere geluidsniveaus worden waargenomen dichter bij de luchthaven, waar mogelijk meer mensen wonen die professioneel betrokken zijn bij luchthavenactiviteiten.

e) **Andere effecten**. Sommige studies suggereren een mogelijk verband tussen blootstelling aan vliegtuiglawaai en bijvoorbeeld de incidentie van borstkanker of ongunstige geboorte-uitkomsten. Grootschalige epidemiologische studies hebben geconcludeerd dat blootstelling aan lawaai van wegverkeer in verband kan worden gebracht met stofwisselingsziekten, waaronder obesitas en diabetes type II. De interpretatie van de observationele studies wordt echter vaak bemoeilijkt door de meervoudige *confounders* waarvoor de nodige informatie niet altijd beschikbaar was.

2.1.2. Luchtverontreinigende stoffen

De verbranding van vliegtuigbrandstof resulteert in de uitstoot van verschillende schadelijke verontreinigende stoffen, waaronder fijnstof (waaronder ultrafijn stof, UFP), vluchtige organische stoffen (VOC's), polycyclische aromatische koolwaterstoffen (PAK's), organofosfaatesters (OPE's), metalen, stikstofoxiden, enz. Het is al vele jaren bekend dat deze luchtverontreinigende stoffen een grote invloed kunnen hebben op de menselijke gezondheid, vooral op hart- en vaatziekten, chronische obstructieve longziekten en verschillende soorten kanker (zie ook adviesrapport nr. 9404 van de Hoge Gezondheidsraad). Het documenteren van deze effecten in epidemiologische studies in de omgeving van luchthavens is echter erg moeilijk, omdat er veel andere bronnen van vervuiling aanwezig zijn en omdat de menselijke epidemiologie niet over de nodige gevoeligheid en onderscheidend vermogen beschikt. Moleculaire epidemiologische studies en studies die fysiologische functies meten zouden een meer mechanistisch gebaseerd beeld van de gezondheidseffecten kunnen geven. Alleen UFP werd uitgebreid gemeten en gemodelleerd in de buurt van Brussels Airport door VITO. De grootste hoeveelheid UFP werd gevonden net benedenwinds van de luchthaven in Steenokkerzeel (gemiddeld 10-100 nm, ca. 15 900 partikels(pt)/cm³). Het UFP verspreidt en verdunt naar het noordoosten door de overheersende zuidwestenwind (gemiddeld 10-100 nm Kampenhout, ca. 6600 pt/cm³). Aangezien de deeltjesdiameter van vliegtuig-UFP overwegend kleiner is dan 20 nm, resulteert het hoge specifieke oppervlak van het UFP in adsorptie van meer toxische stoffen (waaronder kankerverwekkende PAK's) in vergelijking met grotere deeltjes. Deze schadelijke verbindingen komen terecht in de meest distale longgebieden. Terwijl UFP steeds meer aandacht krijgt in onderzoek, zijn er maar weinig studies beschikbaar over de gezondheidseffecten van alle luchtvaartemissies samen. In Nederland zijn verschillende relevante studies uitgevoerd in de buurt van de luchthaven van Schiphol. Een 5-jarig onderzoeksprogramma van het RIVM nabij Schiphol (algemeen rapport gepubliceerd in 2022) maakte onderscheid tussen acute en chronische gezondheidseffecten geassocieerd met blootstelling aan UFP afkomstig van de uitstoot door vliegtuigen:

a) **Cardiovasculair systeem**. Er zijn indicaties voor een associatie met zowel chronische als acute effecten op het cardiovasculaire systeem. Er werden positieve associaties gevonden met de incidentie van medicijngebruik voor hartaandoeningen en sterfte door hartritmestoornissen. In enquêtes werden associaties gevonden met meer zelfgerapporteerde beroertes en hartaanvallen. In experimenten met kortdurende blootstelling van volwassenen werd een verlenging van het QTc-interval (ECG) waargenomen.

b) **Hypertensie**. De onderzoeken (Gezondheidsmonitor) vonden een positief verband met hoge bloeddruk. Daarentegen werd er geen positief verband gevonden met het gebruik van medicijnen tegen deze aandoening.

c) **Ongunstige geboorte-uitkomsten**. Verschillende studies vonden verbanden tussen blootstelling aan UFP in de buurt van luchthavens en vroeggeboorte, laag geboortegewicht en geboorteafwijkingen.

d) **Aandoeningen van de luchtwegen**. Er werden geen associaties gevonden tussen langdurige blootstelling aan UFP uitgestoten door vliegtuigen en aandoeningen van de luchtwegen (mortaliteit: totale luchtwegaandoeningen, COPD, longkanker; morbiditeit: astma). Daarentegen werd kortdurende blootstelling in verband gebracht met meer luchtwegklachten bij kinderen en meer gebruik van bronchodilatoren. Bij volwassenen werd een afname van de longfunctie (FVC, *forced vital capacity*) waargenomen.

e) Er werden geen duidelijke/consistente verbanden gevonden tussen blootstelling aan UFP afkomstig van vliegtuigen en de algemene gezondheid (all-cause mortaliteit, zelf ervaren gezondheid), neurologische effecten of het metabolisme.

Deze verschillende effecten komen ook in meer of mindere mate voor in de brede literatuur over de effecten van UFP op de menselijke gezondheid. De resultaten van de RIVM-studie geven een idee van de mogelijke effecten van UFP geassocieerd met uitstoot van vliegtuigen, maar kunnen niet zomaar geëxtrapoleerd worden van Schiphol naar Brussels Airport, aangezien de geografische context erg verschillend is (d.w.z. een veel hogere bevolkingsdichtheid in de onmiddellijke omgeving van Brussels Airport dan in Schiphol). Bovendien is het niet eenvoudig om een onderscheid te maken tussen gezondheidseffecten van UFP die afkomstig zijn van de luchtvaart of van het wegverkeer in de omgeving. Een specifieke studie is nodig in de context van Brussels Airport.

Verschillende andere vervuilende stoffen worden niet specifiek gemeten rond luchthavens of specifiek in verband gebracht met luchthavenemissies. Dit wil niet zeggen dat ze niet aanwezig zijn in schadelijke concentraties of niet op elkaar kunnen inwerken, aangezien ze samen voorkomen en de blootstelling meervoudig is (verschillende luchtvervuilende stoffen en lawaai).

Meerdere studies associëren een verhoogd risico op kanker met luchthavengerelateerde blootstelling aan UFP, $PM_{2.5}$ en NO_2, waaronder kwaadaardige hersentumoren. In 2005 concludeerde een groot ecologisch bevolkingsonderzoek in de buurt van Schiphol echter dat de totale incidentie van kanker in de buurt van Schiphol vergelijkbaar was met de landelijke incidentie. Hoewel de incidentie van hematologische kwaadaardige aandoeningen significant verhoogd was, kon er geen specifieke relatie met de luchtkwaliteit worden vastgesteld.

De vraag kan worden gesteld of de aandacht voor door vliegtuigen veroorzaakte luchtverontreiniging (en geluidsoverlast) niet moet worden verbreed naar andere aspecten van luchthavenactiviteiten. Luchtvervuiling door vliegtuigen is inderdaad een probleem op zich, maar het effect van de luchthaven op de luchtkwaliteit is veel groter omdat meer vluchten en vooral meer vrachtactiviteit zullen leiden tot een aanzienlijke toename van het wegverkeer, zowel 's nachts als overdag, met zware vrachtwagens in een dichtbevolkt gebied dat al te kampen heeft met verzadigde verkeersomstandigheden en de daarmee gepaard gaande geluidsoverlast en luchtvervuiling. Hetzelfde geldt voor de geluidshinder, zeker gezien de ambitie van Brussels Airport om het vrachtverkeer meer dan te verdubbelen tegen 2032. Op basis van het Milieueffectrapport (MER) valt te verwachten dat het aantal vrachtwagens in 2032 met 43 % zal toenemen ten opzichte van 2019. Recente gegevens en vooruitzichten zijn te vinden in het MER-rapport dat in december 2023 openbaar is gemaakt.

2.2 Is er een verschil in effecten met betrekking tot vluchten overdag, in de vroege ochtend en 's nachts?

2.2.1. Vliegtuiglawaai

Op basis van een uitgebreide literatuurstudie en het advies van haar deskundigen concludeert de HGR dat niet alleen de gemiddelde geluidsniveaus, maar vooral het aantal vliegbewegingen waarbij het geluidsniveau een bepaalde drempel overschrijdt, een aanzienlijke invloed hebben op het type en de ernst van de gezondheidseffecten van vliegtuiglawaai.

Gezien het belang van een voldoende goede slaapkwaliteit voor zowel fysieke als mentale gezondheid, genereert blootstelling aan vliegtuiglawaai 's nachts de meest schadelijke effecten. Zowel zelfrapportering als polysomnografische studies hebben aangetoond dat dit resulteert in een grotere kans op ontwaken, meer tijd tot het begin van de slaap en meer slaapstoornissen. Cruciaal is dat stijgende nachtelijke vliegfrequenties in verband worden gebracht met een statistisch significante toename van het aantal gehinderde bewoners. De akoestische nacht, d.w.z. de periode waarin het L_{night}-niveau wordt gemeten, begint om 23.00 uur en eindigt om 7.00 uur. De nachtperiode op het vliegveld, zoals gedefinieerd door de federale overheid en waarin vliegbeperkingen gelden, begint echter om 23.00 uur en eindigt om 6.00 uur. De HGR wil het belang van dit hiaat van één uur benadrukken. Omdat de slaap in de vroege ochtend minder diep is, hebben vluchten tijdens dat tijdstip een aanzienlijk effect op de perceptie van de slaapkwaliteit, aangezien slapers gemakkelijker wakker worden, maar ook op objectieve slaapparameters. De NORAH-studie toonde aan dat het verbod op nachtvluchten in Frankfurt weliswaar resulteerde in minder, door vliegtuiglawaai veroorzaakt, ontwaken tot de vroege ochtend (het verbod op nachtvluchten eindigt in Frankfurt om 5 uur 's ochtends), maar dat de hogere concentratie vluchten in de vroege ochtend (5 tot 7 uur 's ochtends) overeenkwam met de perceptie van een slechtere slaapkwaliteit. Terwijl vluchten in de avond geen invloed leken te hebben op de wachttijd bij het begin van de slaap in Frankfurt, nam deze wachttijd wel significant toe rond de luchthaven van Keulen-Bonn, waar geen verbod op nachtvluchten geldt.

Naast L_{night} moet vermeld worden dat de blootstellingsfrequentie een dominante invloed heeft op de objectief gemeten slaapkwaliteit via polysomnografie. Om de blootstellingsfrequentie goed te karakteriseren moeten meer gekwalificeerde indicatoren voor enkelvoudige geluidsgebeurtenissen (L_{Amax} of SEL) gebruikt worden.

Zelfs terwijl blootstelling 's nachts het allerbelangrijkst is, werd blootstelling aan vliegtuiglawaai overdag ook in verband gebracht met gezondheidseffecten, waaronder ernstige hinder en leerproblemen bij schoolkinderen.

2.2.2. Luchtverontreinigende stoffen

De concentratie van luchtverontreinigende stoffen in de atmosferische grenslaag (onderste troposfeer) wordt beïnvloed door zowel de weersomstandigheden als het tijdstip van de dag. De verspreiding van verontreinigende stoffen wordt afgedwongen door de onstabiele toestand van de lucht als gevolg van advectie (wind: horizontaal transport, hoewel gemedieerd door de vorm van de bebouwde omgeving) en convectie, die atmosferische turbulentie veroorzaakt.

Er is sprake van een stabiele toestand als de lucht dicht bij de grond kouder is dan de hogere lucht. Dit creëert een temperatuurinversie waarbij koude lucht dicht bij de grond wordt ingesloten door een laag warmere lucht. Hierdoor wordt de verspreiding en verdunning van verontreinigende stoffen beperkt. Verontreinigende stoffen kunnen dicht bij de grond gevangen blijven, wat leidt tot een opeenhoping van verontreiniging rond de emissiebron. Een grote stabiliteit wordt waargenomen kort na zonsopgang (vroege ochtend) en kort voor zonsondergang (late namiddag, vroege avond). Het is daarom niet aan te raden om de uitstoot van verontreinigende stoffen (zoals UFP) op deze momenten van de dag te concentreren.

2.3 Is er in de internationale wetenschappelijke literatuur sprake van een evolutie in de inschatting van deze effecten en bestaan er goede onderzoeken op dit vlak in de omgeving van vergelijkbare West-Europese luchthavens waarvan de methodologie in ons land bruikbaar zou kunnen zijn?

De meeste studies focussen op blootstelling aan lawaai. Luchtvervuiling mag echter niet vergeten worden. Bovendien zijn er maar weinig studies die het gecombineerde effect van luchtvervuiling en blootstelling aan lawaai behandelen.

2.3.1. *Vliegtuiglawaai*

Er bestaat uitgebreide literatuur over de effecten van vliegtuiglawaai op de menselijke gezondheid. Verschillende recente studies hebben vliegtuiglawaai in verband gebracht met (ernstige) hinder en slaapverstoring, maar ook met cognitieve stoornissen, hoge bloeddruk en hart- en vaatziekten (zie hierboven).

In Europa uitgevoerde onderzoeken zijn onder andere het DEBATS-onderzoek (Frankrijk), het NORAH-onderzoek (Duitsland) of het HYENA-onderzoek (EU). De HGR-deskundigengroep heeft hoorzittingen gehouden met vertegenwoordigers van de twee eerstgenoemde onderzoeken.

De DEBATS-studie werd opgezet door de Franse autoriteit voor de controle op luchthavenoverlast (ACNUSA) om de mogelijke effecten van vliegtuiglawaai op de slaap in de omgeving van 3 luchthavens (Parijs Charles-De-Gaulle, Lyon-Saint-Exupéry, Toulouse-Blagnac) te beoordelen, waarbij drie soorten methodologische benaderingen werden gebruikt:

a) Een ecologische studie, gericht op het relateren van geaggregeerde gezondheidsindicatoren aan de gewogen gemiddelde blootstelling.

b) Een individueel longitudinaal onderzoek: personen werden gedurende minstens 4 jaar gevolgd met herhaalde metingen van hun gezondheidstoestand (vragenlijsten, metingen van bloeddruk, hartslag en speekselcortisol).

c) Een slaapstudie uitgevoerd bij deelnemers aan de longitudinale studie met als doel een gedetailleerde en specifieke beschrijving te krijgen van de acute effecten van vliegtuiglawaai op de kwaliteit van de slaap. Deze studie werd uitgevoerd in samenwerking met *Bruitparif,* dat betrokken was bij de instrumentatie van de deelnemers thuis (akoestische metingen, actimetrische bewegingsmetingen van de ledematen, hartslag).

De NORAH-studie werd uitgevoerd over een periode van 3 jaar (2011, 2012, 2013) en richtte zich op de gezondheidseffecten van vliegtuiglawaai rond de luchthaven van Frankfurt voor en na de opening van een nieuwe landingsbaan en de invoering van een verbod op nachtvluchten:

a) Geluidshinder werd voornamelijk onderzocht door middel van online en telefonische enquêtes. De studie onderzocht ook of de invoering van het verbod op nachtvluchten leidde tot veranderingen in geluidshinder door vliegtuigen. Er werd gekeken naar de gecombineerde effecten van vliegtuig- en spoorweglawaai enerzijds en vliegtuig- en verkeerslawaai anderzijds en de blootstellings-responsrelaties werden onderzocht. Daartoe werd de blootstelling aan lawaai van vliegtuigen, spoorwegen en wegverkeer berekend voor elke deelnemer op zijn/haar thuisadres in het jaar voorafgaand aan het onderzoek.

b) Slaapverstoring werd zowel objectief gemeten door middel van polysomnografie en ECG + actigrafie, als subjectief door middel van vragenlijsten. De geluidsdrukniveaus en individuele geluidsgebeurtenissen werden continu gemeten aan het oor van de slaper. De studie vergeleek de resultaten van twee bedtijdgroepen (de ene groep blootgesteld aan vliegtuiglawaai 's avonds en 's ochtends vroeg, de andere alleen 's ochtends vroeg) met die van een veldstudie die werd uitgevoerd in de buurt van de luchthaven Keulen-Bonn, waar geen verbod op nachtvluchten gold.

c) Wat hart- en vaatziekten betreft, werden in het onderzoek gegevens van drie ziekenfondsen gebruikt en gekoppeld aan de huidige en vroegere blootstelling aan lawaai.

d) De effecten op de cognitie van kinderen en de gezondheidsgerelateerde levenskwaliteit werden onderzocht door te kijken naar drie indicatoren: leesvaardigheid en verwante vaardigheden bij kinderen, zelfgerapporteerde levenskwaliteit en zelfgerapporteerde hinder. De focus lag op kinderen in het tweede leerjaar (7-9 jaar) in scholen rond de luchthaven van Frankfurt/Main. De blootstelling aan lawaai op de scholen en thuis werd voor elk kind berekend met behulp van radargegevens van het *Flight Track and Aircraft Noise Monitoring System* (FANOMOS) voor een periode van 12 maanden voorafgaand aan de gegevensverzameling. Scholen werden in verschillende groepen verdeeld, afhankelijk van de geluidsniveaus die overdag heersten, en er werd speciaal op gelet dat deze groepen vergelijkbaar waren (migratieachtergrond, sociaaleconomische status, enz.). Metingen werden verkregen door middel van testen op school (lezen, non-verbale intelligentie, enz.) en vragenlijsten.

2.3.2. Luchtverontreinigende stoffen

Verrassend genoeg is de impact van straalvliegtuigemissies op de gezondheid van omwonenden van internationale luchthavens niet uitgebreid bestudeerd in de huidige internationale literatuur. Een veelbelovende, baanbrekende studie werd onlangs uitgevoerd in de buurt van de luchthaven Schiphol door het Nederlandse RIVM (2022), waarbij gebruik werd gemaakt van een multidisciplinaire aanpak. Hun methodologie was drieledig:

a) Meten en modelleren van lange termijn UFP-concentraties.
Het verspreidingsmodel bleek geschikt voor toepassing in epidemiologische studies naar effecten van langdurige blootstelling.

b) Studies naar gezondheidseffecten van kortdurende blootstelling aan UFP.
Er werden drie studies uitgevoerd:
   - Een panelstudie met basisschoolkinderen in woonwijken met reële UFP-concentraties.
   - Een vrijwilligersstudie met gezonde volwassenen die blootgesteld werden aan hoge UFP-concentraties in een mobiel laboratorium naast de luchthaven (metingen van longfunctie, uitgeademd NO, ECG, bloeddruk, zuurstofsaturatie).
   - Een toxicologische studie (in vitro) met menselijke bronchiale epitheelcellen.

c) Studies naar gezondheidseffecten van langdurige blootstelling aan UFP. Gemodelleerde UFP-concentraties werden gekoppeld aan gegevens van de blootgestelde bewoners, met behulp van bestaande gezondheidsregisters en enquêtes.

Statistische analyses corrigeerden voor de effecten van lawaai om de effecten van UFP niet te overschatten. Op basis van een gelijkaardige methodologie zou een interessante vergelijkende studie rond Brussels Airport kunnen worden opgezet.

2.4 Welke weerslag hebben deze effecten op de budgetten en de organisatie van de gezondheidszorg?

Wat de organisatie van de gezondheidszorg betreft, kan verwacht worden dat een vermindering van de vervuiling en het lawaai rond Brussels Airport de last op morbiditeit en mortaliteit, maar zeker ook op huisartsen en ziekenhuizen (ernstige pathologieën) zal verminderen. Op basis van evenwichtige dosis-respons curves en *disability-adjusted life years*

(DALY's) voor zowel geluidshinder als UFP, zou het Federaal Kenniscentrum - *Centre federal d'expertise* (KCE) de impact op het gezondheidszorgbudget moeten kwantificeren. Deze berekening valt buiten de opdracht van de Hoge Gezondheidsraad en dit rapport kan enkel gerelateerde informatie documenteren die werd verkregen in de literatuur of via de bovenvermelde hoorzittingen.

In het algemeen zou elke studie dezelfde methodologie moeten volgen: (1) evalueer de blootgestelde populatie, (2) evalueer de fractie van die populatie die gezondheidseffecten zal ondervinden, (3) evalueer het aantal verloren jaren met goede gezondheid (ziekte en sterfte) en (4) ken een monetaire waarde toe aan elk verloren jaar. Elk van deze stappen kan worden bekritiseerd en kan worden verfijnd door middel van nieuw onderzoek, maar dit neemt niet weg dat, over het geheel genomen, elk onderzoek een aanzienlijke gezondheidseconomische kostenpost toekent aan het luchtverkeer in de buurt van luchthavens.

In dit verband merkt de HGR op dat de blootstellings-responsfuncties die voor deze berekeningen werden gebruikt, gebaseerd zijn op systematische reviews van individuele studies zoals die welke in het WHO-rapport van 2018 worden genoemd, waarbij de kwaliteit van de individuele studies werd beoordeeld volgens de GRADE-richtsnoeren, rekening houdend met de onderzoeksopzet, het aantal deelnemers en de opgenomen *confounders*. Hoewel de meeste studies rekening hielden met een reeks *confounders*, waren er gevallen waarin deze informatie niet beschikbaar was, waardoor de bewijskracht mogelijk werd verzwakt. In studies naar geluidshinder kunnen niet-akoestische factoren[4] tot 33 % van de variantie verklaren. Meer in het bijzonder beoordeelden de WHO-deskundigen de associatie van L_{night} met slaapverstoring en L_{den} met ernstige hinder, leesvaardigheid en mondeling begrip bij kinderen als van matige kwaliteit.

2.4.1. Vliegtuiglawaai

Er zijn studies naar de gezondheidskosten van blootstelling aan lawaai in termen van financiële impact, terwijl andere studies de effecten van maatregelen om de blootstelling te verminderen onderzoeken.

In Frankrijk worden de totale sociale kosten van lawaai geschat op 147 miljard euro per jaar, gebaseerd op bestaande gegevens en studies (ADEME-rapport). Daarvan zijn 97,8 miljard euro transportgerelateerd, waarvan 6,1 miljard te wijten is aan vliegtuiglawaai.

In België werd in 2023 in opdracht van "Bond Beter Leefmilieu" een korte studie uitgevoerd door een Frans adviesbureau, ENVISA, om de gezondheidseconomische impact van vliegtuiglawaai op de omwonenden van de luchthaven van Brussel te beoordelen. De auteurs gebruikten dezelfde methodologie als voor een studie die in 2021 werd uitgevoerd door Bruitparif in Île de France (Coût social du bruit en Île de France) en hun resultaten komen overeen met die van deze laatste. Met geluidscontouren van 2019 voor L_{den} 45 dB(A) en L_{night} 40 dB(A), volgens de aanbevelingen van de WHO-richtsnoeren van 2018, werd berekend dat 220 000 mensen ernstige hinder ondervinden, wat neerkomt op 4380 DALY's en een gezondheidseconomische kost van 578 miljoen euro per jaar. Ernstige slaapverstoring bij 109 999 mensen komt overeen met 7630 DALY's en een bijbehorende kostenpost van 1 miljard euro per jaar. Daarnaast geeft blootstelling aan lawaai 2000 mensen een verhoogd risico op ischemische hartaandoeningen (evidence base van zeer lage kwaliteit, WHO 2018) en 51 000 mensen een verhoogd risico op hypertensie (evidence base van lage kwaliteit, WHO 2018),

[4] Mogelijke niet-akoestische *confounders* in geluidsstudies zijn: geslacht, leeftijd, opleiding, subjectieve geluidsgevoeligheid, extraversie/introversie, algemene stressscore, co-morbiditeit, verblijfsduur, verblijfsduur in de woning overdag, oriëntatie van de vensters in een slaapkamer of woonkamer naar de straat toe, persoonlijke beoordeling van de bron, houding ten opzichte van de geluidsbron, incasseringsvermogen met betrekking tot lawaai, perceptie van wanpraktijken door de verantwoordelijke autoriteiten, lichaamsmassa-index, rookgewoonten.

wat kan overeenkomen met 6.800 DALY's en een potentiële gezondheidseconomische kostprijs van bijna 900 miljoen euro per jaar.

2.4.2. Luchtverontreinigende stoffen

Op dit moment zijn er geen gegevens beschikbaar. Deze berekening kan pas gemaakt worden nadat de blootstelling en de ziektelast voor de omwonenden van Brussels Airport beter bestudeerd zijn. Het is te verwachten dat enkel moleculair epidemiologische studies, waarin ook fysiologische functies gemeten worden en waarin associaties met specifieke interne blootstellingen bestudeerd worden, betrouwbare inzichten kunnen opleveren. Het zou echter wel mogelijk moeten zijn om de gezondheidseconomische kosten te berekenen van luchtvervuiling door wegverkeer veroorzaakt door vrachtwagens van en naar de luchthaven.

2.5 Welke beleidsaanbevelingen zijn er in het kader van dit dossier?

a) De blootstelling aan vliegtuiglawaai verminderen

- Gezien het substantiële bewijs voor (ernstige) **negatieve gezondheidseffecten**, die voornamelijk gerelateerd zijn aan slaapverstoring, is de HGR van mening dat een **volledig verbod op nachtvluchten** tussen 23.00 en 07.00 uur vanuit gezondheidsperspectief het meest wenselijk is om het welzijn van de ongeveer 163 518 inwoners binnen de L_{night} > 45 dB(A) geluidscontouren van 2019 te beschermen. Deze maatregel moet de omwonenden van de luchthaven ten minste **7 uur, idealiter 8 uur, laten slapen, ongestoord door vliegtuiglawaai.** Bovendien moet bijzondere aandacht worden besteed aan het vermijden van een hoge concentratie van vluchten in de randuren 's ochtends vroeg en 's avonds laat.

- Het is ook belangrijk eraan te herinneren dat de gewesten verantwoordelijk zijn voor de ruimtelijke ordening. Hieruit volgt dat zowel het Brussels Hoofdstedelijk Gewest als Vlaanderen een einde moeten maken aan verdere verstedelijking voor woondoeleinden in de betrokken gebieden, in tegenstelling tot de huidige praktijk.

- De vliegroutes moeten zo worden uitgelijnd dat niemand onaanvaardbare hinder ondervindt van het aantal overschrijdingen van de drempel van 60 dB(A) L_{Amax}, vooral 's nachts. In overeenstemming met dit concept (d.w.z. het primaire belang van zowel de piekintensiteit (L_{Amax}/SEL) als het aantal blootstellingen) moet het met deze blootstellingsparameters gerelateerd aantal personen dat verstoord wordt in hun slaap en het aantal personen dat gehinderd wordt zo laag mogelijk worden gehouden. Niet alleen mag niemand aan een onaanvaardbaar blootstellingsniveau worden blootgesteld, maar er moet ook voor worden gezorgd dat het aantal personen dat ernstig wordt gehinderd zo laag mogelijk blijft.

- Een uitbreiding van de luchthaven met als doel een toename van het aantal vluchten is niet aanvaardbaar gezien de huidige hoge belasting van de omwonenden in termen van luchtvervuiling en blootstelling aan lawaai.

- In het licht van het groeiende bewijs dat chronisch vliegtuiglawaai de cognitie en het leervermogen van kinderen aantast, is de HGR van mening dat zowel L_{Aeq} als het aantal dagelijkse overvluchten boven de 60 dB(A)-drempel waaraan **schoolkinderen** worden blootgesteld, moet worden verminderd. Het is twijfelachtig of het geluiddicht maken van scholen zou bijdragen aan het verminderen van het lawaai waaraan kinderen worden blootgesteld, terwijl het uitvoeren van deze maatregel met zich mee zou brengen dat er bijzondere zorg moet worden besteed aan het waarborgen van voldoende ventilatie.

- Hetzelfde geldt voor het geluiddicht maken van slaapkamers: dit is niet realistisch en kan niet worden gerechtvaardigd, onder andere omdat het gebrek aan ventilatie tot dezelfde problemen leidt als in klaslokalen. Geluid van buiten komt binnen via de ventilatieopeningen, de ventilatie zelf is lawaaierig en gebrek aan ventilatie leidt tot een aanzienlijke toename van de luchtvervuiling binnenshuis, evenals een grondige verstoring van de slaapkamerbiotoop (vochtigheid, temperatuur) - een probleem dat steeds ernstiger zal worden met de opwarming van de aarde - zoals blijkt uit tal van studies.

- Er is veel bewijs voor de nadelige gezondheidseffecten van vliegtuiglawaai, maar de indicatoren die worden gebruikt om de blootstelling aan lawaai te kwantificeren, leiden tot een onderschatting van zowel de impact van het lawaai als het aantal getroffen mensen. Om blootstelling in verband te brengen met verschillende soorten gezondheidseffecten (hinder, slaapverstoring, cardiovasculaire en cognitieve effecten, enz.) moet een set van geïntegreerde indicatoren gebruikt worden die toelaat om betrouwbare data te verzamelen die dan publiek moeten worden gemaakt. De belangrijkste indicator voor de beoordeling van de impact van nacht- en dagvluchten is de frequentie waarmee het maximumniveau dat door elke vlucht wordt bereikt hoger is dan 60 dB(A) L_{Amax} en de mate waarin deze drempel wordt overschreden. Jaargemiddelde akoestische niveaus (L_{den}, L_{night}, L_{Aeq}) worden veel gebruikt bij het maken en opvolgen van beleid en bij de communicatie tussen belanghebbenden en omwonenden. De werkgroep benadrukt het feit dat, vanuit het oogpunt van de gezondheidseffecten van lawaai, het aantal keren dat een bepaald gebeurtenisgerelateerd geluidsniveau wordt overschreden gedurende een bepaalde tijdsperiode veel relevanter is dan gemiddelde akoestische niveaus. Dit betekent dat, hoewel een vermindering van de gemiddelde geluidsniveaus (bv. L_{den}) welkom zou zijn, dit niet kan worden gebruikt als excuus om de vluchtfrequentie te verhogen.

- Gezien de ligging van Brussels Airport zouden de verschillende gewesten moeten samenwerken en het eens worden over een gemeenschappelijke reeks indicatoren, een gemeenschappelijke reeks drempelwaarden voor de bescherming van de gezondheid en de handhaving ervan. Deze indicatoren vormen immers de basis voor een vreedzaam, constructief en collaboratief overleg tussen belanghebbenden en zouden een systematisch instrument vormen om de vooruitgang te meten en verdere risicobeoordelingen uit te voeren.

- Momenteel liggen de niveaus van blootstelling aan lawaai die door de WHO worden beschouwd als drempelwaarden voor schadelijke gezondheidseffecten (45 dB(A) L_{den}, 40 dB(A) L_{night}) onder de niveaus die worden gebruikt voor rapportering en risicobeoordeling. Daarom adviseert de HGR om de regionale drempelniveaus aan te passen aan de WHO-niveaus voor rapportage en risicobeheer en tegelijkertijd de meest recente dosis-effectrelaties te gebruiken, en daarbij ook te voldoen aan de Europese wetgeving (Richtlijn (EU) 2020/367 van de Commissie tot wijziging van bijlage III bij Richtlijn 2002/49/EG).

b) Vermindering van luchtverontreiniging

De HGR beveelt aan maatregelen te nemen om de **blootstelling aan UFP** in woongebieden in de buurt van de start- en landingsbanen te **verminderen**. UFP-concentraties moeten zowel in het Vlaamse Gewest als in Brussel-Hoofdstad meer continu worden gemonitord. Naast UFP moeten ook andere emissies en/of fracties bestudeerd worden (bv. $PM_{2.5}$, PAK's, VOC's, OPE's, NO_x). Een permanente monitoring van deze emissies moet worden geïmplementeerd in de buurt van Brussels Airport. De bestaande gegevens over UFP tonen aan dat omwonenden die dicht bij de start- en landingsbanen wonen en verder langs de noordoostelijke as (geleidelijk afnemend) aanzienlijk worden blootgesteld. Het is belangrijk dat in de vroege ochtend en avond, wanneer de lucht het meest stabiel is, de emissies zeker niet verder toenemen omdat de gemeten piekniveaus van UFP nu al zorgwekkend zijn in Diegem en Steenokkerzeel.

c) Verbetering van de wetenschappelijke kennis

De bestaande gegevens over de frequentie van vliegbewegingen die een bepaalde drempel overschrijden, moeten worden geanalyseerd, (geografisch) in kaart worden gebracht en er moet een studie worden opgestart om deze gegevens te koppelen aan de beschikbare gezondheidsinformatie (bv. gebruik van geneesmiddelen tegen hypertensie en depressie, incidentie van beroertes, myocardinfarct, hartfalen, regionale mortaliteit). Conventionele epidemiologische studies missen echter de nodige gevoeligheid en het onderscheidend vermogen om de impact te meten van één omgevingsfactor (de luchthavengebonden activiteiten) op de incidentie van chronische ziekten in een complexe milieusituatie zoals de regio rond Brussel. Een moleculair-epidemiologische aanpak kan preciezer zijn en een veel duidelijker inzicht verschaffen in de mate waarin omwonenden van de luchthaven lijden onder verhoogde interne blootstellingen en geassocieerde gezondheidseffecten in verband met luchthavenactiviteiten. Longitudinale en moleculair-epidemiologische studies zullen meer accurate informatie opleveren over de mogelijke gezondheidseffecten van het verminderen van vliegtuiglawaai en luchtvervuiling rond Brussels Airport. Het is belangrijk om in deze studies ook andere invloedsfactoren dan lawaai mee te nemen, zoals socio-economische en psychologische factoren. Bovendien is het bekend dat er socio-economische ongelijkheden bestaan in de blootstelling aan lawaai en deze moeten ook gedocumenteerd worden.

- Het zou interessant zijn om beschikbare kankerregistratiegegevens te gebruiken en een epidemiologische studie op te zetten om te bepalen of de incidentie van kanker (inclusief hersenkanker, hoewel de populatie mogelijk te klein is) hoger is in de omgeving van de luchthaven dan in de rest van het land en of de incidentie van kanker verband houdt met hogere niveaus van vliegtuiglawaai en luchtvervuiling in de regio.

- Slaaponderzoek wordt voornamelijk uitgevoerd met invasieve apparatuur. Recentelijk maken minder invasieve protocollen het mogelijk om korte termijn biologische effecten op de slaap te meten in reële omstandigheden. Grotere studies die reële blootstelling aan lawaai en biologische reacties op korte termijn combineren zijn haalbaar en betaalbaar, wat de kennis aanzienlijk zal vergroten. Er lopen verschillende studies in de VS en het VK, maar de resultaten zijn nog niet beschikbaar. Gelijkaardige studies kunnen in België worden uitgevoerd.

- Gelijkaardig aan de benadering van het RIVM, beveelt de HGR aan om de effecten van kortdurende blootstelling aan UFP op de longfunctie, en UFP en geluid op ontstekingsparameters bij kinderen en volwassenen te onderzoeken.

- **Gezien het bestaande overtuigende bewijs voor de nadelige gezondheidseffecten van vliegtuiglawaai en -emissies, mag de implementatie van maatregelen echter niet worden uitgesteld terwijl nieuwe wetenschappelijke studies worden uitgevoerd.**

d) Communicatie

De HGR beveelt aan te investeren in overleg en het vertrouwen van de bewoners in de autoriteiten en het luchthavenbeheer te verbeteren. **Effectieve communicatie is van het grootste belang**. De genomen maatregelen moeten **transparantie** van het besluitvormingsproces, de implementatie van **eerlijke procedures** waarin alle belanghebbenden vertegenwoordigd zijn en **openheid over de verdeling van kosten en baten** omvatten. Tijdens het besluitvormingsproces moeten omwonenden tijdig waarheidsgetrouwe, volledige en duidelijke informatie krijgen over de omvang, duur en niveaus van het lawaai en de vervuilende stoffen rond Brussels Airport.

e) Naar een groener luchtvervoer:

De luchtvaart heeft een aanzienlijke invloed op zowel de menselijke gezondheid als het milieu en op de klimaatverandering. Reductiestrategieën moeten worden overwogen in een internationaal kader om tot een meer duurzame vervoers- en mobiliteitsstrategie te komen. De deskundigengroep heeft gewezen op de belangrijke negatieve effecten van het luchtvervoer op de volksgezondheid. Er kunnen verbeteringen worden bereikt door de verschillende beleidsaanbevelingen in dit rapport op te volgen, maar deze potentiële verbeteringen zullen teniet worden gedaan als het luchtverkeer blijft groeien. De belangrijkste vermindering van de gezondheidseffecten van het luchtvervoer zal inderdaad komen van een vermindering van het luchtverkeer. Als samenleving moeten we nadenken over onze (recente) afhankelijkheid van onmiddellijke goederenleveringsprocessen en over de waarde die we hechten aan het vaak vliegen naar nabije of verre bestemmingen voor zaken of vrije tijd. De vergroening van het luchtvervoer zal voornamelijk afhangen van ons collectieve vermogen om het luchtverkeer te verminderen.

Keywords and MeSH *descriptor terms*[5]

MeSH terms*	**Keywords**	**Sleutelwoorden**	**Mots clés**	**Schlüsselwörter**
Aircraft	*Aircraft*	Vliegtuig	*avion*	*Flugzeug*
Aviation	*Aviation*	Luchtvaart	*aviation*	*Luftfahrt*
Cancer	*Cancer*	Kanker	*Cancer*	*Krebs*
Cardiovascular disease	*Cardiovascular disease*	Hart- en vaatziekte	*Maladie cardiovasculaire*	*Herz-Kreislauf Erkrkankung*
Chemicals	*Chemicals*	Chemische stoffen	*Sustances chimiques*	*Chemikalien*
Cognitive dysfunction	*Cognitive impairment*	Cognitieve stoornis	*Trouble cognitif*	*Kognitive Störung*
Endocrine disruptors	*Endocrine disruptors*	Hormoonverstoorders	*Perturbateurs endocriniens*	*Endocrine Disruptoren*
Noise	*Noise*	Lawaai	*bruit*	*Lärm*
Pollution	*Pollution*	Pollutie	*Pollution*	*Umweltverschmutzung*
Sleep	*Sleep*	Slaap	*Sommeil*	*Schlaf*
	Spatial planning	Ruimtelijke planning	*Aménagement du territoire*	*Raumplanung*
Transportation noise	*Transportation noise*	transportlawaai	*Bruit des transports*	*Transportlärm*
	Urban areas	Stedelijke gebieden	*Zones urbaines*	*Städtische Gebiete*

MeSH (Medical Subject Headings) is the NLM (National Library of Medicine) controlled vocabulary thesaurus used for indexing articles for PubMed http://www.ncbi.nlm.nih.gov/mesh.

[5] De Raad wenst te verduidelijken dat de MeSH-termen en sleutelwoorden worden gebruikt voor referentiedoeleinden en een snelle definitie van de scope van het advies. Voor nadere inlichtingen kunt u het hoofdstuk "methodologie" raadplegen.

INHOUDSTAFEL

ABBREVIATIONS AND SYMBOLS

ACNUSA	*Autorité de Contrôle des Nuisances Aéroportuaires* – Airport Pollution Control Authority
BATC	Brussels Airport Traffic Control
CA	Chromosal aberration
CI	Confidence interval
COPD	Chronic obstructive pulmonary disease
CVD	Cardiovascular Disease
DALY	Disability-adjusted Life Years
DEBATS	*Discussion sur les Effets du Bruit des Aéronefs Touchant la Santé*
DNA	Deoxyribonucleic acid
ECG	Electrocardiogram
EEG	Electroencephalogram
EIA	Environmental Impact Assessment
EMG	Electromyography
FEV1	Forced Expiratory Volume in 1 second
Fpg	Formamidopyrimidine DNA glycosylase
FVC	Forced Vital Capacity
GC	Gas Chromatography
HA	High annoyance
HR	Hazard Ratio
HRV	Heart Rate Variability
HSD	High sleep disturbance
HYENA	Hypertension and Exposure to Noise near Airports
ICBEN	International Commission on Biological Effects of Noise
IQR	Interquartile range
IR	Intermittency ratio
IRR	Incidence rate ratio
L_{Aeq}	A-weighted, equivalent continuous sound level
$L_{A,max}$	Maximum sound level reached during a measuring period
L_{den}	Day-evening-night sound level
L_{night}	Night sound level
LDSA	Lung Deposited Surface Area
LTO	Landing and Take-Off
MER	*Milieu-effecten rapport* (Environmental Impact Assessment, EIA)
NORAH	Noise-Related Annoyance, Cognition, and Health
NO_x	Nitrogen Oxides
OPE	Organophosphate Ester
OR	Odds Ratio
PAH	Polycyclic Aromatic Hydrocarbon
PM	Particulate Matter
PNC	Particle Number Count
PTB	Preterm Birth
RANCH	Road traffic and Aircraft Noise exposure and Children's Cognition and Health
REM	Rapid Eye Movement
RIVM	*Rijksinstituut voor Volksgezondheid en Milieu*
RMSSD	Root Mean Square of Successive Differences between normal heartbeats
RR	Risk Ratio

SCE	Sister chromatid exchange
SDNN	Standard deviation of the normal-to-normal intervals
SE	Sleep efficiency
SEL	Sound Exposure Level
SHC	Superior Health Council
SIR	Standardised Incidence Ratio
SOL	Sleep onset latency
SO_x	Sulfur oxides
SWS	Slow Wave Sleep
Tm	Tail moment values from untreated cells
Tmenz	Tail moment values from Fpg-enzyme treated cells
TNO	*Nederlandse Organisatie voor Toegepast-Natuurwetenschappelijk Onderzoek*
UFP	Ultra Fine Particle
ULSJ	Ultra-Low Sulfur Jet Fuel
VITO	*Vlaamse Instelling voor Technologisch Onderzoek*
VLOPS	*Vlaamse Operationeel Prioritaire Stoffen*
VOC	Volatile Organic Compound
WASO	Wakefulness After Sleep Onset
WHO	World Health Organisation

GLOSSARY (WHO 2018)

A-weighting:	A frequency-dependent correction that is applied to a measured or calculated sound of moderate intensity to mimic the varying sensitivity of the ear to sound for different frequencies
$L_{Aeq,T}$	A-weighted, equivalent continuous sound pressure level during a stated time interval starting at t1 and ending at t2, expressed in decibels (dB), at a given point in space[6]
$L_{A,max}$	Maximum time-weighted and A-weighted sound pressure level within a stated time interval starting at t1 and ending at t2, expressed in dB[7]
L_{day}	Equivalent continuous sound pressure level when the reference time interval is the day[8]
L_{den}	Day-evening-night-weighted sound pressure level as defined in section 3.6.4 of ISO 1996-1:2016[9]
$L_{evening}$	Equivalent continuous sound pressure level when the reference time interval is the evening[10]
L_{night}	Equivalent continuous sound pressure level when the reference time interval is the night[11]

[6] *Source*: ISO (2016).
[7] *Source*: ISO (2016).
[8] *Source*: ISO (2016).
[9] *Source*: ISO (2016).
[10] *Source*: ISO (2016).
[11] *Source*: ISO (2016).

I INTRODUCTION AND ISSUE

On October 28th 2022, the Superior Health Council (SHC) received a request for advice from the Federal Minister of Social Affairs and Public Health concerning the issue of noise in the vicinity of Brussels Airport. More specifically, the following questions were put to the SHC:

a. What are the direct and indirect effects on public health of the environmental noise generated by aircraft, both in terms of noise level and flight frequency, in the wider vicinity of the airport?
b. Are there any differences in the effects of daytime, early morning and night flights?
c. Is there any evolution in the assessment of these effects in the international scientific literature, and have any good studies been conducted on this subject in the vicinity of comparable airports in Western Europe whose methodology could be useful in Belgium?
d. What impact do these effects have on healthcare budgets and organisation?
e. What are the policy recommendations on this issue?

This report provides an overview of current published scientific knowledge obtained by collecting data from several existing studies on Brussels Airport, but also by identifying gaps in knowledge and holding expert hearings on study reports from airports in France (ACNUSA), Germany (NORAH study), and the Netherlands (RIVM study). Although the request for an advisory report focused on the effects of aircraft noise, which usually receives the most attention in the debate on Brussels Airport, the SHC opted for a broader analysis by also including the effects of air pollution related to aircraft movements. Air pollution is one of the largest environmental risks to human health, significantly increasing the incidence of diseases, especially cardiovascular diseases and several types of cancer, premature deaths, and disability-adjusted life years (DALYs). Adverse health effects are not only caused by single factors but are often amplified by the interference between multiple factors. Measures taken to counteract negative influences should therefore take these interferences into account. Besides, the SHC acknowledges that emissions from aviation also contribute to global environmental issues such as climate change and nitrogen deposition, and therefore calls for a wider reflection on sustainable aviation. Finally, the SHC provides recommendations for further actions to improve the protection of residents' health in the vicinity of Brussels airport. Other interests (e.g., economic and financial) are not considered.

The national airport of Belgium, Brussels Airport, is located in the Flemish region, just northeast of Brussels-Capital Region (Figure 1). This Region, consisting of Brussels Capital and the Flemish Periphery ("*Vlaamse Rand*"), is a highly populated zone due to (sub)urbanisation after World War II. As a result, the airport at this location, which has its origins in a German military airfield, has slowly become embedded in what is today one of the most populated area of the country. As a result, the various activities of the airport have an impact (in the broadest sense: economy, health, etc.) on a large number of residents. Hence, due to the specific location of Brussels Airport near the border of different Regions and Communities, adequate handling and burden-sharing of aircraft noise and other aviation-related problems has been a complex political issue for decades. For a comprehensive overview of the history of Brussels Airport, its development and the various issues, the reader is referred to the paper of Boussauw & Vanoutrive (2019) in "Sustainability".

The airport has three runways, in a Z-shaped layout (Figure 2). Two main, parallel runways are oriented more or less east-west, facing Brussels (07L/25R and 07R/25L). Due to the dominant south-western wind during the day, take-offs mainly occur towards the capital. The third runway is an auxiliary runway with a north-south orientation (01/19) that is more often used in the event of strong north or south winds, but sometimes also when there are no strong winds. Weak winds at night offer more freedom on the choice of the runway.

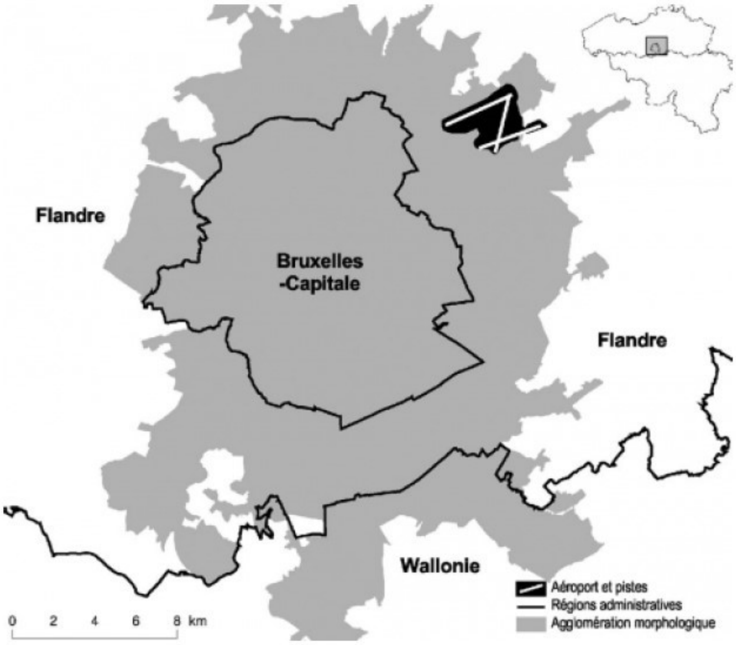

Figure 1: *Location of Brussels Airport (Belgium) and the Flemish Periphery surrounding Brussels. The boundaries of the Regions are indicated with a black line, the morphology of the highly urbanised agglomeration is grey-coloured. Source: Dobruszkes (2008)*

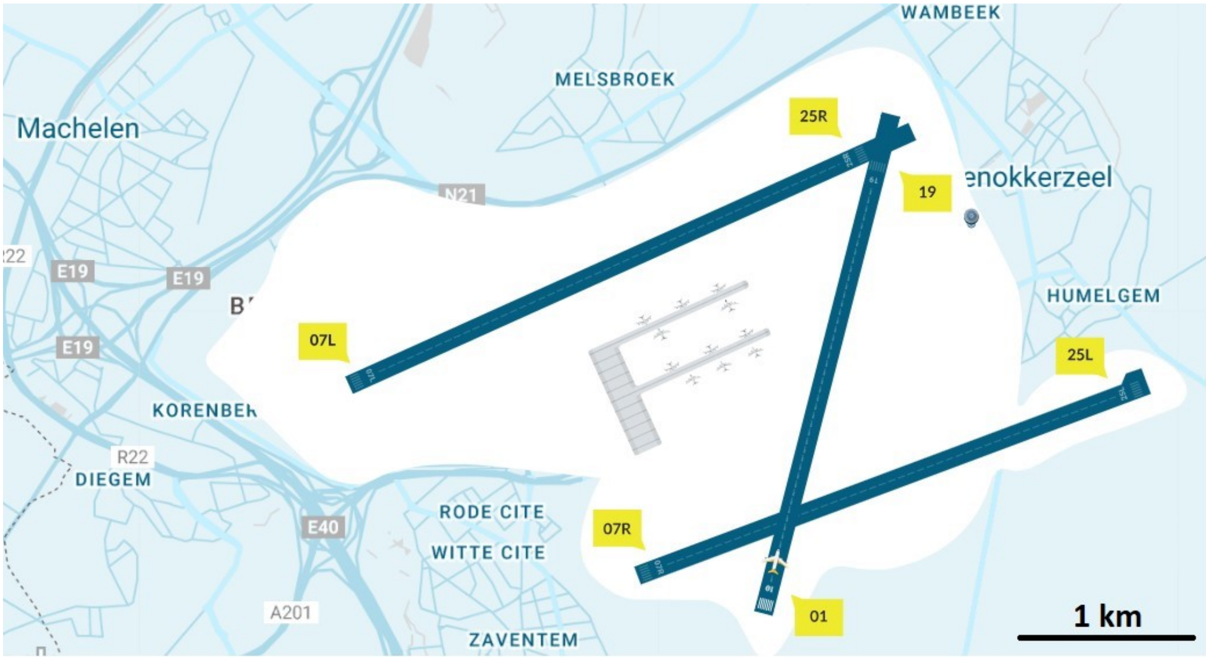

Figure 2: *Location of the runways of Brussels Airport (Belgium), based on the map provided by BATC*[12]

According to Brussels Airport Traffic Control (BATC), 234 461 aircraft movements were performed in 2019 (vs. 178 924 in 2022), of which 17 347 (16 916 in 2022) were executed between 11 pm and 6 am. Given the fact that air traffic had not yet returned to pre-Covid levels in 2022 (figures between brackets), the figures of 2019 are used, as they provide a more reliable picture of the usual aircraft movements at Brussels Airport. Most of the night flights between 11 pm and 6 am are cargo transports (45.68 % in 2019 and 53.34 % in 2022).

[12] https://www.batc.be/en/noise/noise-contours (accessed on 28/3/2024)

However, these numbers heavily underestimate the real burden of air traffic at night as 10 029 flights (7 804 in 2022) happen between 6 and 7 am. According to the reported noise contours around Brussels Airport for the year 2019, 163 718 residents lived within the > 45 dB(A) L_{night} noise contours (11 pm – 7 am) (in 2022: 151 901 residents) (official noise contour reports: Dekoninck et al., 2020; Vinkx & Graas, 2023). More than 163 518 residents experienced more than 10 flyovers with a $L_{A,max}$ over 60 dB(A) per night (in 2022: 123 239 residents).

The impacted area regarding noise (L_{den} >55 dB(A)) in the Flemish Region contains 11 pre- and elementary schools. In Brussels, the impacted region contains 141 school buildings (Brussels Environment, 2016). According to the noise contour maps of Brussels Airport in 2019, 97 624 inhabitants were exposed to L_{den}>55 dB(A) (78 326 in 2022; Dekoninck et al., 2020; Vinkx & Graas, 2023).

Different governing levels are involved. The Federal government is responsible for running Brussels Airport, as well as for managing the airspace. In this capacity, the Federal government can impose restrictions and coordinate slots at Brussels airport. However, the Regions (Flemish Region and Brussels-Capital Region) can impose their own environmental policy concerning noise and pollution. As Brussels Airport is entirely located within the Flemish Region, the environmental permit is granted by the Flemish government based on the laws and standards set there.

Historically, various endeavours have been undertaken to address the challenges posed, particularly by noise disturbances. Back in 1974, then-Minister of Traffic, Chabert, instituted national flight routes known as the "Chabert routes." These routes included specific take-off procedures for heavy aircraft and straight-line take-offs along the runway axis. Subsequent ministers, such as Dehaene, Coëme, Durant, Onkelinx, Anciaux, Schouppe and Wathelet contributed their own directives and adjustments to night-time and daytime take-off procedures over the years, aiming to strike a balance between flight path concentration and dispersion, as well as regulating weekend and night-time flights.

The airport is also a provider of employment, including a large number of low-skilled jobs. This is an important aspect considering the proximity of the airport with an urban area. The complex political situation meant that the different governmental levels could not reach a consensus regarding night flights in 2008, which led the international courier company DHL to move part of its business from Brussels Airport to Leipzig, resulting in the loss of an estimated 1250 jobs. For more context on this topic, see Boussauw & Vanoutrive (2019).

The impact of aviation on air quality is also a subject of concern, but it has often been ignored in the public debate (Boussauw & Vanoutrive, 2019). The airport's operations contribute to emissions of various pollutants. It is only in recent years that the scientific literature has increasingly focused on the health effects of ultrafine particles (UFPs) near airports around the world. Efforts to mitigate these emissions have included the use of cleaner aircraft technology and the implementation of ground-based measures to reduce engine idling.

II METHODOLOGY

After analysing the request, the Board and the co-Chairpersons of the working group identified the necessary fields of expertise. An *ad hoc* working group was then set up which included experts in acoustics, atmospheric pollution, cancerology, cardiology, environmental effects of transport and mobility, environmental health, environmental metrology, human ecology, human health, pain management, psychiatry, somnology, toxicology, and transport geography.
The experts of this working group provided a general and an *ad hoc* declaration of interests and the Committee on Deontology assessed the potential risk of conflicts of interest.

This advisory report is based on a review of the scientific literature published in both scientific journals and reports from national and international organisations competent in this field (peer-reviewed), as well as on the opinion of the experts. The scientific literature was collected using search engines such as Google Scholar, and databases such as PubMed, Web of Science and Scopus.

The working group organised three hearings of international organisations, viz. ACNUSA (*Autorité de Contrôle des Nuisances Aéroportuaires* – Airport Pollution Control Authority) on 19 September 2023, NORAH (Noise-Related Annoyance, Cognition, and Health) on 25 September 2023 and RIVM (*Rijksinstituut voor Volksgezondheid en Milieu)*on 20 September 2023. A hearing of the Flemish research institute VITO (*Vlaamse Instelling voor Technologisch Onderzoek)* was held on 9 November 2023.

ACNUSA is a French independent administrative authority that is competent for both aircraft noise as well as pollution and covers the 12 busiest airports in France. It is invested with a normative (e.g. defining indicators, trajectory measurements and visualisation systems) as well as a consultative (provide advice to the government), and authoritative mission (by publishing opinions, advice and recommendations), but it also has the power to impose fines in the event of non-compliance with environmental regulations pertaining to civil aviation.

The NORAH study was conducted between 2011 and 2015 to assess the effect of traffic noise, and especially aircraft exposure on residents living near Frankfurt airport. More specifically, it looked at the effects on health, quality of life and the cognitive development of children.

The Dutch RIVM performed an extensive study on the potential health effects of ultrafine particles from air traffic near Schiphol.

The Flemish VITO measured UFP near Brussels Airport and modelled the air pollution around Brussels.

Once the advisory report was endorsed by the working group, it was ultimately validated by the Board.

III ELABORATION AND ARGUMENTATION

1 Aircraft noise: exposure and health effects

The first question that was examined by the SHC concerned "*the direct and indirect effects on public health of the environmental noise (both in terms of noise level and flight frequency) and air pollutant emissions generated by aircraft in the wider vicinity of the airport*".

Before going into the details of this issue, it is important to note that studying adverse health effects (direct and especially indirect) is complicated and requires complex, unbiased epidemiological studies that are statistically corrected as much as possible for possible confounders. **An "association" between exposure and a particular effect does not necessarily imply a "causal relationship".** To establish a causal relationship, some criteria need to be verified: (1) Consistency of the association in different studies with different methods; (2) Strength of the association; (3) Specificity of the association between a single cause and the effect; (4) Existence of a temporal relationship; (5) Coherence of the association with existing knowledge (and, preferably, existence of a mechanistic relationship); and (6) Existence of a dose-response relationship. The studies and systematic reviews considered by the SHC took these criteria and the existence of confounders into account as much as possible.

Although the levels of aircraft-induced noise are generally too low to cause biological damage to the auditory system[13], there is increasing awareness that excessive aircraft noise is harmful to health and well-being. As a result, the 2018 Environmental noise guidelines for the European Region (WHO) [14], and the 2022 update of the Compendium of WHO and other UN guidance on health and environment, recommend limiting the average exposure to environmental noise to 45 dB(A) L_{den} during the daytime and 40 dB(A) L_{night} during the night-time (i.e. between 11 pm and 7 am) (for the definitions of L_{den} and L_{night}, see section 1.2.2 below. This is based on two systematic reviews to support the revision of noise guidelines by the WHO (Clark et al., 2018). The reviews assessed the quality of the evidence across studies on the effect of environmental noise (road traffic noise, aircraft noise, railway noise, wind-turbine noise) on quality of life, well-being and mental health.

Aircraft noise above 45 dB(A) L_{den} during the daytime is associated with adverse health effects while levels above 40 dB(A) L_{night} during the night-time are associated with adverse effects on sleep. Note that these recommendations concern noise levels outdoors, viz. at the most exposed façade of the dwellings.

More specifically, excessive noise exposure is associated with different health outcomes: self-reported **high annoyance and sleep disturbance** (which constitutes the most serious effect on health of exposure to noise according to the WHO, 2009 and 2018)**,** as well as a series of health effects that may emerge indirectly as a result of these stressors**, viz. cognitive impairment, hypertension, cardiovascular diseases, mental health and depression, etc.** Importantly, in order to gain a reliable picture of the associations between aircraft noise and health effects, it is necessary to understand the nature of noise, how exposure to noise can be measured and quantified as well as how selected indicators may affect any conclusions.

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[13] Although most of the effects described are non-auditory, some studies have nevertheless reported effects on hearing (e.g. Chen et al. 1997). In contrast, other studies indicate that exposure to environmental and leisure noise at L_{Aeq},24h < 70 dB(A) does not cause hearing loss in the vast majority of people (> 95%), even with lifetime exposure (Passchier & Passchier-Vermeer 2000). The WHO review found no studies or evidence to support a link between residential exposure to aircraft noise and permanent effects on hearing. This is of course not true for those working around aircraft on the ground, who must wear protective equipment at all times

[14] https://www.who.int/europe/publications/i/item/9789289053563 (accessed on 22/1/2024)

1.1 Aircraft noise: origin

We will concentrate here only on the noise perceived outside the aircraft which is the noise component that has an impact on people living near airports (as opposed to the interior noise perceived by passengers and crew).

Aircraft noise pollution results essentially from two sources: engine noise (fan, compressor, combustion, turbine, jet) and airframe noise induced by the turbulent airflow around and behind the aircraft (especially around slats, flaps and landing gear). For more detailed information on how noise is generated, see e.g. Coyette & Migeot (2017).

1.2 Noise: definition and indicators

1.2.1 What is noise?

Noise is generally defined as a set of sounds perceived as unwelcome. As a result, it is an inherently subjective notion: what is perceived as unwelcome by some is not necessarily so by others. Context and timing also play a role. For instance, while music is generally enjoyable, it can become disturbing when it disrupts concentration or relaxation, such as during sleep or focused work. Moreover, the intensity of sound plays a crucial role; beyond a certain threshold, even seemingly benign sounds can become irritating or detrimental to health.

It follows that when addressing noise, and especially aircraft noise, both its physical properties (such as sound intensity, average energy levels, individual sound events, and intermittency) and its potential impact on human health need to be considered. Various types of indicators have been developed to assess impacts on human health, reflecting the multifaceted nature of noise pollution and its effects on exposed individuals.

1.2.2 Assessing noise exposure: types of indicators

Sound **intensity,** measured in Watts per square meter (W/m^2), quantifies the local flux of **power** conveyed by a sound wave. Yet, human perception of loudness is not directly related to the sound intensity but rather to the logarithm of that quantity. The *sound intensity level* (L), which is expressed in decibels (dB), is defined as 10 times the logarithm of the ratio between the sound intensity and a reference intensity of 10^{-12} W/m^2, which represents an average human auditory threshold (ISO, 2003).

A sound event unfolds over **a given stretch of time** (one minute, one hour, one day). While the **total sound intensity level** associated with the event offers valuable insight, it lacks the ability to provide any information about how the spectrum and the sound intensity level fluctuate during this particular stretch of time. To address this, the signal is segmented into very brief time intervals, typically half a second each. For each interval, an intensity level is computed, providing a time history of the noise exposure.

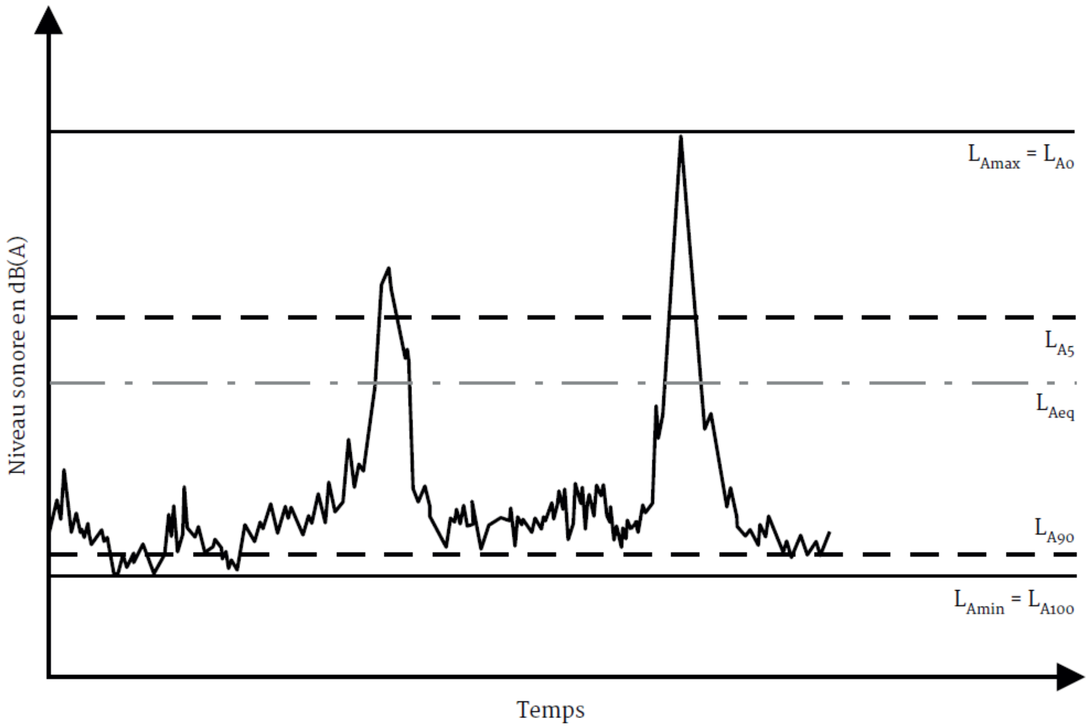

Figure 3: *Changes in instantaneous levels over time (Source: Coyette & Migeot 2017:36)*

As depicted in the figure above, the same noise event can be examined from very different perspectives. Thus, among other things, the figure provides information about (see Coyette & Migeot, 2017):

- Noise **peaks (and nadirs)**, viz. the minimum and maximum levels (expressed as L_{Amin} and $L_{A,max}$, respectively, in the figure above)
- **Average sound levels**: the **equivalent sound level over time.** In the figure above, the horizontal line labelled $L_{Aeq,}$ corresponds to the average level during the sound event. Averaging times can range from brief moments to one working day (8 h), one calendar day (24 h) or a full year. Typically, these indicators average out the noise peaks. Yet, as will be discussed below, these noise peaks are in fact a key aspect of the noise-induced annoyance and health effects.
- **Time fraction of threshold exceedances**. In the figure above, L_{A5} is exceeded 5% of the time whilst L_{A90} is exceeded 90 % of the time.

It follows that specific indicators have been tailored for analysing noise from each of these different perspectives. Given the fact that indicators such as those broached above focus on different aspects of noise, they also serve different purposes. For instance, single noise events can be described using either the **$L_{A,max}$** or **SEL** (Sound Exposure Level) indicators. $L_{A,max}$ denotes the maximum sound level reached during a measuring period, representing a peak level. In contrast, SEL encompasses the total acoustic intensity measured over the entire duration of a single noise event, accounting for both sound intensity and duration. This enables comparisons among events of varying durations and levels. For instance, a long flyover by a quiet airplane may have the same SEL as a short flyover by a noisier aircraft (Coyette & Migeot 2017: 38).

As regards **average-energy level energy based indicators**, aside from the L_{Aeq} mentioned above (A-weighted, equivalent continuous sound level), another commonly used indicator in the field of environmental noise assessment is L_{den} (Day-evening-night sound level). This indictor denotes the **noise level calculated over a designated reference period** (spanning one day, month, or year), with distinct levels computed for daytime (7 am to 7 pm), evening (7 pm to 11 pm) and night (11 pm to 7 am). Notably, the evening and night levels are weighted by factoring in penalties of 5 and 10 dB(A), respectively. Therefore, this weighting gives more importance to noise levels in the evening and at night, reflecting the heightened disturbance caused by noise during these times compared to daytime. This indicator is mainly used to draw up noise exposure plans and is imposed by European regulations. Its nocturnal component, viz. L_{night}, is used by the WHO in its recommendations (WHO, 2018).

Average-energy level energy based indicators provide information that is very different from that provided by e.g. (flight) **frequency-based indicators (number of threshold exceedances)**, which count how many times a given threshold (e.g., 55 dB(A) has been surpassed or noise intensities capable of disrupting sleep during night-time have occurred. Consequently, while a single loud event, like a passing moped at night, might minimally impact the average annual noise level, it can wake up the whole neighbourhood, illustrating the disparate focus of these indicators.

It follows that the selection of the appropriate indicator holds significant importance. Similarly, the units employed to measure the sound intensity to which individuals are exposed are vital considerations. Notably, noise consists of vibrations across various frequencies, yet the human auditory system is not equally sensitive to all frequencies. To address this, sonometers are fitted with filters, called A, B, C, and D filters, that adjust the measured signal to reflect the sensitivity of the human ear to different frequencies (Coyette & Migeot, 2017:35). The **A-filter** is extensively used in environmental acoustics. It essentially tones down low and high frequencies, to which the human ear is least sensitive, and accentuates frequencies around 3-6 kHz instead. This adjustment yields a weighted sound intensity level that is expressed in dB(A) units. However, it’s important to note that the A filter largely overlooks the disruption of sleep caused by lower sound frequencies, which penetrate more easily into homes. In contrast, the C-filter is more adept at capturing high and low frequencies (Gezondheidsraad, 2004: 82). Moreover, the A-weighting is less effective for addressing high noise peaks. Nonetheless, despite this limitation, it remains the most widely used weighting method and serves as a standard feature for quantifying ambient noise levels.

1.2.3 Indicators

Indicators are essential for communication about noise. Yet whilst indicators such as L_{Aeq} and L_{den} are those that are most commonly used to describe the link between noise exposure and health effects, other indicators such as event-based indicators have more recently been used. Essential for the assessment of the real health and physiological impact of noise on human beings is the frequency at which specific noise levels are reached or exceeded and the magnitude of such exceedances.

An important observation is that, **whilst noise indicators are rarely questioned, they are not as neutral as might be expected.** At the crossroads between politics and science, they are the outcome of political compromises that were influenced by political goals. Environmental indicators chosen by public authorities directly define the content of environmental reports. Some indicators tend to underestimate the number of residents exposed: thus, a study by Cidell (2008) found that some residents living outside official noise contours in Minneapolis still reported suffering from aircraft noise pollution, suggesting that official noise assessment may not sufficiently cover all aspects of noise pollution, such as perceived annoyance (see below). The indicators used in Europe are no exception (Dobruszkes & Efthymiou, 2020). This could be solved by lowering the thresholds of average noise based indicators, e.g. from 55 dB(A) L_{den} to 45 dB(A) L_{den}, or by using indicators based on the frequency of noise events. The following illustration shows the different spatial extent of areas around Brussels airport based on A) lowering the thresholds and B) the 55 dB(A) L_{den} indicator vs. noise events of at least 60 dB(A) that occur at least 50 times during daytime.

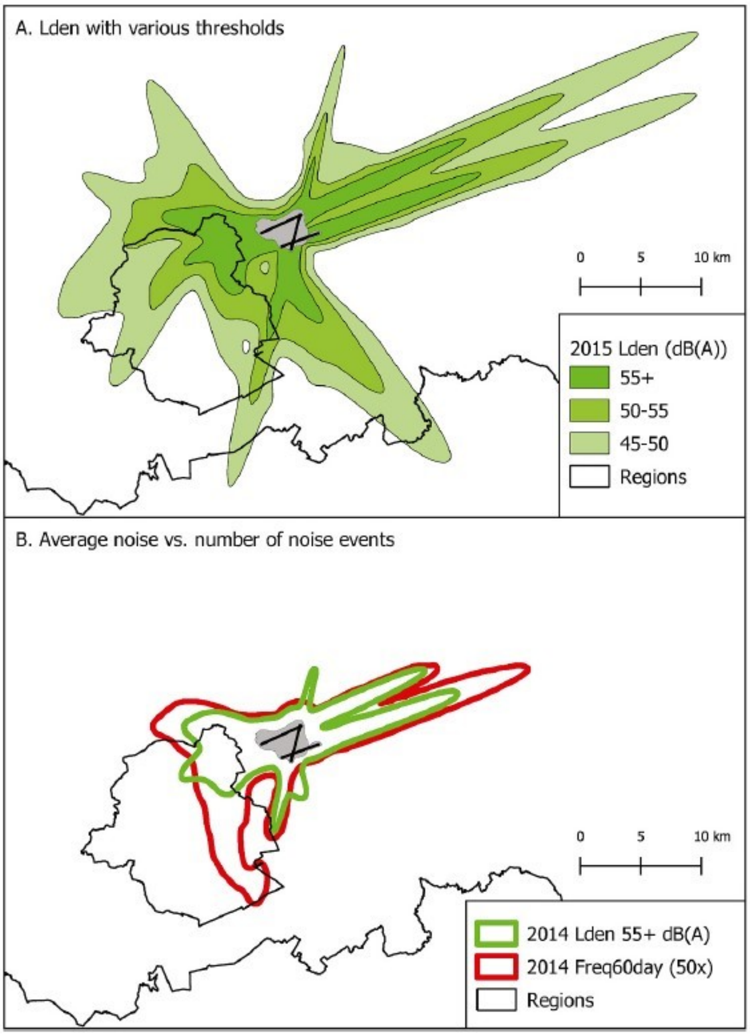

Figure 4: *Comparing extent of various noise indicators (Source: Dobruszkes & Efthymiou, 2020)*

It follows that both the type of indicator used (average energy based indicator vs. event-based indicator) and the thresholds used have a significant influence on the assessment of the noise exposure.

In sum: Official noise indicators imposed by EU, national or regional regulations are, by nature, the result of political decisions and thus of a compromise between diverging interests. These decisions may be guided by scientific knowledge to some extent, but also by social considerations and economic interests. In addition, official indicators may be based on outdated scientific knowledge.

It is important to note that the co-existence of so many different noise metrics creates serious communication issues and undue confusion. Therefore it is imperative for the different Belgian authorities to reach consensus on a standardised set of indicators, as this would certainly contribute to a more peaceful debate among stakeholders.

In the following sections, we will see that the significance of selecting the appropriate indicators to assess the health effects of aircraft noise cannot be overstated.

1.3 Effects of exposure to aircraft noise

1.3.1 Noise exposure around Brussels airport

In order to measure the noise exposure of the population residing in the vicinity of the airport, a network of noise monitoring terminals continuously captures noise levels. However, the number of monitoring stations is inevitably restricted, thus preventing the creation of a truly exhaustive map covering the entire area.

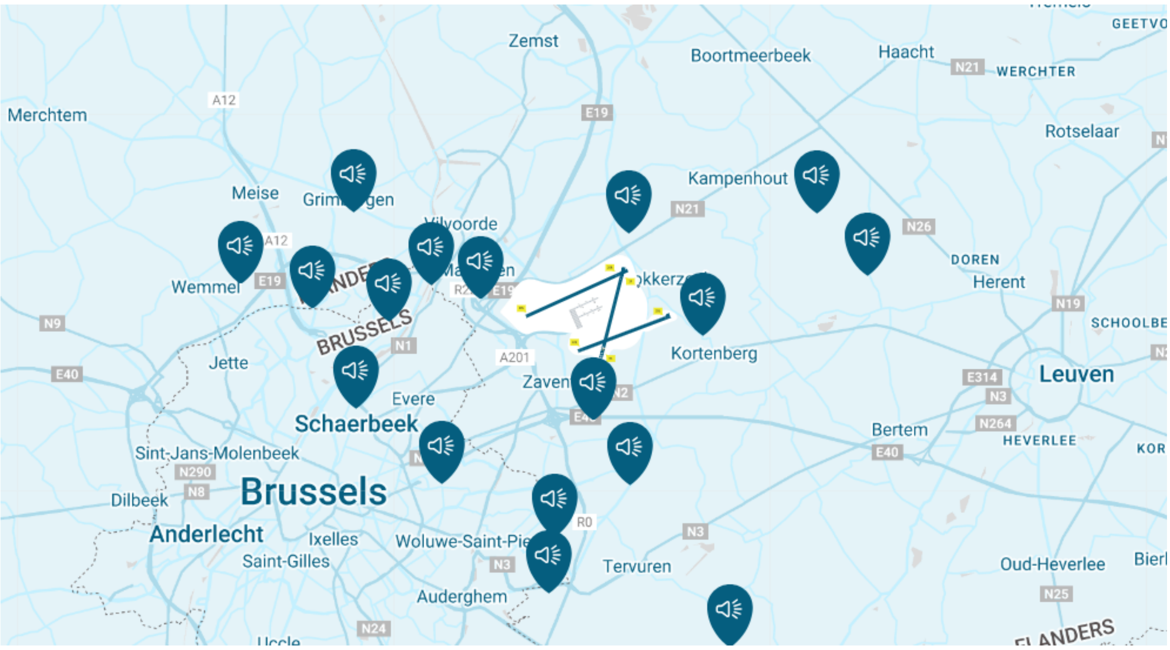

Figure 5: *Network of noise monitoring terminals around Brussels airport (Source: https://www.batc.be/fr/bruit/mesures-sonores)*

All airports, including Brussels Airport, are mandated by law to develop a Noise Exposure Map. This map is generated by integrating data on flight schedules, aircraft trajectories, flight procedures, aircraft models, and meteorological factors. The figure below shows a noise exposure map for Brussels airport in 2019. The two contours delineate areas where the L_{den} level exceeds 65 (dashed line) and 55 (solid line) dB(A), respectively.

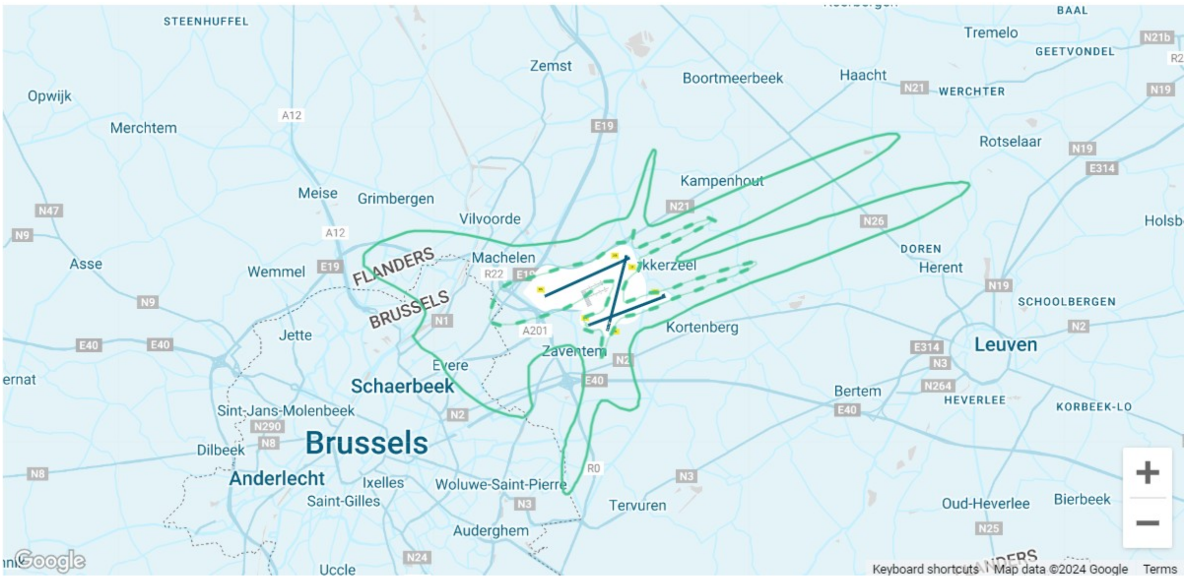

Figure 6: *noise exposure map for Brussels airport in 2019 (Source: https://www.batc.be/fr/bruit/contours-de-bruit)*

1.3.2 *Noise-induced high annoyance (HA)*

As mentioned above, noise is commonly defined as a set of sounds perceived as unwelcome. When noise disrupts a person's thoughts, emotions or activities, it can provoke feelings of anger, irritation, discomfort, dissatisfaction or injury, ultimately culminating in noise-induced annoyance (Gezondheidsraad, 1994). More specifically, Guski et al. (1999) define noise-induced annoyance as "*a psychological concept which describes a relation between an acoustic situation and a person who is forced by noise to do things he/she does not want to do, who cognitively and emotionally evaluates this situation and feels partly helpless*". Therefore, annoyance is defined as a negative response to noise that includes cognitive, emotional, and behavioural aspects.

Crucially, the WHO writes that "*a vast amount of evidence proves the **association between aircraft noise and annoyance***" (WHO 2018: 68).

Unsurprisingly, even though there are complicating factors (see below), the level of annoyance goes up with the level of exposure. Research has found that when the L_{den} values at the most exposed outer wall of the building are slightly above 40 dB(A), the number of exposed people reporting serious annoyance goes up.

More specifically, based on its systematic review of the literature regarding the health effects of traffic noise, the WHO (2018) concluded that, for aircraft noise, 40 dB(A) L_{den} corresponds to a 1.2 % prevalence of high annoyance (HA). Using exposure response functions (quadratic regression analysis) from 12 studies and aggregating data from 17 094 study participants around national and international airports, the correlation between L_{den} and HA was found to be highly significant ($r = 0.436$; 95 % CI = 0.368–0.499; $p < 0.001$). The analysis allowed to calculate the mean percentages of residents highly annoyed by aircraft noise at certain noise levels (Guski et al., 2017). At 45 dB(A) (L_{den}) increases to 9.4 %, at 50 dB(A) (L_{den}), HA increases to 17.9 % and at 60 dB(A) (L_{den}) it rises further to 36 % HA (see table below).

Table 1: *Association between exposure to aircraft noise (L_{den}) and annoyance (% HA) (WHO 2018: 69)*

L_{den} (dB)	%HA
40	1.2
45	9.4
50	17.9
55	26.7
60	36.0
65	45.5
70	55.5

Also according to the WHO assessment, a 10 dB(A) rise in aircraft noise (L_{den}) is associated with an almost five-fold increase of HA risk (OR = 4.78). This is a notably higher increase compared to the three-fold rise associated with 10 dB(A) increments in road or railway traffic noise.

Aircraft noise induced annoyance, and more specifically, high annoyance, has been considered an **early warning for adverse health effects**. Indeed, aircraft noise can trigger **psychological stress** responses, which, when exposure is prolonged, can in turn lead to a continuous state of stress. This prolonged psychological stress affects the ability to recuperate effectively, consequently elevating the risk for certain adverse health effects. For instance, research has associated aircraft-noise induced annoyance with an increased risk of mental health outcomes, including depression and anxiety (cf. e.g. Spilski et al., 2019; Baudin et al., 2021; Benz et al., 2022; Gong et al., 2022). Thus, in the meta-analyses conducted by Gong et al., the risk of depression was 1.23 times greater in highly noise-annoyed individuals (for all types of noise), whilst there was a 55 % higher risk of anxiety and 119 % increased risk of mental health problems. Baudin et al (2021). found significant associations between e.g. aircraft-noise induced annoyance and the use of anxiolytic-hypnotic sedatives.

In addition, aircraft-noise induced annoyance has been found to trigger measurable **physiological stress** (increased stress hormone levels, blood pressure and heart rate) and is in turn a risk factor for hypertension (cf. e.g. Hahad et al., 2019). This is especially true in the event of high exposure at night-time (cf. e.g. Haralabidis et al., 2008, Eriksson et al., 2010, Babisch et al., 2013; Baudin et al., 2021.; Benz et al., 2022). Thus, Dimakopoulou et al. (2017) found that the OR for hypertension per 10 dB(A) L_{night} increase was 2.63 (95 % CI 1.21 to 5.71).

In fact, the WHO assessment concluded that **aircraft noise at a level above 45 dB(A) L_{den} is associated with adverse health effects** (WHO, 2018:61), which corresponds to a high-annoyance level of approximately 10 % (cf. table above). It follows that there is enough evidence in the literature that there is an increased risk of adverse health effects at a level of 10% HA (WHO, 2018).

The literature review conducted as part of the ANIMA project (Kranjec et al., 2019) raised questions regarding the relationship between noise induced high annoyance and health outcomes. Specifically, it questioned whether high annoyance directly leads to health issues or whether it serves as a mediator, and conversely, whether existing health problems exacerbate annoyance. They found that few studies focus on the nature of the link between aircraft noise-induced annoyance and health outcomes. Furthermore, they noted that these studies employ varied methodologies, rendering it challenging to quantify reliable and generalisable results. Most of the evidence is derived from cross-sectional studies, limiting the ability to explore causal pathways. Consequently, additional research is necessary. Nonetheless, these studies suggest that annoyance may indeed mediate the relationship between aircraft noise exposure and certain adverse health outcomes. Benz et al. (2022) drew a similar conclusion. Also, whilst high annoyance may contribute to the onset and maintenance of health conditions such as depression, the opposite may also be true: those suffering from health conditions may be more likely to experience high aircraft noise induced annoyance (Benz et al., 2022).

<u>Methodology used for measuring HA</u>

There exists abundant evidence establishing a link between reported (high) annoyance and exposure to environmental noise. However, due to the subjective nature of this response to noise, it cannot be objectively measured. Noise-induced high annoyance can only be ascertained approximately when individuals affected report their experience. Hence, it is appropriate to refer to it as *reported* high annoyance.

Reported high annoyance can be assessed using either a 5-point or 11-point scale developed by the International Commission on Biological Effects of Noise (ICBEN) (Fields et al., 2001). For example, the NORAH-study focussed on aircraft noise-induced annoyance in residential areas within the 40 dB(A) contour around Frankfurt airport (both $L_{pAeq06\text{-}22h}$ and $L_{pAeq22\text{-}06h}$) (cf. Schreckenberg 2017, 2019). The assessment was conducted during three separate years: 2011 (before the opening of a new runway), 2012 and 2013 (after opening new runway + night curfew). It used a survey questionnaire based on the (verbal) ICBEN 5-point scale that included the key question "*Thinking about the last 12 months, when you are here at home, how much does noise from aircraft bother, disturb or annoy you*". (cf. ISO/TS 15666:2003).

Respondents were requested to rate the annoyance caused by aircraft noise on a scale from 1 (not at all annoying) to 5 (extremely annoying):

1. Not at all
2. Slightly
3. Moderately
4. Very
5. Extremely

The top two tiers of the annoyance scale (4. Very, 5. Extremely) were used to determine the percentage of high annoyance (% HA), as per the ICBEN recommendation, with tier 4 (very) weighted by a factor 0.4. This equates to a cutoff set at 60 % of the entire scale's length.

Conversely, the HYENA (Hypertension and Exposure to Noise near Airports) study (Babisch et al., 2013) used the non-verbal 11-point ICBEN scale ranging from 0 (not at all) to 10 (extremely) (Fields et al. 2001). This numerical scale is designed to assess annoyance levels associated with environmental noise exposure without relying on verbal descriptors. Though there are no recommendations on the cutoff for high annoyance, the top 3 categories are commonly considered to indicate high annoyance (Brink et al., 2021).

Note that the different scales have different cutoff points for high annoyance. It follows that some correction is needed when comparing studies conducted with different ICBEN-scales (cf. e.g. Morinaga et al., 2021).

<u>Dose-response relations and affected populations</u>

The relationship between noise exposure and (reported) annoyance is described by empirical formulas, such as the Miedema dose-response curves and amended versions of the latter (WHO, DEBATS). The Miedema curves, derived from cross-sectional surveys conducted between 1967 and 1993 across Europe, North America and Australia, were adopted by the European Commission as standard curves for assessing and managing environmental noise in the European Union until March 2020. However, the Miedema curves were developed exclusively for adults. What is more, subsequent studies have highlighted their obsolescence, noting that people today tend to experience more annoyance at a given level of aircraft noise compared to three decades ago (e.g. Lercher et al., 2008; Guski et al., 2017, etc.). To address this, the WHO introduced updated curves (cf. Guski et al., 2017 and WHO, 2018), which were used in the revised version of the European Noise directive of March 2020 (Commission Directive 2020/367). However, these curves only take into account acoustical factors. More recently, the DEBATS study developed dose-response curves that factored in non-acoustical factors as well (Lefèvre et al., 2020), recognising their role in aircraft noise-induced annoyance. These curves predict higher levels of annoyance compared to the Miedema curves, but lower levels compared to the WHO curves upon which the new EU standard are based.

Comparison Miedema-WHO-Debats curves:

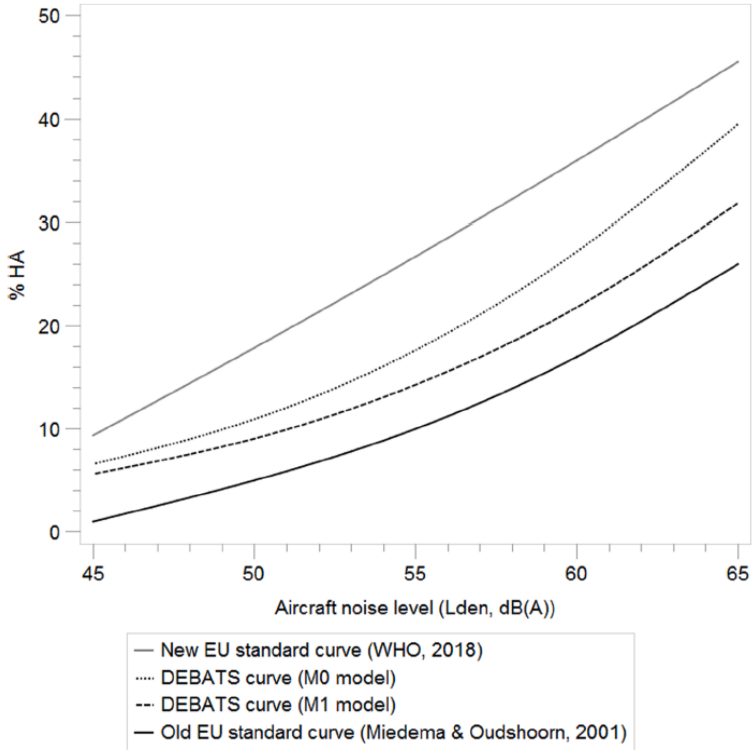

Figure 7: *Comparison Miedema-WHO-Debats curves (Source: Lefevre et al. 2020)*

Yet, note that the % of HA in the NORAH study was higher than would have been predicted not only based on the Miedema curves, but even WHO-curves, (cf. NORAH presentation on 25 September 2023 and also Haubrich et al., 2020), as shown in the following figure:

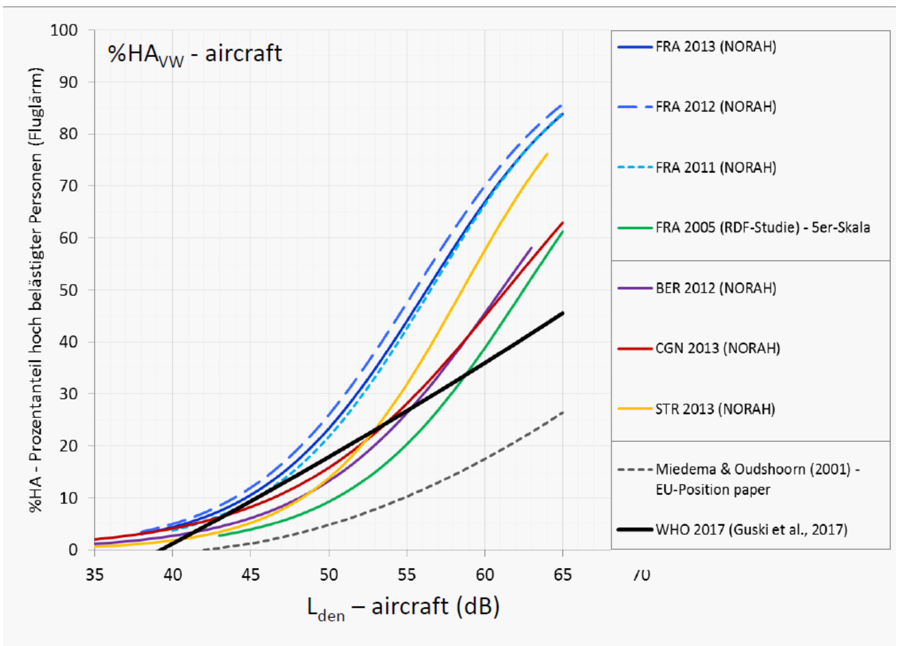

Figure 8: *Curves obtained in the NORAH-study vs. Miedema & Oudshoorn and WHO curves (Source: NORAH-presentation D. Schreckenberg 25.09.2023)*

In light of the above as well as the fact that the WHO curves are the ones that are used for the EU standard, the following discussion will be based on the WHO curves.

As regards Brussels airport, ENVISA (2023) calculated the percentage of HA population within the L_{den} contours 45 dB(A) to 75 dB(A) for the year 2019 using the dose-effect relationship from the WHO. They arrived at a total of 220 000 HA residents.

Air vs road and railway noise

Interestingly, research has also shown that a **distinction** needs to be drawn between annoyance caused by road, rail and air traffic (Gezondheidsraad, 1994; Miedema & Oudshoorn, 2001). Thus, the degree of annoyance reported for the same noise level averaged over longer periods (e.g. L_{den}) for air traffic is higher than for both road and rail traffic.

New independent studies conducted since the WHO assessment in Switzerland (SiRENE study; Brink et al., 2019), Austria (Innsbruck; Lechner et al., 2019), and Germany (LIFE study Leipzig/Halle airport; Starke et al. 2023) have confirmed that aircraft noise has a higher effect on noise-induced annoyance compared to rail or road traffic noise at the same L_{den}.

Figure 9 below shows an empirical relationship between the percentage of people reporting being seriously annoyed and the noise level to which they are exposed.

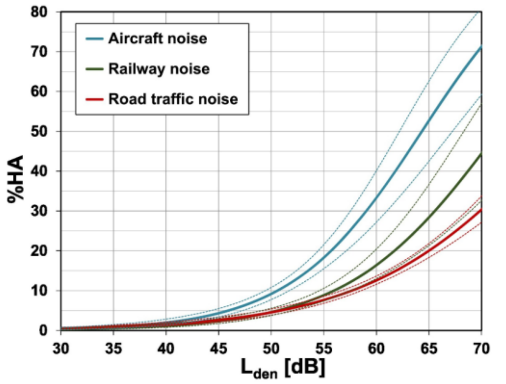

Figure 9: *Empirical relationship between the percentage of people reporting being seriously annoyed and the noise level to which they are exposed (Source: Brink et al. 2019)*

Complicating factors

It follows that the **relationship between perceived annoyance and noise level is complicated by several factors** and that energy-based indicators of noise exposure such as L_{den} cannot fully account for the annoyance and health impacts associated with aircraft noise.

First, the most likely explanation for the higher impact of aircraft noise on annoyance compared to road traffic is its **higher intermittency**, characterised by distinct peaks in sound pressure levels that clearly stand out from the background noise and punctuate quieter intervals (Brink et al., 2019). A pivotal aspect lies in the difference in sound pressure levels between a specific sound and all other concurrent noises perceptible at the same time, commonly referred to as the “background levels” (cf. De Coensel et al., 2009). This phenomenon also helps explain why an increase in the number of flyovers can correspond to a rise in reported noise-induced annoyance, even if the noise level attributable to each individual aircraft movement is lower (due to quieter aircraft) and, consequently, the L_{den} remains unaffected or only marginally so. Thus, the **nature of the noise pattern** is a critical factor (Sato et al., 1999 ; Roberts et al., 2003; Quehl et al., 2017).

Another key factor to take into account is the heightened prominence of these sound pressure peaks during night-time, when the background noise levels tend to be lower. The metric used to quantify the extent to which a noise event stands out from the background is known as **the intermittency ratio** (IR) (Wunderli et al., 2016). It's unsurprising that, with lower background noise levels during night-time, individual sound events such as aircraft flyovers become more perceptible, leading to an increased intermittency ratio. However, this coincides with the time of day when individuals most require rest and sleep. Both observations account for the fact that higher levels of annoyance are reported at night (cf. e.g. Bartels et al., 2022). In fact, many studies have shown variations in annoyance levels depending on the **time of day** (Hume et al., 2003; Hoeger, 2004; Baudin et al., 2021; Bartels et al., 2022).

Similarly, **changes** in noise exposure, which may be induced by delays (see e.g. Zidajic, 2022) or, in the longer term[15], by new aircraft trajectories, new runways, etc., increase the annoyance reported (Babisch et al., 2009; Bartels et al., 2022). The effects of such changes have been the subject of studies such as those by Brink et al. (2008), Brown & van Kamp (2009), Quehl et al. (2017), Schreckenberg et al. (2017), and Nguyen et al. (2020). They indicate that the reactions of people living near airports to sudden changes in noise exposure are more intense than would be anticipated solely based on acoustic considerations. For instance, the level of annoyance reported after the opening of a new runway (accompanied by an increased number of flyovers) tends to be higher compared to airports experiencing similar exposure levels but no such change.

Crucially, not only the physical characteristics of the noise, but also non-acoustic factors related to individual characteristics and the social context come into play (cf. e.g. Flindell & Witter, 1999; Miedema & Vos, 1999; Guski, 2017a; Bartels, 2022, ENVISA, 2022; Levere at al., 2022). These factors encompass previous experience with noise, individual sensitivity to noise as well as one's perception or stance toward the source of the noise: a favourable attitude towards the source of the noise, such as recognising its significance for the local economy, tends to mitigate annoyance. Conversely, a negative attitude – stemming from concerns like apprehension about plane crashes or potential devaluation of property, as well as social disputes related to the source of noise in the case of airport expansion projects – exerts the opposite effect, heightening annoyance levels (cf. e.g. Miedema & Vos, 1999; van Kamp et al., 2004; Bartels et al., 2022, ENVISA, 2022). Thus, in the study by Quehl et al. (2021), the short-term annoyance response in children was not affected by night-time aircraft noise, but rather by non-acoustic factors such as aircraft-induced fear.

The negative attitude towards the noise source is exacerbated by a sense of being forced to take certain actions, such as closing windows even in warm weather to block out the noise, or pausing conversations when aircraft pass overhead. Additionally, feelings of helplessness arise when the measures taken prove ineffective in blocking out the noise, coupled with a pervasive sense of lack of control that can be enhanced by being excluded from the decision-making process that will affect one's exposure to aircraft-noise. This sense of lack of control can be mitigated by fostering inclusive **participation of all stakeholders**. This entails ensuring that all concerned citizens are represented in the decision-making process, thereby empowering them to contribute to shaping policies and actions affecting noise exposure.

It follows that effective communication is paramount, as it enhances residents' trust in the authorities and the airport management. Indeed, non-acoustic factors such as **trust in the authorities** and **perceived fairness** also play a pivotal role (cf. Hauptvogel et al., 2021; Bartels et al., 2022). This, in turn, entails residents' belief that the authorities engage in transparent and truthful communication, take the concerns of residents seriously and actively

[15] Cf. Janssen & Guski (2017) call airports with sustained change of aircraft movements "high-rate change airports", as opposed to "low-rate change airports".

implement all possible measures to minimise unnecessary noise, while equitably distributing the noise burden among affected parties. Crucially, with respect to Brussels Airport, the ENVISA report notes that

> " *arbitrary changes to overflight patterns in recent years have led to an exacerbation of the public profile and aircraft noise perceived by neighbouring populations*. *This situation has been exacerbated by the fact that the noise generated by BRU has been raised at a political level. The rules, limitations and rationale for these decisions and related operational rules were not sufficiently explained to stakeholders. The same applies to reports on compliance by operational stakeholders. This has led to a loss of confidence and credibility with official stakeholders and the general public" (*ENVISA 2022: 15)."

Lastly, poor satisfaction with the living environment and a pessimistic outlook regarding the neighbourhood's quality of life are also factors that have been associated with high annoyance (cf. Lefevre et al., 2020).

<u>Summary</u>

Internationally standardised scoring methods reveal that elevated noise is associated with self-reported **high annoyance**, which includes cognitive, emotional, and behavioural aspects and is considered to be an **early warning for adverse health effects**. More specifically, reported high annoyance has been associated with increased risk for mental health outcomes, including depression and anxiety. Residents with high annoyance scores have significantly higher levels of physiological stress, which is a risk factor for hypertension. The WHO assessment concluded that 40 dB(A) L_{den} corresponds to a 1.2 % high annoyance (HA) prevalence. At 50 dB(A) (L_{den}), HA prevalence increases to 17.9 %, and at 60 dB(A) (L_{den}) to 36 % HA. At the same average noise level (L_{den}), a higher fraction of the population reports high annoyance in relation to aircraft noise compared to road traffic. This is probably due to the higher intermittency of aircraft noise and the nature of its noise pattern.

1.4 Self-reported and measured sleep disturbance

1.4.1 Sleep function

Sleep is an essential process that allows our bodies as well as our minds to recuperate, thus playing a key role in maintaining both physical and mental health. The immediate after-effect of poor sleep quality is of course that it affects performance during the day (sleepiness, poor ability to concentrate, irritability ….). However, insufficient sleep and poor sleep quality also have long-term effects, as a lack of adequate sleep contributes to various chronic conditions over time, such as type 2 diabetes, cardiovascular disease and depression. The reason is that sleep is an essential process during which metabolic waste products are cleared from the body, specific hormones are secreted and our memory is consolidated. This section will broach these different functions of sleep and the consequences of lack of sleep. For more detailed information, see e.g. Verbraecken et al. (2020), Garofalo et al. (2020) and Basner (2022).

Sleep plays a significant part in regulating energy **metabolism**. During sleep, the body undergoes essential metabolic processes that help maintain energy balance and support overall health. These include processes such as hormone regulation, cellular repair, and restoration of glycogen stores. Conversely, lack of sleep and disturbed sleep can disrupt these processes, resulting in metabolic changes that have been associated with obesity and type 2 diabetes, i.e. lower glucose tolerance, greater insulin resistance and increased appetite (cf. e.g. Iyegha et al. 2019).

In addition, sleep is closely intertwined with our **immune system**. It has been shown that numerous cytokines and chemokines, which play a role in immune responses to infection, affect sleep. Thus the cytokines IL-1 and TNF-α, on which extensive research has been conducted, have been shown to regulate sleep, e.g. by increasing non-rapid eye movement (NREM) sleep, whilst TNF-α and IL-1β are believed to mediate increased sleepiness, reduced appetite and tiredness in the event of an infection. During sleep, the plasma levels of anti-inflammatory hormones cortisol, adrenaline and noradrenaline go down, whilst prolactin and melatonin levels, which are not only involved in cell growth but also in the activation and production of proinflammatory cytokines, go up, thus contributing to immune activation. Immune cells too are circadian regulated and peak at the onset of sleep. Given this intricate connection between sleep and the immune system, it follows that lack of sleep alters the body's immunological and **inflammatory response**. A weakened immune system and elevated levels of inflammation have been linked to conditions such as **cardiovascular disease** (cf. e.g. Boyalla et al. 2023).

Objectively measured (with actigraphy) short sleep duration and poor sleep quality were prospectively correlated not only with all-cause cardiovascular mortality, but also with **cancer** mortality in a large database (Saint-Maurice et al., 2023), viz. a cohort of 88 282 adults (40-69 years) in UK Biobank who wore a wrist-worn triaxial accelerometer for 7 days. Similarly, self-reported sleep disorders were recently prospectively correlated with a risk of all-cause cancer. (Li et al., 2023). It is also believed that poor quality sleep could contribute, indirectly, to breast cancer (Verkasalo et al., 2005). Indeed, poor sleep has been associated with a number of risk factors for cancer, which include metabolic dysfunction, a weakened immune system and chronic inflammation, which were mentioned above, but also a disrupted circadian rhythm and impaired cellular and DNA repair (Chen et al., 2021; Song et al., 2021).

In addition, a recent systematic review and meta-analyses by Sun et al. (2021) demonstrated a high prevalence of sleep disturbances in **chronic pain** patients. Poor sleep can trigger **glial overactivation**, which in turn prompts a low-grade inflammatory response. This can contribute towards heightened sensitivity to pain, which is typically observed in individuals suffering from chronic pain (Nijs et al., 2017). Therefore, chronic pain patients can benefit from conservative treatments including sleep management. There is also a significant link between glial activity and **neurodegenerative diseases** such as Alzheimer's disease, Parkinson's disease, amyotrophic lateral sclerosis, and multiple sclerosis (Garofalo et al., 2020; Belojevic, 2023). Consequently, poor sleep quality has been associated with an increased risk of neurodegenerative disease and is believed to contribute to their progression.

Also, poor quality sleep can result in disruptions in hormonal regulation and neurotransmitter release, significantly affecting mood regulation and potentially fostering the onset of depression and other mood disorders (Novati et al., 2008; Longordo et al., 2009; Daut & Fonken, 2019; Lefter et al. 2022). Various neurotransmitters, including serotonin, noradrenaline, and acetylcholine, are pivotal for maintaining emotional balance. Serotonin helps regulate mood, sleep, and appetite, and low levels of serotonin have been strongly linked to **depressive symptoms**. Noradrenaline is involved in the body's stress response and plays a crucial role in mood regulation. Imbalances in noradrenaline levels have been associated with mood disorders such as depression and **anxiety**. Acetylcholine also affects mood and cognition, and disruptions in acetylcholine signalling have been implicated in mood disorders and cognitive decline.

Apart from inducing physical benefits, sleep also plays a vital role in **cognitive functions** such as memory, learning, and emotional regulation. Research has shown that during sleep, stored memories are reactivated, leading to improved memory consolidation. Additionally, sleep has also been shown to play an essential part in priming the brain to acquire new information effectively. Thus, experiments have revealed that poor sleep quality not only affects people's ability to retain new information, but also weakens the formation of memory traces. Furthermore, sleep plays a part in fostering creativity and insight. Studies have shown that individuals exhibited better problem-solving abilities after a period of sleep compared to an equivalent period of wakefulness and that more than 80 % of individuals reported finding solutions to problems they encountered during the day while asleep (Verbraecken et al., 2020). Moreover, there is a significant link between dream content and the emotions experienced during wakefulness. Sleep is involved in regulating emotions by attenuating the emotional component linked to memories. Conversely, lack of sleep often results in increased heightened irritability and emotional instability, emphasising the importance of quality sleep in maintaining emotional balance (cf. e.g. Verbraecken et al. 2020).

Considering the profound impact on both mental and physical well-being that disrupted sleep can have, it's hardly surprising that the WHO (2009) considers **sleep disturbance as the most serious health consequence of noise exposure**, directly impacting overall quality of life.

1.4.2 Sleep structure and quality

Importantly, the recuperative power of sleep is not only determined by its duration, but also by its quality (cf. e.g. Basner et al., 2006; Basner et al., 2010a; Basner, 2022).

Sleep is a complex process that consists of several stages, broadly classified into two main types: Rapid Eye Movement (REM) sleep and Non-Rapid Eye Movement (NREM) sleep. NREM sleep can be further divided into light sleep (stages N1 and N2) and deep sleep (stage N3), also known as slow wave sleep (SWS). These stages contribute to varying degrees to the body's recovery during sleep, with REM and SWS sleep being crucial for recuperation, emotional processing and memory consolidation. In contrast, stage N1 contributes the least to the overall recuperative effect of sleep.

When a person falls asleep, they progress through these different stages of sleep. Initially, they enter stage N1, which typically lasts only a few minutes. They are initially still awake, but drowsy and their brain activity, heart rate and muscle tension gradually decrease as they drift into stage N2. At this stage, they can still easily be woken up during a few dozen minutes, whilst the brain activity continues to slow down and they enter deep sleep (also known as SWS). During SWS, the brain becomes increasingly less sensitive to external stimuli, making it more difficult to wake up. After a period of SWS, the sleeper transitions to light sleep before entering a phase of REM sleep. During REM sleep, the brain becomes as active as in stage N1 and vivid dreams commonly occur. REM sleep typically lasts for just a few minutes, marking the end of the first sleep cycle. Throughout the night, individuals experience a total of four or five sleep cycles, each lasting for about 90 minutes. However, these cycles vary in composition. Early cycles have longer stretches of SWS, whilst later cycles have longer stretches of REM sleep.

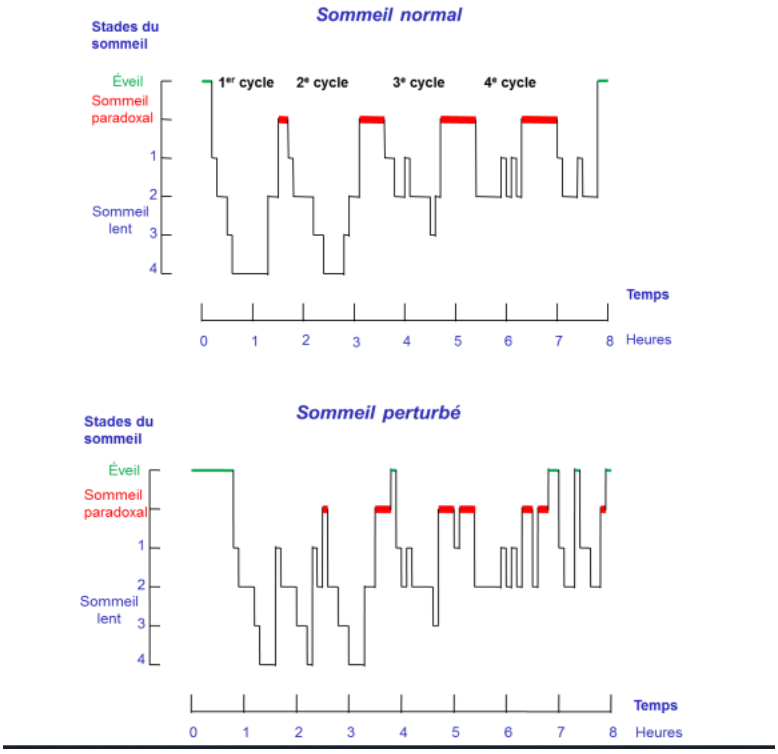

Figure 10: *Hypnograms of normal sleep and sleep disturbed by noise (Source: Nassur 2018: 27)*

As illustrated in the figure above, disrupted sleep can result from awakenings, but also from transitions between different sleep stages (such as transitioning from SWS to lighter stages), as well as "**arousals**", which involve brief activations in EEG and EMG readings (cf. e.g. Basner et al., 2006, 2010b; Basner, 2022). It's important to note that people with normal sleep patterns also undergo sleep stage changes (averaging around 52 per night in normal sleep; Basner et al., 2006) and arousals (approximately 20/hour in normal sleep, cf. Basner et al., 2006). However, the increased frequency of these occurrences may lead to fragmented sleep patterns. Sleep disruption can occur even if the individual doesn't fully wake up.

Given this fact and the observation that most awakenings are too brief for individuals to fully regain waking consciousness, studies on the effects of aircraft noise on sleep using the "push button methodology" (where the study participants are required to push a button each time they wake up) have low sensitivity.

Of course, other methods have been used to assess the quality of sleep and sleep disturbance, using both subjective and objective measurements. Subjective assessments typically involve the use of questionnaires and sleep logs to collect self-reported data, whilst objective measurements of sleep physiology parameters (and therefore the quality of sleep) can be obtained through **polysomnography**. Polysomnography generates signals that enable the differentiation of different sleep stages (Basner, 2022) and is considered the gold standard for assessing sleep quality. Another method, **actigraphy**, is less expensive and less invasive. Actigraphy measures body movements and heart rate to provide information on sleep-wake patterns. However, it is generally considered less accurate than polysomnography, as it tends to overestimate sleep time and underestimate wake time (Marino et al. 2013). Despite this drawback, actigraphy remains valuable, particularly in situations where polysomnography may not be feasible or practical.

Studies based on these methods reveal that the effects of noise on sleep manifest across a variety of parameters, including sleep onset latency, frequency of arousals, duration of wakefulness, number of sleep stage transitions, and frequency of body movements. Sleep disturbances are observed when these parameters deviate from normal values in the presence of noise and return to normalcy once the noise dissipates (Nassur, 2018).

1.4.3 The influence of average noise levels vs noise events on sleep quality

Disrupted sleep can stem from various factors, including age-related alterations in sleep patterns and underlying health conditions, and of course external factors such as exposure to noise. Most importantly for the topic under discussion, this includes aircraft noise.

The table below presents the assessment from the 2009 WHO report regarding the correlation between average exposure levels to noise and sleep outcomes.

***Table 2**: Effects of different levels of night-time noise on the population's health (source: WHO 2009:108)*

Average night noise level over a year $L_{night, outside}$	Health effects observed in the population
Up to 30 dB	Although individual sensitivities and circumstances may differ, it appears that up to this level no substantial biological effects are observed. $L_{night,outside}$ of 30 dB is equivalent to the NOEL for night noise.
30 to 40 dB	A number of effects on sleep are observed from this range: body movements, awakening, self-reported sleep disturbance, arousals. The intensity of the effect depends on the nature of the source and the number of events. Vulnerable groups (for example children, the chronically ill and the elderly) are more susceptible. However, even in the worst cases the effects seem modest. $L_{night,outside}$ of 40 dB is equivalent to the LOAEL for night noise.
40 to 55 dB	Adverse health effects are observed among the exposed population. Many people have to adapt their lives to cope with the noise at night. Vulnerable groups are more severely affected.
Above 55 dB	The situation is considered increasingly dangerous for public health. Adverse health effects occur frequently, a sizeable proportion of the population is highly annoyed and sleep-disturbed. There is evidence that the risk of cardiovascular disease increases.

More recently, the WHO assessment (2018) concluded that 40 dB(A) L_{night} is associated with high sleep disturbance (HSD) among 11.3 % of study participants. Using pooled data from 6 studies including a total of 6371 participants, the proportion of self-reported HSD in relation to L_{night} could be estimated. At 50 dB(A) (L_{night}), these figures increased to 19.7 %, whilst at 60 dB(A) (L_{night}), they rose to 32.3 % HSD.

Table 3: *Association between exposure to aircraft noise (L_{night}) and sleep disturbance (%HSD) (source: WHO 2018:70)*

L_{night}	%HSD	95% CI
40	11.3	4.72-17.81
45	15.0	6.95-23.08
50	19.7	9.87-29.60
55	25.5	13.57-37.41
60	32.3	18.15-46.36
65	40.0	23.65-56.05

This led the WHO to "*strongly recommend reducing noise levels produced by aircraft during night-time below 40 dB(A) L_{night}, as night-time aircraft noise above this level is associated with adverse effects on sleep*." (WHO 2018).

An update to the WHO assessment carried out by Smith et al. (2022) found comparable sleep disturbance at low night-time noise levels, corresponding to the WHO noise limit recommendations for night-time noise, but also found deviations suggesting that the risk of sleep disturbance in populations exposed to high levels of aircraft noise may in fact be higher than previously assessed.

Interestingly, the WHO admits that the annual average L_{night} of 40 dB(A) it recommends is in fact not fully protective:

"*The GDG acknowledged that the guideline recommendation for L_{night} may not be fully protective of night health, as it implies that around 11% (95% CI: 4.72–17.81) of the population may be characterised as highly sleep-disturbed at the recommended level. This is higher than the 3% absolute risk L_{night} considered for setting the guideline level*."

Indeed an annual average L_{night} of 40 dB(A) still allows 29 overflights during an 8-hour night of 60 dB(A) $L_{A,max}$ (= awakening threshold at the façade, cf. WHO 1999), giving an Awakening Risk (S1) of 53% according to Basner and McGuire's formula.

Crucially, the WHO guidelines are formulated in terms of **average energy levels** (L_{night}). Yet, recent research based on polysomnography, actimetry, and questionnaires has provided substantial evidence that the effect of noise on the quality of sleep depends on several factors, namely the **number of noise events (frequency of flyovers), their acoustic properties, and their distribution (i.e. timing and the intervals between them).** (cf. e.g. Nassur et al., 2019a ; Casario et al., 2020).

Indeed, as pointed out in section 1.3.2 above, aircraft noise differs from e.g. road traffic noise in the sense that it is **intermittent noise**. This means that there are intervals of less noisy or even noise-free periods between consecutive noise events (cf. e.g. Basner et al., 2017). Additionally, Wunderli et al. (2016) introduced the concept of the **intermittency ratio** as a relevant factor (cf. section 1.3.2 above). Consequently, it's not surprising that several studies have shown that while particularly loud flyovers are more likely to cause people to wake up, the difference between the general background noise and the peak noise level during the flyover also plays a role (cf. Brink et al., 2019). Higher intermittency ratios, which are typically found during night-time, have been associated with increased frequency of awakenings among participants.

Along the same lines, in their systematic review from 2018, Basner & McGuire (2018) found a significant positive association between indoor maximum noise levels of single events and the probability of awakening or transitioning to sleep stage 1 (for all transportation modes). Also, indoor noise levels associated with a non-zero probability of additional awakening range from 33 to 38 dB(A), which is consistent with previous findings in Passchier-Vermeer et al. (2002) and Basner et al. (2006). Basner & McGuire also conducted a pooled analysis of polysomnographic studies on the acute effects of aircraft noise on sleep and found that the unadjusted odds ratio for the probability of awakening for a 10 dB(A) increase in the indoor L_{max} was significant (OR 1.35; 95 % CI 1.22–1.50).

The duration of the waking state, and therefore the likelihood that it can be recalled the next morning (and result in annoyance), is determined by the intensity ($L_{A,max}$/SEL) of the aircraft noise (Basner et al., 2006).

Based on these observations, it follows that single noise event parameters, i.e. the frequency of flyovers and $L_{A,max}$/SEL, as well as the number of exceedances of a given threshold, are more important indicators than average energy based indicators.

Yet, another important consideration is the fact that it is more harmful for sleep to be disturbed more than a dozen times by noise just above the internationally established standards, than to be shaken once or twice by louder noise (Gezondheidsraad, 2004). This is because during sleep, the human organism reacts to every noise event (passing aircraft, lorries). A monotonous noise can certainly be annoying, but it is much less of a problem than a **series of sound peaks** (as is the case when aeroplanes fly over).

Therefore, it may be more appropriate to collect data on the average number of noise events contributing to L_{night}. Thus, Basner et al. (2006, 2010a) add that though awakenings are short (15-45 sec) and are not recalled the next morning, they are related to long-term **cardiovascular conditions**. Noise events can affect the heart rate (cf. e.g. Nassur et al., 2019b), induce changes in posture and responses from the autonomic system. Sleepers may become agitated, potentially going into another stage of sleep, e.g. from SWS to a lighter stage of sleep, or even waking up. Noise-event induced **arousals**, sleep stage changes and awakenings are liable to reduce the duration of SWS and/or REM sleep, thus affecting its recuperative power.

Also, sound events occurring at the onset of sleep have a lesser influence on sleep quality compared to those occurring towards the end of sleep. This is because during the latter part of the night, individuals typically spend more time in lighter sleep stages, which are more susceptible to disruption from external stimuli.

It follows that the **distribution of the sound events over the night**, i.e. their timing and intervals between them, is also an important factor to take into account (Passchier-Vermeer et al., 2003; Passchier-Vermeer et al., 2007; Griefahn et al., 2008).

From the above discussion, it can be concluded that the most important indicator for assessing the impact of night flights on sleep is the frequency with which the maximum level reached by each flight exceeds 60dB(A) $L_{A,max}$ and the extent to which this threshold is exceeded.

1.4.4 Noise-induced effects on sleep for different population groups

As discussed above, sleep fulfills critical physiological functions that cannot be replaced, including memory consolidation and vital immunological and endocrine processes. Unsurprisingly, the WHO (2009) considers sleep disturbance as the most serious health consequence of noise exposure, directly impacting overall quality of life.

Aircraft noise at night can have significant effects on sleep quality and overall well-being across all age groups. Still, the specific impact may vary depending on factors such as age-related changes in sleep patterns, individual susceptibility to noise, and underlying health conditions.

The **amount of sleep** required to maintain good health depends on a number of individual factors. One of them is age. Thus, whilst toddlers require 11-14 hours of good-quality sleep per day (including naps), adults require a good-quality sleep duration of **8 hours**. School-age children and teenagers range in between (Iglowstein et al., 2003; McLaughlin Crabtree, 2009)[16]. Thus, the American Academy of Sleep Medicine recommends that children aged 6-12 years sleep 9-12 hours per night on a regular basis[17].

Children are a particularly vulnerable group, as their cognitive and physical development requires more sleep. In fact, lack of sleep is liable to cause long-lasting damage, as suggested by Yang et al. (2022). Their propensity score matched, longitudinal, observational cohort study among 9–10-year-olds in the US found evidence for the long-lasting effect of insufficient sleep on neurocognitive development during early adolescence. Brain imaging conducted at both the study's onset and two years later unveiled disparities in brain structure and function between the group experiencing insufficient sleep and those with adequate sleep. These findings suggest that sleep plays a pivotal role in shaping learning and behaviour through distinct alterations in brain functioning.

As regards sleep disturbance caused by aircraft noise, the NORAH study, which was conducted among 1209 children aged 7-9 (2^{nd} grade) living in the vicinity of Frankfurt airport, found that 20 % of the children exposed to noise levels above 55 dB(A) stated that they had "never" slept well in the previous week, whilst this was the case for 15 % of those exposed to noise levels below 47 dB(A) (Bergström et al., 2015).

1.4.5 Summary

In summary, sleep disturbance is the most important adverse health outcome caused by aircraft noise at night. Sleep serves an essential physiological function that cannot be replaced (memory consolidation, key immunological and endocrine processes). Its restorative power is determined by sleep duration as well as sleep quality, both of which are affected by the intensity of the noise as well as the number of flights and their temporal distribution. Sleep disturbance is caused by both awakenings, arousals, and sleep stage changes. Short sleep duration and poor sleep quality have both short-term (poor performance during the day and irritability) and long-term effects (contributing to chronic conditions such as obesity, type 2 diabetes, cardiovascular diseases, chronic pain, chronic fatigue syndrome, neurodegenerative diseases, depression and possibly, indirectly, breast cancer). Self-reported sleep disorders were recently prospectively correlated with a risk of all-cause cancer. Objectively measured (with actigraphy) short sleep duration and poor sleep quality were also prospectively correlated with all-cause cardiovascular and cancer mortality in a large database. Polysomnographic studies, which objectively measure parameters of sleep physiology, showed adverse effects of nocturnal aircraft noise on sleep duration and quality. This supports the associations between self-reports of sleep disturbance and nocturnal aircraft noise. The WHO assessment concluded that 40 dB(A) L_{night} is associated with high sleep disturbance (HSD) among 11.3 % of study participants. At 50 dB(A) (L_{night}), these figures increased to 19.7 %, whilst at 60 dB(A) (L_{night}), they rose to 32.3 % HSD. Yet, crucially, recent research has shown that the frequency of the flyovers is of paramount importance for the quality of sleep. Children are a more vulnerable group, as their cognitive and physical development requires more sleep. As for adults, in general, a good-quality sleep duration of 8 hours is ideal.

[16] https://www.sleepfoundation.org/how-sleep-works/how-much-sleep-do-we-really-need
[17] https://aasm.org/resources/pdf/pediatricsleepdurationconsensus.pdf

1.5 Indirect health effects

Due to these stressors, the following adverse effects may emerge **indirectly**:

1.5.1 Cognitive impairment

Noise exposure can significantly complicate both learning and working environments. Noise has been shown to impair concentration and focus, leading to decreased attention spans, particularly in settings where cognitive demands are high, such as classrooms and workplaces. Additionally, noise can interfere with verbal communication, which is crucial in business settings for effective collaboration, ensuring safety protocols are understood, and conveying information accurately. In educational environments, clear communication is essential for effective knowledge transfer and student comprehension. Furthermore, noise can disrupt social interactions, making it difficult for individuals to engage in meaningful conversations or establish connections with others.

Research has consistently demonstrated a negative association between aircraft noise exposure and the cognitive development of **children**. In their systematic review of the evidence from the literature (which included the RANCH study on the effects of road and air traffic noise on children's cognition[18]), the WHO (2018) *concluded that "most of the studies (10 of 14) demonstrated a statistically significant association or at least demonstrated a trend between higher aircraft noise exposure and poorer reading comprehension".*

A cross-sectional study conducted within the framework of the NORAH study (Klatte et al. 2015, 2017) among >1000 second graders from 29 schools around Frankfurt/Main Airport found that there was a linear relationship between increasing exposure and three different indicators: ratings of quality of life, noise-induced annoyance, and reading performance. While noise exposure was less than 60 dB(A) (L_{Aeq}, 8-14) in all schools investigated (and thus significantly below the levels of previous studies), a 10 dB(A) and 20 dB(A) increase compared to the schools with the lowest exposure (39 to 46 dB(A) L_{Aeq}, 8-14) was associated with, respectively, a 1- and 2-month delay in reading acquisition. Conversely, they were unable to identify a link between noise levels on the one hand, and phonological processing and episodic memory on the other. The following figure shows the exposure-response relationship for children's reading (the T score is the global reading score calculated for each individual child by averaging the standard scores of three subtests measuring fluency and accuracy of reading comprehension on the level of words, sentences, and text paragraph):

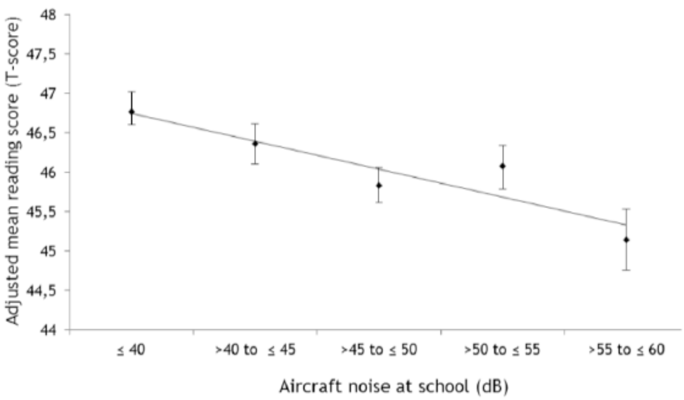

Figure 11: *exposure-response relationship for children's reading (Source: Klatte et al. 2015)*

[18] The RANCH (Road traffic and aircraft noise exposure and children's cognition and health) examined the effect of road traffic and aircraft noise exposure on the cognitive performance of 2844 children aged 9-10 from schools around three major airports in the Netherlands, Spain and the United Kingdom. It was conducted over a period that included the years 2003, 2005, and 2010.

Note that the WHO (1999) advises that the background sound level in classrooms should not exceed 35 dB(A) L_{Aeq}, whilst it should not exceed 55 dB(A) L_{Aeq} in outdoor playgrounds.[19]

1.5.2 Hypertension

Multiple studies have shown a link between prolonged exposure to road, rail or air traffic noise and hypertension.

In a cohort of 4854 participants from Stockholm County between 1992 and 2006, Pyko et al. (2018) observed an increase in the incidence of hypertension with rising aircraft noise levels, with a HR of 1.16 (95% CI 1.08 to 1.25) per 10 dB(A) increase in day-evening-night aircraft noise levels one year prior to the event, and of 1.16 (95 % CI 1.08 to 1.24) per 10-dB(A) increase in day-evening-night time-weighted average of aircraft noise levels five years preceding the event. Similarly, Dimakopoulou et al. (2017) followed 420 HYENA (Hypertension and Exposure to Noise near Airports, performed around 6 European airports) participants living near the Eleftherios Venizelos airport in Athens (Greece). This cohort study reported an increased incidence of hypertension with rising exposure to aircraft noise at night (OR 2.63, 95 % CI 1.21 to 5.71 per 10 dB(A) increase in L_{night}).

Yet, whilst the 2018 WHO review did not find the incidence of hypertension per 10 dB(A) L_{den} increment to be significantly increased, a more recent longitudinal cohort study of 1244 adults living near three major French airports (DEBATS in France) has confirmed an association between aircraft noise exposure and prevalent as well as incident hypertension. (Evrard et al. 2017, Kourieh et al., 2021). The association was significant for L_{day} (IRR 1.41, 95 % CI 1.07; to 1.85) and L_{night} (IRR 1.31, 95 % CI 1.01 to 1.71). Systolic and diastolic blood pressure increased with all noise indicators.

These findings are consistent with observations from cross-sectional studies and with proposed mechanisms for short-term impacts of nocturnal aircraft noise on cardiovascular diseases from experimental studies, including sleep disturbance, elevated blood pressure and stress hormone levels and impaired endothelial function. Mechanistic pathways known to damage the vascular system, such as vascular oxidative stress and activation of the sympathetic nervous system, may lead to the acute onset of a cardiovascular event.

Wojciechowska et al.'s (2022) study further reveals that a temporary reduction in noise, such as that experienced during the COVID-19 lockdowns, can reverse these adverse effects. Wojciechowska et al. (2022) focused on residents experiencing noise exposure levels of 61.7 dB(A) during the day and 55.4 dB(A) at night near the Krakow John Paul II airport in 2019. These exposure levels dropped to 47 dB(A) and 43.4 dB(A), respectively, during lockdown from April 2020. They found that "*The decline in noise exposure since April 2020 was accompanied with a significant decrease of noise annoyance, 24-hour systolic (121.2 versus 117.9 mmHg; P=0.034) and diastolic (75.1 versus 72.0 mmHg; P=0.003) blood pressure, and pulse wave velocity (10.2 versus 8.8 m/s; P=0.001)*". As pointed out by Hahad et al. (2022), "*the implications are clear enough to call on politicians and policymakers to bring (aircraft) noise exposure levels in line with the recommendations of the World Health Organization.*"

1.5.3 Cardiovascular diseases

Numerous studies have established a correlation between cardiovascular diseases and aircraft noise via noise-induced annoyance leading to e.g. heightened stress levels, and sleep

[19] The same background sound level in classrooms is set by the American National Standards Institute (ANSI).

deprivation (e.g. Correia et al., 2013; Hansell et al., 2013; Basner, 2014; Bączalska et al., 2022).

According to the WHO review (2018), there was a 9 % increased Risk Ratio (RR) for the incidence of Ischemic Heart Disease (IHC) (95 % CI: 4-15 %) per 10 dB(A) increment (L_{den}) in 2 ecological studies (evidence rated very low quality) (van Kempen, 2018). The NORAH study, based on insurance claims from over a million residents aged over 40, demonstrated that aircraft noise levels exceeding 60 dB(A) ($L_{Aeq,\ 24h}$) are associated with a heightened risk of myocardial infarction (Seidler, 2016b). However, it is noteworthy that the overall risk increase associated with road noise and railroad noise was estimated to be higher. Moreover, the NORAH study found that insurees exposed to low background noise ($L_{Aeq,24h}$ < 40 dB(A)) but $L_{A,max}$ > 50 dB(A) exhibited an increased risk of 7 % for stroke (cf. Seidler et al., 2018) and 6 % for heart failure (Seidler et al., 2016c).

As discussed above, exposure to aircraft noise has been linked to hypertension, a phenomenon that manifests early in the cardiovascular system. Importantly, this effect is considered reversible, as highlighted by Wojciechowska et al. (2022). Early-stage cardiovascular alterations triggered by noise exposure, including heightened stress hormone levels, oxidative stress, endothelial dysfunction, and arterial stiffness, are recognised as potentially reversible. Conversely, prolonged exposure to aircraft noise may lead to the development of established cardiovascular diseases, as corroborated by numerous epidemiological studies. Unlike the early-stage cardiovascular changes, these conditions are deemed irreversible, as emphasised by Bączalska et al. (2022).

Of course, people living in the vicinity of airports are usually exposed to both aircraft noise and air pollution (see also Beghelli, 2018), both of which have negative effects on the cardiovascular system. While the precise interaction remains unclear (Beelen et al., 2009), research has linked aircraft noise to a higher risk of mortality from myocardial infarction that is independent of air pollution and socioeconomic factors. For instance, a study by Huss et al. (2010) of the Swiss National Cohort supports an association between mortality by myocardial infarction and aircraft noise. Between 2000 and 2005, 4.6 million persons older than 30 years were followed. The strongest association was observed in the analysis fully adjusted to sociodemographic and geographical variables and PM_{10} air pollution levels and restricted to persons who had been exposed for 15 years or longer (HR 1.5 95 % CI 1.0 to 2.2). Their results could not be explained by socioeconomic position, the exact mechanisms remain unclear.

1.5.4 Mental health and depression.

Aircraft noise annoyance is associated with mental health problems. For instance, a meta-analysis conducted by Gong et al. (2022) suggests that noise-induced annoyance, rather than the noise levels themselves, may be a significant factor in contributing to mental health issues, making high noise-induced annoyance a potential mediator of the link between aircraft noise and mental health issues. They found that those experiencing high levels of noise-induced annoyance were approximately 1.23 times more liable to suffer from **depression**. They also found an approximately 55 % higher risk of **anxiety** in highly noise-annoyed people and an almost 119 % higher risk of **mental health problems** (based on Short Form (SF) or General Household Questionnaires (GHQ), albeit with high heterogeneity and risk of publication bias).

In the NORAH study (Seidler et al., 2016a), an inverse "U"-shaped association was found between aircraft noise and **depression**. An 8.9 % increase in the risk of depression per 10 dB(A) in $L_{Aeq,24h}$ was found, but a decrease was noted at higher noise levels, probably due to some sort of habituation. An alternative explanation could be that, in general, higher noise levels are obtained closer to the airport, where more people live that are, one way or another, professionally involved in airport activities.

1.5.5 Other effects.

Some studies suggest a possible association between exposure to aircraft noise and e.g., the incidence of **breast cancer** or **adverse birth outcomes** (cf. e.g. the review by Clark et al., 2020).

Based on insurance records of women aged >40 in the Frankfurt region, Hegewald et al. (2017) found an increased odds ratio for oestrogen receptor-negative breast cancer at 24-hour aircraft noise levels 55–59 dB(A) [OR 55–59 dB(A) 1.41, 95 % CI 1.04–1.90], but not for oestrogen receptor-positive breast cancers (OR 55–59 dB(A) 0.95, 95 % CI 0.75–1.20). They concluded that increased aircraft noise may be an etiologic factor for oestrogen receptor negative breast cancers.

As regards adverse birth outcomes, Nieuwenhuijsen et al. (2017) found very low-quality evidence for associations between aircraft noise and preterm birth, low birth weight and congenital anomalies.

A study by Wing et al. (2022) on 174,186 births near Los Angeles International Airport (LAX) suggest a synergistic effect, i.e. more than additive interaction for exposures to airport-related noise and traffic-related air pollution, on preterm birth in women who are also highly exposed to UFPs from aircraft. An adjusted odds ratio of 1.1 (95 % CI 1.01 to 1.19) was derived.

Large-scale epidemiological studies have concluded that exposure to road traffic noise can be linked to metabolic diseases including obesity and type 2 diabetes (cf. e.g. Ohlwein et al., 2019; Zuo et al., 2022; Veber, 2023). Liu et al. 2022 found a linear association between exposure to road noise and type 2 diabetes with a 6% increase in the risk of type-2 diabetes per 10 dB(A) increase. They observed a similar trend for exposure to aircraft noise (and railway noise), but added that these associations need to be further studied. Similar conclusions can be found in the overview by Basner et al. (2019).

However, the interpretation of the observational studies is often hampered by the multiple confounders for which the necessary information was not always available.

2 Aircraft pollutant emissions: exposure and health effects

According to the World Health Organization (WHO), exposure to air pollution (both indoor and outside) is one of the largest threats for global human health. The combined effects of ambient air pollution and household air pollution is associated with 7 million premature deaths annually (WHO, 2024). With 645 aircraft movements per day (Brussels Airport, 2022), the national airport in Zaventem has a significant impact on the air quality of the surrounding area. The total airport-related emissions are a mix of engine exhaust from airplanes, combined with emissions from ground service equipment and road traffic (passenger buses, baggage and food carts, container loaders, refilling trucks, cleaning and lavatory services, anti-icing vehicles, etc.). Emission levels depend on the engine conditions, the fuel type and operation modes (idling, taxi, take-off, climb-out, landing) (Mazaheri et al., 2011). Jet fuel is extracted from the middle distillates of crude oil (kerosene fraction) between the gasoline and diesel fractions, the most common fuels for civil aviation are of Jet A and Jet A-1 grades together with some additives (Masiol & Harrison, 2014). Emissions contain both gases and particles. The gaseous content of airliner engine exhausts mainly consists of N_2, O_2, CO_2, H_2O and a small fraction of residual products like NO_x, CO, SO_2, hydrocarbons and aerosol particles with organic and inorganic, volatile and semi-volatile components (Masiol & Harrison, 2014) (Figure 12). When released in the atmosphere, the emitted gases and compounds undergo further reactions, having an impact on the local and global environment and also affecting human health.

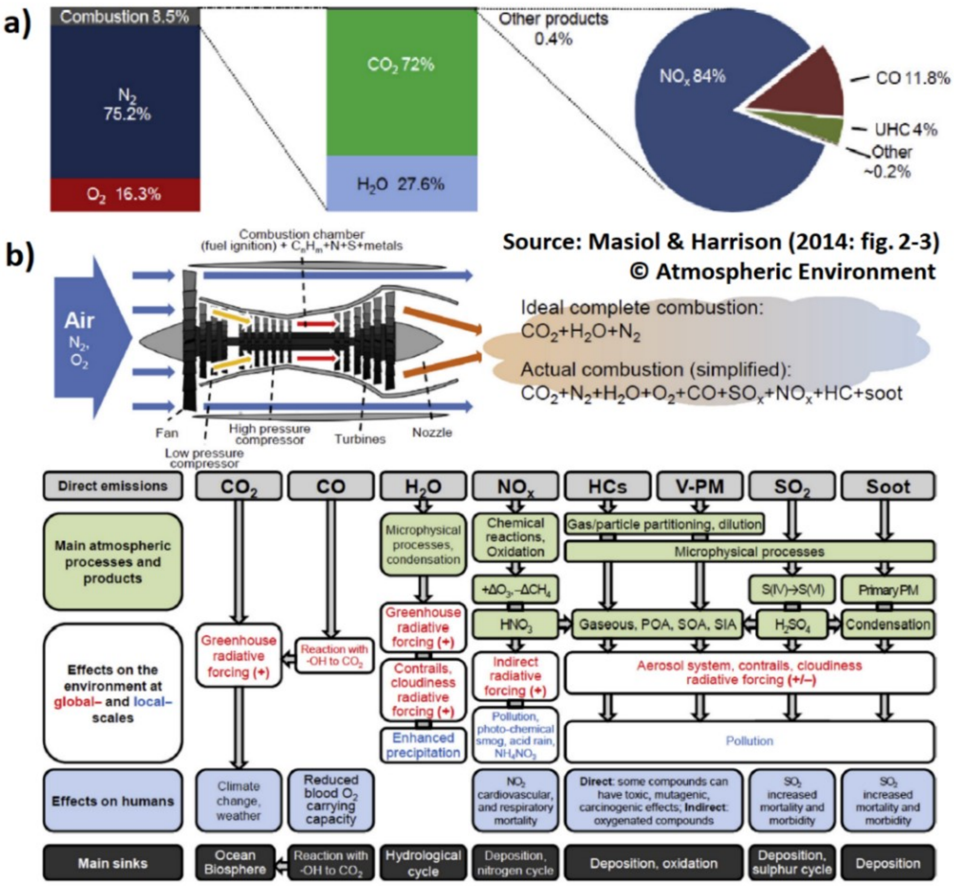

Figure 12: *a) Combustion products of an aircraft engine. b) Exhaust of a turbofan engine, with its ideal and actual combustion products before and after atmospheric processes. HCs: hydrocarbons. V-PM: volatile particulate matter. (Source: Masiol & Harrison, 2014: figs 2-3)*

Incomplete combustion of kerosene and other fossil fuels results in the formation of carbon-rich (>60 %) aromatic products like char and condensates (soot) (Bendtsen et al., 2021). Aircraft emissions from kerosene combustion can be expected to have carcinogenic effects, given that the kerosene lies between the distilled crude oil fractions of gasoline (exhaust IARC group 2B, "*possibly carcinogenic to humans*") and diesel (exhaust IARC group 1, "*carcinogenic to humans*") (IARC, 2013). Particulate aircraft emissions are dominated by very small Ultra Fine Particles (< 20 nm), which makes them stand out from other emission sources like road traffic. The nano-sized particles of jet emissions have a larger surface area and reach the lower airways upon inhalation, causing similar adverse health effects as exposure to diesel exhaust particles (Bendtsen et al., 2021). The particle surface area is the biologically most effective dose metric for the acute nanoparticle toxicity in the lungs (Schmid & Stoeger, 2016). Therefore, the number of these very small particles is more important than their weight.

2.1 Characteristics of aircraft pollutant emissions

Multiple fractions and components of aircraft emissions are of specific interest for human health.

2.1.1 Particulate matter: Ultra Fine Particles (UFPs)

Particulate matter is a designation for mixtures of solid particles and liquid droplets found in the air. Important fractions are coarse particulate matter PM_{10} (aerodynamic particle diameter ≤ 10µm) and fine particulate matter $PM_{2.5}$ (≤ 2.5 µm), these are often described in terms of mass distribution. The finest fraction consists of Ultra Fine Particles (UFPs; $PM_{0.1}$), representing aerosols with an aerodynamic particle diameter ≤ 100 nm (Gezondheidsraad, 2021; Abdillah & Wang, 2023). In contrast to PM_{10} and $PM_{2.5}$, UFP is better quantified by particle number concentration (PNC) instead of mass. UFP can be carbonaceous, metallic, volatile/semi-volatile in nature, and is predominantly formed in combustion processes (Stacey, 2019). However, the majority of UFPs from aviation are not directly emitted as UFPs, but only formed in the air through nucleation and condensation of sulphur-containing gaseous compounds, released during kerosene combustion (Gezondheidsraad, 2021). The sulphur content in kerosene is usually between 550-750 ppm, although the maximum permitted sulphur content is 3000 ppm (Gezondheidsraad, 2021). Transitioning to ultra-low sulfur jet fuel (ULSJ) is likely to lower the amount of fine particulate matter emitted by airplanes, but it cannot completely remove the impact of aviation. While ULSJ may decrease the adverse health impacts of aviation-induced (ultra) fine particulate matter, it may increase the globally averaged climate warming caused by aviation, as the emission of SO_x exerts a cooling effect (Barrett et al., 2012; Hileman and Stratton, 2014; Kapadia et al., 2016).

The higher specific surface area (x 25 compared to $PM_{2.5}$) enables UFPs to adsorb more toxic materials (heavy metals, black carbon, PAHs; see Abdillah & Wang, 2023). Given the very small particle size, the UFP fraction can reach the most distal lung regions (alveoli) and circumvent primary airway defences (Kwon et al., 2020; Abdillah & Wang, 2023). UFP is especially important in aircraft emissions where the diameter of most particles is < 20 nm (Peters et al., 2016). The highest UFP concentrations are measured downwind of airport locations (Stacey, 2019; Bendtsen et al., 2021), associated with landing and take-off operations (Campagna et al., 2016).

<u>UFP at Brussels airport</u>

In 2015, VITO executed measurements of UFP during 2 months in the area around Brussels Airport to investigate the potential contribution of operations at Brussels Airport on the local air quality in surrounding residential areas (Peters et al., 2016). Five locations were selected in a range of 7 km, both upwind and downwind, around Brussels Airport. The dominant wind direction near the airport is SW. The UFP number concentration was continuously measured with a Scanning Mobility Particle Sizer (measurement resolution of 5 minutes during 2 months). While UFP *sensu stricto* consists of particles < 100 nm, this study included particle sizes between 10 and 294 nm for technical reasons. Special attention goes to the 10-20 nm fraction, which is enriched by air-traffic-emitted particles. The following observations were made (Peters et al., 2016) (Table 4):

- The average and peak concentration (P99) of the 10-20 nm class were largely increased near Diegem and Steenokkerzeel, the locations closest to the airport. The measured values were more than triple the rural background values of Kampenhout (Figure 13).
- The UFP concentrations vary between the hours of the day, coinciding with the morning rush and evening rush of LTO (Landing and Take-Off) operations at Brussels Airport. A correlation was revealed by a multiple linear regression model.
- Increased 10-20 nm UFP concentrations were measured when the monitoring locations were situated downwind of the airport. In contrast, no directionality towards the airport was measured for the larger UFP sizes (>70 nm).
- The peak concentration (P99) of the 10-20 nm class was two times higher in rural Kampenhout (downwind of the airport), compared to urban Evere (upwind of the airport).
- A conservative additive model was used to account for the contribution of the airport operations to the UFP concentration at nearby downwind operations. For 25 % of the time, an airport contribution to the 10-20 nm fraction of 20 000 to 28 000 pt/cm^3 was estimated, for 10 % of the time a contribution of 44 000 - 58 000 pt/cm^3 and for 5 % of the time 66 000 - 82 000 pt/cm^3.

It was concluded that the contribution of airport traffic to the 10-20 nm number concentration decreases with increasing distance, effects were still measurable at Kampenhout (7 km). Besides the distance to the airport, a relationship exists with LTO operations and wind direction.

Table 4: *Summarised UFP measurements performed by VITO in 2015 (measurement resolution of 5 minutes during 2 months) at four locations with increasing distance to Brussels Airport. The ratio "Average" and "P99" (peak concentration) compares values to the rural background measurements of Kampenhout. (Source: Peters et al., 2016)*

	Distance & direction from airport	Average 10-20 nm		P99 10-20 nm		Average 10-100 nm	Average 10-294 nm
		Pt/cm³	Ratio	Pt/cm³	Ratio	Pt/cm³	Pt/cm³
Diegem	250 m W	8119	3.1	68 992	3.5	16 461	18 096
Steenokkerzeel	750 m E	7776	3.0	74 370	3.7	15 912	17 182
Evere	5,000 m SW	2891	1.1	10 063	0.5	8723	10 224
Kampenhout	7,000 m NE	2615	1	19 660	1	6583	7685

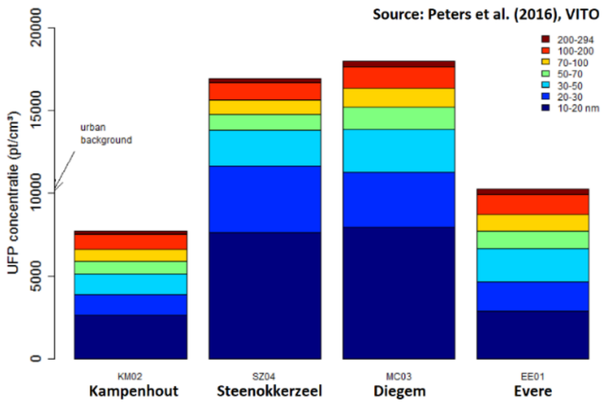

Figure 13: *Contribution of different particle size classes to the total UFP measurements per site, measurements performed by VITO in 2015. (Source: Peters et al., 2016)*

Levebvre et al. (2019) (VITO) modelled the UFP fractions emitted by the air- and road traffic in the surroundings of Brussels Airport, based on the measurements of Peters et al. (2016). After calibration, a good temporal agreement was found between the model and the measurements, if averaging was done over a sufficient time period (24h). It was found that the contribution of the road- and air traffic differs between different monitoring stations. While the total particle numbers were similar at Diegem and Steenokkerzeel, the contribution of aircraft emissions in Steenokkerzeel (62 %) was twice that of Diegem (34 %). In contrast, the contribution of road traffic was 40 and 10 % respectively. The Steenokkerzeel monitoring station was just east of the airport (downwind), with the wind coming predominantly from the SW. The influence of SW wind on aircraft UFP dispersion can also be seen clearly on the map of UFP annual averages modelled by VITO (Figure 14). At all locations, a fraction of the UFP exists that cannot be attributed to road or air traffic.

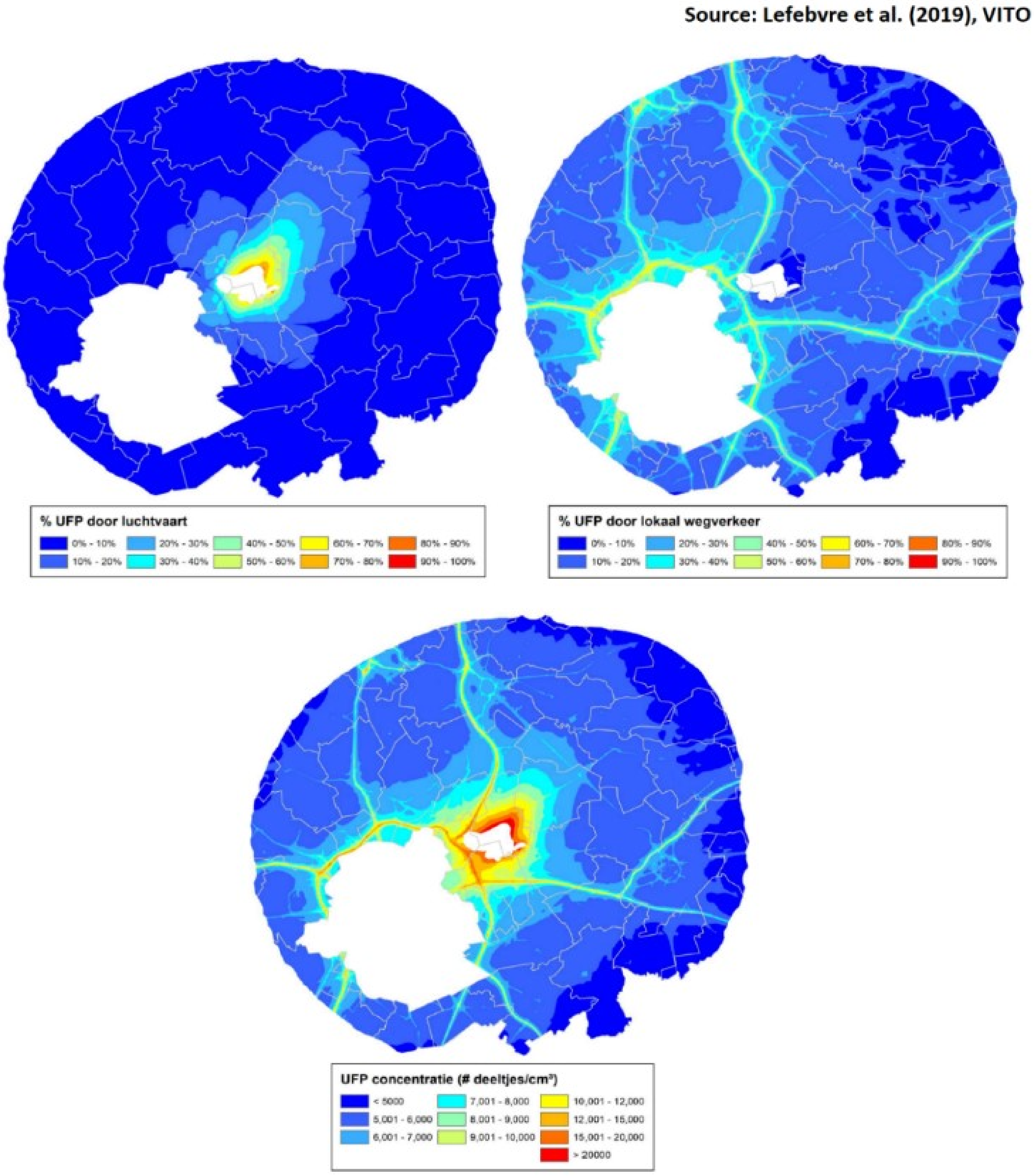

Figure 14: *Spatial distribution of annual UFP concentrations in the Flemish Periphery near Brussels Airport according to the VITO model (bottom), next to the share (%) of UFP due to air traffic (top left) and road traffic (top right) (Source: Lefebvre et al., 2019)*

A final VITO report was published on the monitoring campaign of UFP concentrations near Brussels Airport between 2018-2019 (Peters et al., 2019). This campaign was performed with an exceptionally high measurement frequency (1 measurement / 10 s), compared to the preceding one in 2015 (Peters et al., 2016). Measurements were performed during two summer months (2018) and two winter months (2018-2019). The findings of the earlier reports were confirmed. Peters et al. (2019) found that:

- The UFP concentrations near Brussels Airport are elevated due to aircraft emissions. Just outside the airport, the share of the air traffic as source for these emissions (particle numbers) is similar to the share of road traffic in the inner-city area.
- The highest peak concentrations occur on the airport site at the head of the runways. While the background UFP-concentrations (P10, P25) were similar at the different monitoring stations, strongly elevated P95 UFP-values were measured at the head of the 25R runway: between 6 am to midnight, P95 UFP values ranged between 300 000 and 1 000 000 pt/cm^3. During the same interval, The P25 UFP concentrations were much lower, ranging between 10 000 – 20 0000 pt/cm^3 (Figure 15). Also on other locations, large differences between the P25 and P95 values were found.
- Landing aircraft at low altitudes cause elevated UFP levels near the approach runway. Take-off causes elevated UFP levels at locations in the extension of the runway up to a height of 500 m. On the ground, the impact on UFP concentration decreases sharply with increasing distance from the point of departure and aircraft height.
- The time it takes UFP emissions to reach downwind locations depends on the speed of the wind and meteorological conditions. About 1 to 3 minutes after departure, the UFP emissions have moved 1 km further downwind and 3 km after 5 to 15 minutes.
- Seasonal differences were relatively small. While there were circa 28 % more flights during the summer, UFP concentrations were not systematically higher compared to the winter months.

***Table 5**: The contributions of air and road traffic to the number of UFPs at four locations. (Source: Lefebvre et al., 2019)*

	Distance & direction from airport	UFP Air-traffic	UFP Road-traffic
Diegem	250 m W	34%	40%
Steenokkerzeel	750 m E	62%	10%
Evere	5000 m SW	8%	45%
Kampenhout	7000 m NE	18%	12%

Source: Peters et al. (2019), VITO

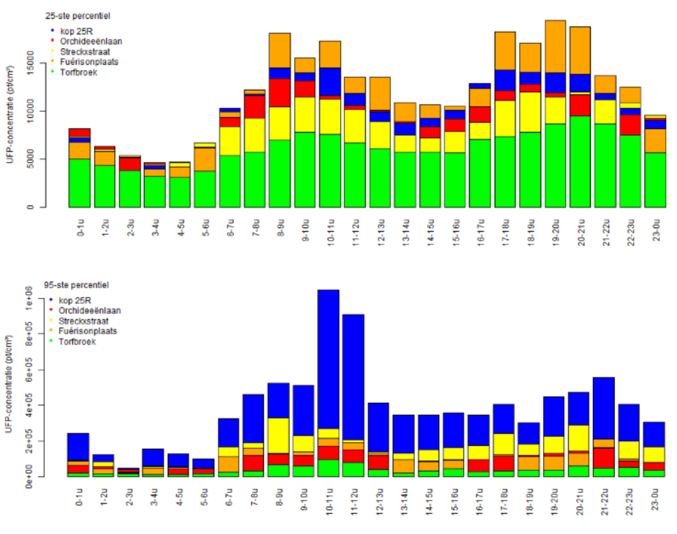

***Figure 15:** UFP fractions (P25 and P95) in function of the hours and different locations. The P95 values measured at the head of the 25R runway are strongly elevated compared to other monitoring locations. (Source: Peters et al., 2019)*

<u>UFP at other airports</u>

Campagna et al. (2016) measured UFP levels near a military airport in Italy and the surrounding rural and urban areas. The median UFP count ranged between 3.7×10^3 and 2.9×10^4 pt/cm^3, the highest UFP count (4.0×10^6 pt/cm^3) was measured during taxiing and take-off. The UFP number concentrations were more elevated in the winter season, in contrast to the findings of Peters et al. (2019) near Brussels Airport. No correlation was found between flight activities and the UFP counts in residential areas. However, the UFP count showed a tendency to decrease with distance from the emission source.

In a survey near Montréal-Trudeau International airport, the geometric mean and P99 values of the observed nanoparticle number densities (<100 nm) at the apron were 1.0×10^5 and 1.2×10^6 respectively (Rahim et al., 2019). These values were statistically higher than corresponding measurements in downtown Montreal and at major highways during rush hour.

In the Netherlands, a recent study of the RIVM calculated and measured UFP particles (between 7-100 nm) in the surroundings of Schiphol airport (Voogt et al., 2019). According to the calculations of Voogt et al. (2019), the annual average exposure of local residents to UFP from air traffic at Schiphol (2017-2018) is the highest near the airport and decreases with increasing distance. Small annual differences were noted, caused by variations in weather conditions and runway use. These are similar trends to those concluded by the VITO studies. A previous study on Schiphol (Keuken et al., 2015) found that UFPs are a factor 3 elevated 7 km downwind Schiphol airport, while the size-distribution of these particles is dominated by the 10-20 nm class.

Hudda et al. (2020) monitored air quality at a residence under a flight trajectory of the most utilised runway configuration of Logan International Airport in Boston. It was found that the highest UFP particle number counts coincided with the periods of highest noise co-exposures (i.e. overhead landing flight hours). The need was underlined to account both for aviation air pollution and noise co-exposures to avoid potential confounding of health risk associations.

Zhang et al. (2020) studied the influence of aviation emission on UFP particle number concentrations near Zurich Airport. It was estimated that a total of 1.0×10^{24} particles were emitted per year. Aviation emission increased the annual mean particle mass concentrations in the nearby communities by ca. 0.1 µg/m³, corresponding to 1% of the background concentrations. However, the particle number concentration could be increased by a factor of 2-10 of the background level (10^4 cm^{-3}) for nearby communities.

Chung et al. (2023) studied the impact of arrival aircraft on particle number concentration as a proxy for UFPs, across six study sites 3 km from a major arrival flight path into Boston Logan International Airport. These authors found strong but intermittent contributions from arrival aircraft to ambient particle number concentrations near the airport. While the particle number concentration median was similar at all sites, larger variation existed at the 95th and 99th percentiles with more than two-fold particle number concentration increases near the airport. Especially during the hours with high aviation activity, the closest monitoring sites showed higher values when downwind of the airport. A regression model indicated that the number of arrivals per hour was associated with the particle number concentration measurements at all sites, with a maximum contribution of 50 % at the 3 km site during the hours with arrival activity on the flight path of interest.

2.1.2 *Black carbon*

Black carbon is a component of fine particulate matter, consisting of pure, tiny carbon particles produced by incomplete combustion. Therefore, near airports, it can origin from aircraft emissions, but also from major roadways in the surrounding of the airport (Dodson et al., 2009; Bendtsen et al., 2021).

Black carbon at Brussels airport

Besides UFP, Peters et al. (2016) also measured black carbon concentrations (µg/m³) in the vicinity of Brussels airport. It was concluded that the contribution of airport activities to black carbon was much more limited compared to the contribution of motorway and road traffic.

Black carbon at other airports

Dodson et al. (2009) found a significant positive association between hourly departures and arrivals at the small airport of Warwick (UK), and black carbon concentrations within the community. In contrast, Keuken et al. (2015) measured the presence of black carbon near the Schiphol Airport (the Netherlands). No significant elevation was found.

2.1.3 *Volatile Organic Compounds (VOCs)*

Volatile Organic Compounds (VOCs) are gases with a high vapor pressure at room temperature. It is a diverse group with different physicochemical properties and health effects. Due to their large diversity, no “general” standards can be set for total VOCs exposure. According to the Federal Aviation Administration (2003) in the US, the main aircraft-related VOCs are formaldehyde, acetaldehyde, benzene, toluene, acrolein, 1-3-butadiene, xylene, and propionaldehyde. Formaldehyde, benzene, 1-3-butadiene are classified as carcinogenic to humans by IARC (Group 1). VOCs related to aircraft emissions generally receive less attention in the scientific literature than PAHs, hence their impact on health remains to be further investigated (Masiol & Harrison, 2014).

VOCs at Brussels airport

No specific measuring campaign has been performed to quantify VOCs near Brussels airport.

VOCs Other airports

A TNO study by Thijsse & van Loon (2001) studied the quality of air near Schiphol airport (the Netherlands). VOCs concentrations around Schiphol were on average 66 % determined by large-scale transport from more distant source areas and on average 28 % by road traffic in the surrounding area. It could also be deduced that the annual average contribution of Schiphol (air traffic, storage and transshipment) to the ambient benzene concentration in Badhoevedorp amounted to a maximum of ca 0.13 µg/m^3 (in Badhoevedorp), against a background concentration of about 1.2 µg/m^3.

Similar conclusions were drawn earlier by van den Anker et al. (1991): measurements and model calculations indicated that Schiphol Airport’s relative contribution to regional concentration levels was less than 10% for SO_2, NO_x, CO, VOCs and PAHs. These authors found no increased mutagenicity performing the Ames test on some aerosol samples taken near Schiphol airport.

Visser et al. (2005) reported an average concentration of 1.4, 1.1 and 0.7 µg /m³ of benzene in ambient air measured at different monitoring stations during 2002 near Schiphol airport.

Recently, TNO (2023) calculated the emissions of several VOCs in a study on Substances of Very High Concern (SVHC) near the airports of Amsterdam (Schiphol), Eindhoven, Groningen, Maastricht and Rotterdam. It was found that the hourly average of several compounds exceeds the mass flow limit as used in the industry, indicating that the emissions are non-negligible. It should be noted that airports are no point sources, hence the dispersal of the emissions must be considered when estimating potential health effects.

A study on the exposure of airport employees in the Arrival Hall of Beirut-Rafic Hariri International Airport to VOCs was performed by Mokalled et al. (2021). VOCs levels were measured using gas chromatographic techniques (GC-MS and GC-FID). Except for acrolein, the measured VOCs did not present any additional risks for human health. A correlation was revealed between the aircraft number and the concentrations of certain VOCs groups (heavy alkanes, aldehydes, ketones, and monoaromatics).

2.1.4 Polycyclic Aromatic Hydrocarbons

Polycyclic Aromatic Hydrocarbons (PAHs) arise from incomplete carbon combustion and pyrolysis of organic material. Therefore, PAHs are widespread and found in the air, water, soils and sediment, mostly at trace levels near their primary sources (IARC, 2010). PAHs are frequently adsorbed on the surface of particulate matter, reaching the deeper parts of the lungs together on the UFP fraction. Multiple PAHs are classified as carcinogenic, or probably and possibly carcinogenic to humans (Groups 1, 2A, 2B) (IARC, 2010). Besides, also mutagenic, genotoxic and endocrine disrupting effects are known (Zhang et al, 2016). Benzo[a]pyrene (B[a]P) is an important carcinogenic representative of the PAH-group, often used as a reference to calculate toxic equivalent factors of different PAHs. Jet engines predominantly produce low molecular weight PAHs with 2-3 aromatic rings (Bendtsen et al., 2021). These Lighter-weight PAHs (e.g. naphthalene and its 1- and 2-methyl derivatives) are present almost exclusively in the vapour phase, whereas PAHs with higher molecular weights (>4 aromatic rings) are almost totally adsorbed on particles of particulate matter (Masiol & Harrison, 2014).

<u>PAHs at Brussels Airport</u>

No systematic studies have been done to quantify PAHs emissions near Brussels Airport. However, some regional data are indirectly available. The Flemish Environment Agency (VMM) organises periodic measurement campaigns throughout Flanders to monitor the presence of PAHs in ambient air, by sampling PM_{10}. The 2012 VMM annual report did set up a monitoring station in a wooded park in a residential area in Steenokkerzeel (60SZ02), located 1.5 km east (downwind) of the runways of Brussels Airport (VMM, 2013). Other monitoring stations were Ghent and Borgerhout (metropolitan context), Zelzate (residential area surrounded by industry) and Houtem (rural, studied as background). Compared to Houtem, there was an increase in the total sum of PAHs of 70 % in Ghent, 120 % in Steenokkerzeel, 150 % Borgerhout and 220 % in Zelzate. The 2012 annual averages of 10 PAHs measured at Steenokkerzeel were in line with the other measurement sites (6). However, an increased indeno(1,2,3-cd)pyrene/benzo[a]pyreen ratio was generally observed at measuring station Steenokkerzeel. VMM concluded that this suggests the existence of a specific, nearby source that emits relatively more heavy PAHs. It is unknown whether this was related to the airport activities.

<u>PAHs at other airports</u>

Near Schiphol airport, Thijsse & Van Loon (2001) found that the contributions of air traffic and kerosene storage and handling to hydrocarbon concentrations in ambient air were on average about 1 and 2%, respectively.

Visser et al. (2005) reported an average concentration of 0.14 ng/m³ of benzo(a)pyrene (B[a]P) in ambient air measured at a monitoring station during 2002 in Badhoevedorp near Schiphol airport.

Cavallo et al. (2006) measured the concentrations of 23 PAHs at Fiumicino Airport (Rome). They found 27 703 µg/m^3 total PAHs in the airport apron, 17 275 µg/m^3 in the airport building and 9494 µg/m^3 in the terminal/office area.

Iavicoli et al. (2006) took 12 air samples at 120 l/min for 24h during the winter of 2005 in an Italian airport. In general, PAH levels remained low, the higher levels refer to naphthalene (130-13 050 ng/m), 2-methylnaphthalene (64-28 500 ng/m), 1-methylnaphtalene (24-35 300 ng/m³), and biphenyl (24-1610 ng/m³). Only two compounds showed levels of some concern, benzo[b+j+k]fluoranthene and benzo[a]pyrene (52.2 ng/m³ and 8.6 ng/m³ respectively).

PAH concentration (22 compounds) was also measured by Lai et al. (2012) at the airport apron of the Taipei International Airport in Taiwan. The most abundant species of particle-bound PAHs on all sampling days were naphthalene, phenanthrene, fluoranthene, acenaphthene, and pyrene. Total particle-bound PAHs concentrations were 152.21, 184.83, and 188.94 ng/m^3 in summer, autumn, and winter, respectively.

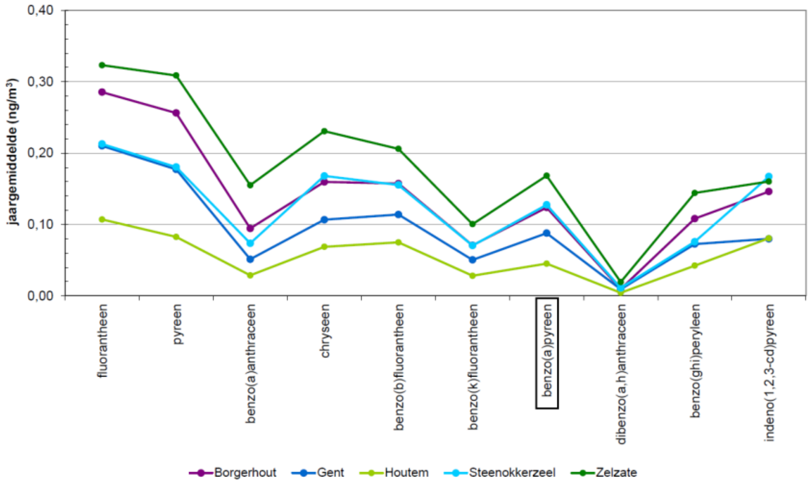

Figure 16: *Annual average of different PAHs measured across five Flemish monitoring stations in 2012. The Steenokkerzeel monitoring station was located 1.5 km east of the airport runway of Brussels Airport in Zaventem. (Source: VMM, 2014)*

2.1.5 Lubrication oil and organophosphate esters (OPEs)

Organophosphate esters (OPEs) are a class of organophosphorus compounds with a central phosphate molecule with alkyl or aromatic substituents. OPEs are used as stabilising agents in aircraft lubricating oil and hydraulic fluids. Also in many other applications, OPEs have been used as flame-retardants and plasticisers due to their low costs and excellent physicochemical properties. The dominant OPE in lubrication oil is tricresyl phosphate, a highly neurotoxic compound, which has been detected in ambient air and aircraft cabins (Masiol & Harrison, 2014; Bendtsen et al., 2021). Oil escaping and burning from lubricated parts may be vented overboard from aircraft engines and may contribute to the total aircraft emissions (Masiol & Harrison, 2014).

<u>OPEs at Brussels airport</u>

No measuring campaign has been performed to quantify OPEs Brussels airport.

OPEs at other airports

Li et al. (2019) studied the occurrence, distribution and human exposure to 20 OPEs in air, soil, dust, river water and pine needles collected near the International airport of Albany (New York, USA). Elevated ΣOPE concentrations were found in outdoor air (1320–20 700 pg/m^3, median: 3880 pg/m^3), soil (1.16–73.1 ng/g dry weight, median 14.3 ng/g), pine needles (23.2–534 ng/g dry weight, median 102 ng/g), outdoor dust (153–2140 ng/g dry weight, median 824 ng/g) and river water (174–24 600 ng/L, median 1250 ng/L). The average daily intake of OPEs via air inhalation and outdoor dust ingestion near the airport was up to 1.53 ng/kg body weight/day for children and 0.77 ng/kg body weight/day for adults. The authors showed that the spatial distribution of OPEs in air, soil and pine needles showed a gradual decreasing trend with increasing distance from the airport. Especially the spatial distribution of OPEs between air and pine needles was highly correlated. An important note was made: Albany airport is a small regional airport, higher concentrations might be expected in the environment of larger airports.

2.1.6 Metals

Particulate matter in aircraft emission is usually investigated in terms of particle size and concentration. However, the chemical composition should also be considered. In general, it is very difficult to investigate the exact relationship of metals in particulate matter to aviation, as there are many variables to cover, such as the exact fuel used, lubrication oil, engine type, engine wear, etc. Furthermore, due to the vicinity of other polluters, such as car traffic, it is not always possible to correctly identify the exact impact of jet engine exhaust. According to Boyle (1996), only the metal Vanadium is specific to jet engine exhaust and could be used for fingerprinting aviation input in the environment. Using single particle mass spectrometry, Abegglen et al. (2016) measured PM composition solely from jet engines. The most abundant metals are: Cr, Fe, Mo, Na, Ca and Al. As the measurements were performed on a stationary jet engine, the authors reasoned the potential sources could be the fuel, lubricants or motor wear. These potential sources were further explored by Turgut et al. (2020) in piston-engine aircraft. Elements In, Sm, Ge, Ga, Ag, La and Zr were not significantly detected in the fuel or oil used but were detected in the exhaust. They must thus originate from engine wear. Furthermore, the amount of Cr in the jet exhaust was found to be variable between different engine types (Agrawal et al., 2008).

Metals at Brussels airport

No measuring campaign has been performed to quantify metals near Brussels airport.

Metals at other airports

One possible outcome of the PM emission is to end up in the soil. A study by Massas et al. (2018) investigated soil samples in the direct vicinity of Athens International Airport in Greece. Although the authors conclude continuous enrichment of the studied soils with metals Pb, Cu, Zn, Mn and Ni, their respective median values are not significantly different from other soil samples nearby. A similar study near Warsaw Chopin Airport concluded the same, no significant metal (Cu, Ni, Pb) contamination of the soil could be measured due to airport activity. (Brtnický et al., 2020)

One way to get more insight on the impact of aircraft emission on air quality could be to use moss bags. Turgut et al. (2019) measured the metal content in moss bags as a biomonitor for air pollution in the vicinity of Eskisehir Hasan Polatkan Airport, Turkey. The authors conclude an agreement in metal composition between the oil/fuel analysis and moss biomonitoring, with Pb as main pollutant followed by Cd, Cu, Mo, Cr, Ni, Fe, Si, Zn, Na, P, Ca, Mg, and Al.

Rahim et al. (2019) reported on the presence of a wide range of metals in aerosols, predominantly composed of nanoparticles (<180 nm), measured in situ at the Montréal-Pierre-Elliott-Trudeau International Airport (Canada). The most abundant metals detected were Al, Fe and Zn.

2.1.7 Nitrogen oxides

Nitrogen emission, deposition and eutrophication are an important environmental issue, mainly due to the intensive livestock farming, the use of fertilisers and industry (Van Landuyt et al., 2008; Staelens et al., 2012; De Pue et al., 2017). Aviation, just like other industrial and transport processes, also contributes to nitrogen deposition. According to Quadros et al. (2023), aviation globally led to 1.39 Tg of N deposition. However, this contribution is very small compared to other sources: in Europe, only a total of 1.1 % of nitrogen deposition is estimated due to aviation. Further growth of flight numbers might increase this contribution in the following decades. Globally, an average of 8 % of aviation's nitrogen deposition is emitted during the landing, taxi and take-off (LTO) phases. In regions with high aviation activity, this is between 16 and 32 % (Quadros et al., 2023).

Nitrogen oxides at Brussels airport

Nitrogen oxides were included in the abovementioned VITO measuring campaign in the surroundings of Brussels Airport (Peters et al., 2016). Measurements were performed in Steenokkerzeel, Diegem, Evere and Kampenhout at a resolution of 30 minutes during October and November 2015 (Table 6). The highest NO_x concentrations are observed in Diegem, followed by Evere. In Steenokkerzeel, NO_x concentrations are lower than in Diegem and Evere. The analysis of NO_x concentration profiles in function of wind direction did not reveal higher contributions from the airport than from other (mainly traffic) sources (Peters et al., 2016).

Table 6: *Summarised NO_x measurements (as NO_2 equivalent; µg/m³) performed by VITO in 2015. Measurements were performed with a 30 minutes resolution. Source: Peters et al. (2016)*

	Kalmthout	Steenokkerzeel	Diegem	Evere
Quartile 1	11.8	19.1	32.7	26.6
Median	21.5	32.2	55.7	45.3
Mean	29.6	48.3	83.1	65.1
Quartile 3	35.3	55.8	97.5	76.1
Maximum	335.1	485.6	1417.7	828.6

The VITO measurements exceed the 2021 Air Quality Guidelines (AQGs) for NO_2 of the WHO (99^{th} percentiles: 1h – 200 µg/m³; 1 day – 25 µg/m³; a year – 10 µg/m³).

In 2018, a large-scale citizen-science project called "CurieuzeNeuzen" measured the concentrations of NO_2 in Flanders. NO_2 is an important indicator for air pollution due to traffic, as this compound is formed during high-temperature combustion in engines. A digital map is provided with all the values measured across the region (https://viewer.curieuzeneuzen.be/; accessed 3/7/2023). No clear relationship with the airport was visible in the surroundings: Machelen (24.6-48.6 µg/m³), Vilvoorde (17.7-46.4 µg/m³), Steenokkerzeel (16.8-33.7 µg/m³), Zaventem (15.5-50.6 µg/m³), Kampenhout (14.6-31.6 µg/m³). Pointwise, it can be seen that the measurements closest to the runway (Melsbroek, Steenokkerzeel, Kortenberg, Nossegem; N, NE, SE, S of the runways) are the lowest. In contrast, the measurements between the runways, but very close to the ring road (Machelen, Diegem, Zaventem; NW, W and S of the runway) are much more elevated. Thus, it seems that NO_2 concentrations are mainly linked to road traffic.

Currently, the Flemish Government uses the atmospheric dispersion model VLOPS (*Vlaamse Operationeel Prioritaire Stoffen*) as an indicator for the total deposition of nitrogen oxides and ammoniacal nitrogen, taking into account cross-border emissions (import and export). The model has a resolution of 1x1 km² and uses the following inputs: meteorological data, emission data from point and surface sources, and data on receptor areas. According to the most recent model (VLOPS22; https://www.vmm.be/lucht/stikstof/stikstofdepositie; accessed 3/7/2023), the surroundings of Brussels Airport show elevated nitrogen deposition, with values generally between 15-20 kg N/ha/year. In the model, the values near the Ring Road in Zaventem and Machelen are slightly elevated (20-25 kg N/ha/year).

In a report on the emissions of different sectors between 2000-2018, the Flemish Environmental Agency studied estimated the emissions of aviation using the EMMOL model (VMM, 2020). Landing and Take Off-flights (LTO, up to 915 m) and cruise flights (> 915 m) were considered separately. Between 2010 and 2018, annual LTO emissions of the Flemish airports were between 1000 and 1400 tonnes of NO_x. More than 95 % of these emissions are related to Brussels Airport (Figure 17). However, in 2018, the share of LTO emissions in Brussels airport was only ca. 10 %, while ca. 90 % was due to cruise flights (Figure 18). This share for LTO emissions is within the range reported by Quadros et al. (2023). Based on the numbers in Figure 17, a total of about 13 500 tonnes of NO_x would therefore have been emitted by Brussels Airport in 2018, of which only 1350 tonnes (LTO emissions) would have an impact on the nearby environment by deposition. The other 90 % spreads much further and causes deposition in a wider region, making it difficult to link it to a specific source.

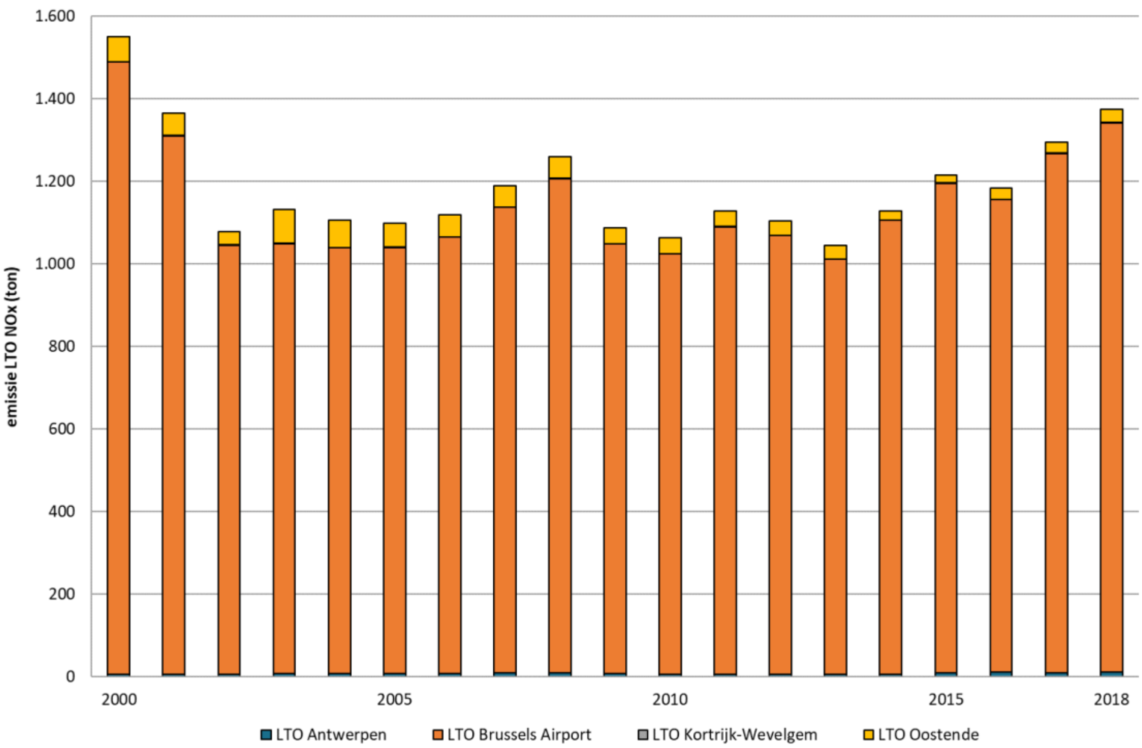

Figure 17: *Share of different Flemish airports in the total NO_x (NO_2) emissions during LTO (Source: VMM, 2020, fig. 4.18)*

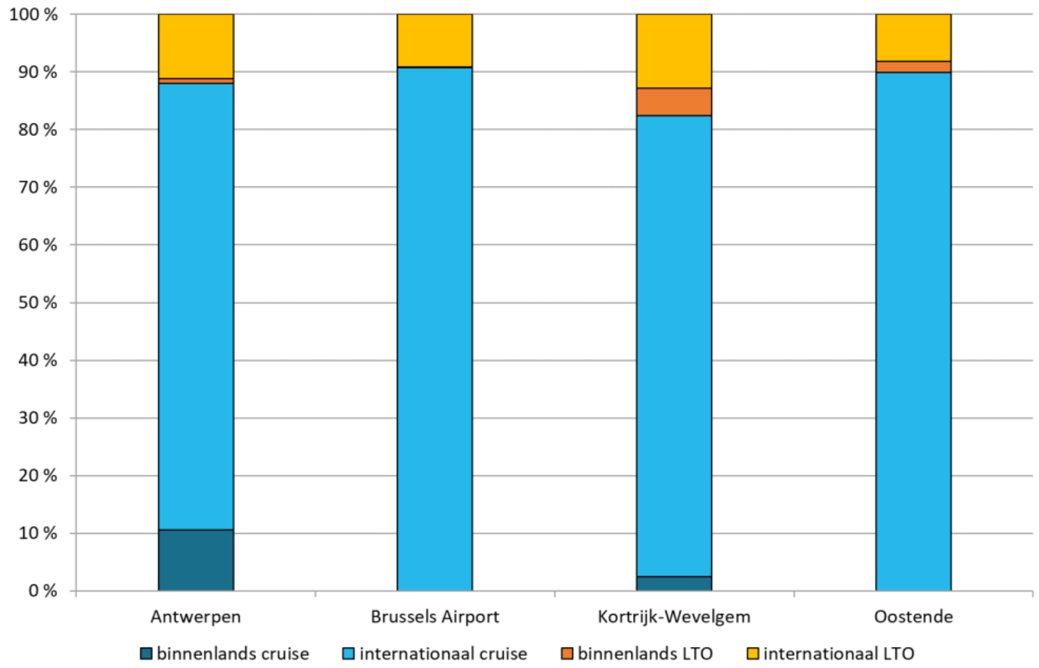

Figure 18: *Share of cruise/LTO and national/international activities causing NO_x emissions in 2018 (Source: VMM, 2020, fig. 4.16)*

2.1.8 Nitrogen oxides at other airports

Studies on nitrogen emission near airports are relatively scarce.

Visser et al. (2005) reported average concentrations of 28, 28 and 21 µg/m³ in ambient air near Schiphol area in monitoring stations Badhoevedorp, Oude Meer and Hoofddorp in 2002.

Mokalled et al. (2021) monitored the NO_2 concentrations in the Arrival Hall of Beirut-Rafic Hariri International Airport, using calorimetric methods. Results show that NO_2 concentrations exceeded 40 µg/m^3 in the arrivals hall at two out of four measurement campaigns to reach values of 41.6 µg/m^3 (26–27 October 2014) and 48.4 µg m^{-3} (24–29 June 2015), followed by 36.1 µg/m^3 (25–26 November 2014).

2.2 Potential health impact of aircraft pollutant emissions

Air pollution is one of the largest known and best studied environmental risks to human health, significantly increasing the incidence of diseases, especially cardiovascular diseases and several types of cancer, premature deaths, and disability-adjusted life years (DALYs). The previous section briefly summarised different types and fractions of air pollution related to airports. Due to the complex interactions, it is not possible to disentangle the contribution of individual compounds to the adverse health effects. Residents are exposed to all these components simultaneously. It is exposure to the complex mixture that causes health effects (see also advisory report no. 9404 of the Superior Health Council). However, documenting these effects in epidemiological studies in the surroundings of airports is very difficult, as many other sources of pollution are present and human epidemiology lacks the necessary sensitivity and distinctive capability. Molecular epidemiological studies and studies measuring physiological functions could give a more mechanistically based view of the health effects.

While health effects are also known from other emissions (e.g. slightly positive associations are observed between long-term exposure to NO_2 and all cause, cardiovascular, respiratory and lung cancer mortality; Atkinson et al., 2018), studies near airports, have focused predominantly on exposure to Ultra Fine Particles (UFPs). As the particle diameter of aircraft UFP is predominantly below 20 nm, the high specific surface area of the UFP results in the adsorption of more toxic compounds (including carcinogenic PAHs) compared to larger particles. These harmful compounds are then delivered to the most distal lung regions, after which a part of these UFPs can be transported via systemic circulation to different target tissues and organs in the body including the brain and the foetus. In the Netherlands, several relevant studies have recently been performed near Schiphol Airport, these can give a rough idea of the effects that can be expected.

Before going into the health effects of air pollution near airports, it is important to briefly outline the current knowledge of the potential health impact of UFPs, independent of the source of emission.

2.2.1 General health effects of UFP

An excellent review of peer-reviewed literature with integrated systematic assessment (weight of evidence) was carried out by the Dutch Health Council according to the EPA guidelines for Integrated Science Assessment (Gezondheidsraad, 2021). Concerning **short-term UFP exposure** (minutes to 1 month), evidence for metabolic effects (diabetes mellitus and metabolic syndrome) and an increase in total mortality was judged **insufficient**, due to the lack of both toxicological and epidemiological evidence at the time the review was conducted. In contrast, **indicative evidence** was found for the following effects:

- ⇨ **Short-term cardiovascular effects**: Within some hours of the UFP exposure, associations are consistently found with subclinical markers of oxidative stress, increased heart rate and heart rhythm abnormalities. To a lesser extent, the same holds for hypertension and blood clotting.
- ⇨ **Short-term respiratory effects**: Consistent, corrected associations (adjusted for confounders) are found with inflammatory markers in the respiratory tract. Associations with hospitalisation and mortality are less consistent.
- ⇨ **Short-term neurologic effects**: Although no human epidemiological studies are available, animal experiments showed inflammation in the brain after short-term UFP exposure.

Also the health effects of **long-term UFP exposure** (1 month to 10 year) were reviewed by the Dutch Health Council (Gezondheidsraad, 2021). Evidence for metabolic effects, cancer and total mortality were judged **insufficient** at the time the review was conducted. **Indicative evidence** was found for the following effects:

- ⇨ **Long-term cardiovascular effects**: Several large cohort studies showed positive associations with hypertension, heart failure, myocardial infarct and dying from ischaemic heart disease. Although not every outcome was statistically significant in each study, associations for several types of effects remained consistent after adjusting for other components of air pollution.
- ⇨ **Long-term respiratory effects**: Two large cohort studies showed associations with the incidence of asthma among children and COPD among adults. In contrast, no or only weak associations between chronic UFP exposure and respiratory diseases were found in some other cohort and cross-sectional studies. Some studies did show an association between long-term UFP exposure and markers of systemic inflammatory processes and airway inflammation.
- ⇨ **Long-term effects on foetal growth and development:** UFP particles can end up in the foetus's amniotic fluid and blood circulation. The smaller the particles, the easier they can cross the placental barrier. Studies on maternal exposure to UFP during pregnancy and the risk of an underweight baby showed different outcomes (association or no association). Studies concerning the risk of preterm birth did find a concentration-response relationship with UFP from air traffic. Canadian studies found no association with the incidence of 7 types of heart defects and atrial septal defect, though one significant association was found with ventricular septal defect.
- ⇨ **Long-term neurological effects**: Experimental evidence shows that UFPs can pass the blood-brain barrier, UFP can reach the cerebrospinal fluid via the blood circulation and it can be transported directly to the brain along the olfactory nerve. Indications exist for inflammatory reactions due to UFP in the brain. Only one epidemiological study was included (not corrected for other air pollutants), showing a slightly poorer cognitive growth among children exposed to high (road traffic) UFP levels, compared to less exposed children.

Since the Dutch Health Council's review in 2021, new epidemiological studies have been published that may further refine the conclusions. At the time, it was concluded that insufficient evidence existed for an association between long-term UFP exposure and total mortality. However, a recent large cohort study by Bouma et al. (2023) in the Netherlands found significant associations between UFP exposure and natural (not cause-specific) mortality [HR 1.012 (95 % CI 1.010-1.015) per interquartile range (2723 particles/cm^3) increment], respiratory disease mortality [HR 1.022 (95 % CI 1.013–1.032)] and lung cancer mortality [HR 1.038 (1.028–1.048)]. The weakest association has been found for cardiovascular disease mortality [HR 1.005 (1.000–1.011)]. This study included 10.8 million adults (>30 years) followed between 2013-2019, the exposure model was run for people who had lived at their baseline address for at least five years. The strength of evidence of this study is high since potential mutual confounding by NO_2, PM_{10}, $PM_{2.5}$ and elemental carbon was taken into account, and the study was adjusted for the degree of urbanisation and socio-economic variables. Indirect adjustment of smoking and BMI was done using Public Health Monitor Data, indicating that smoking and BMI are not major confounding factors in the relationship between long-term UFP exposure and mortality. **In conclusion, the cohort study of Bouma et al. (2023) showed that long-term UFP exposure was independently associated with overall and lung cancer mortality among adults.**

2.2.2 *Health effects of air pollution near airports*

Internationally, chemical and physico-chemical emissions from aviation are a source of concern for health issues among local residents. Studies of various kinds (large epidemiological studies, human biomonitoring programmes, genotoxicity tests on air samples) are conducted to distinguish facts from fears. Especially in the Netherlands, knowledge is advanced, where the National Institute of Public Health and the Environment (RIVM) has been monitoring health effects near Schiphol airport for more than 30 years.

In 2022, the RIVM published a large report on both acute and chronic health effects of UFPs emitted by aviation near Schiphol airport (Janssen et al., 2022a). This report resulted from an integrated research program, investigating both the health effects of short-term and long-term exposure to UFPs. The multifaceted approach makes this research a pioneering study around this globally little understood issue. Clear criteria were used for the classification of a causal relationship (*sensu* Gezondheidsraad, 2021). Associations were mostly insensitive to adjustments for other air pollutants and aircraft noise, providing evidence for the independence of UFP health effects.

First, a model was created of long-term air concentrations of UFP from aviation based on measurements of PNC at 10 locations in the area around Schiphol. Spearman and Pearson correlation coefficients between the concentrations predicted by the model and the measurements were high (> 0.83). The dispersion model allowed to estimate the spatially varying average concentrations due to aircraft emissions in residential areas near Schiphol over 6 months. This made it possible to estimate long-term exposure of residents in the vicinity of the airport for epidemiological studies (Voogt et al., 2023).

For the effects of short-term exposure, three studies were undertaken (Janssen et al., 2019):

- A panel study among 191 primary school children living near Schiphol. During continuous measurements of UFP and soot in the schoolyards, weekly lung function and exhaled NO measurements were performed at school. Daily lung function and respiratory symptoms recording were performed at home.
- A volunteer study with 21 healthy, non-smoking adults (18-35 years) directly next to Schiphol. The volunteers were exposed for 5 h in a mobile lab next to Schiphol airport on 2 to 5 different days. Concentrations of UFP and other air pollutants were measured to estimate the exposure. The PNC was on average 53 500 particles/cm^3 (range 10 500–173 200). Before and after the exposure, lung function, exhaled nitric oxide, ECG, blood pressure and oxygen saturation were measured (Lammers et al., 2020)
- A toxicological study (*in vitro*). Human bronchial epithelial cells (Calu-3) were exposed to UFP samples with an air-liquid interface cloud system. UFP samples were taken at the location of the volunteer study and directly from the exhaust of a turbine engine. Endpoints were cell viability, cytotoxicity and inflammation (He et al., 2020).

The Particle Number Count ≤20nm was used as an indicator for UFP mainly from aviation, while the PNC > 50 nm was used as an indicator for UFP mainly from road traffic. No significant difference was found between the effects of aviation UFP compared to road traffic UFP.

For the effects of long-term exposure (Janssen et al., 2022b), modelled UFP concentrations (Voogt et al., 2023) at the residential addresses were linked to data from health registries and surveys. The approach consisted of five steps: (1) determining the study area, (2) selecting the health effects, (3) statistical analyses, (4) classifying results per health outcome, (5) classification per effect type. Four studies were undertaken linking health data to exposure estimates over 2003-2019:

- A cohort study for mortality follow-up of 1.2 million people between 2008-2019.
- A cohort study on medication use as a proxy for disorders. Follow up between 2008-2019.
- A cross-sectional health monitoring survey (Public Health Monitor). Questionnaires among 90 880 adults in 2012 and 2016.
- A perinatal registration study for birth-outcomes.

All effect estimates were adjusted for individual and neighbourhood level covariates. The health survey was also adjusted for smoking, alcohol use, education, BMI, physical activity. Associations were classified as “clear, probable, possible, no or inverse”, mainly focusing on consistency, and not only on statistical significance.

Besides the RIVM study of Janssen et al. (2022a), some other relevant studies are also included in the descriptive overview of each type of effect.

2.2.2.1. Cardiovascular effects

RIVM study – Short-term exposure to aircraft UFP (Janssen et al., 2019):

- Volunteer study: A 5–95th percentile increase in exposure to UFP (i.e. 125 400 particles/cm^3) was associated with a prolongation of the corrected QT interval (QTc on ECG) by 9.9 ms (95 % CI 2.0 – 19.1). This effect was associated with particles < 20 nm. For other cardiac function parameters, no statistically significant associations were observed with UFP (Lammers et al., 2020). While no association was found with aircraft UFP, UFP from road traffic (PNC>50 nm) was significantly associated with an increase in blood pressure (Bpdia 3.7 mm, 95 % CI 0.1 – 7.5) (Lammers et al., 2020)

RIVM study – Long-term exposure to aircraft UFP (Janssen et al., 2022b):

- The mortality cohort study suggested no association with cardiovascular mortality as primary endpoint, and no association with ischaemic heart disease, stroke and cerebrovascular disease as secondary endpoints. A probable association was suggested with arrythmia.
- The medication use cohort study suggested no association with hypertension, but a probable association with heart disease.
- The cross-sectional public health monitor (surveys) suggested a possible association with stroke and heart disease (medication). A clear association was suggested with heart attack, self-reported hypertension and hypertension medication.

Altogether, (Janssen et al, 2022a) concluded that indications exist for effects of short-term exposure on the cardiovascular system of adults, such as the prolongation of the QTc interval on ECG. **Indicative evidence** exists for effects of long-term exposure to UFP from aviation around Schiphol on cardiovascular health. The evidence was assessed as “indicative” because there is still too much uncertainty to conclude a causal relationship, although there are indications of causality.

A cross-sectional study by Lecca et al. (2021) among 34 male operators, working nearby taxiways, explored the association between exposure to Fine and Ultra Fine Particles and noise with heart rate variability (HRV), an early indicator of cardiovascular autonomic response. Total Lung Deposited Surface Area (LDSA), a parameter of cumulative (U)FP exposure, was significantly associated with a decrease in HRV Total Power and Triangular Index. Noise peak level showed the opposite effect. The results suggested that the effect of UFP exposure on short-term changes in HRV parameters among ground staff seems to be at least partially explained by concomitant exposure to impulsive noise.

A recent systematic review and meta-analysis of 12 studies on short-term effects of UFP on autonomic function, assessed by heart rate variability (HRV), confirmed that short-term exposure to ambient UFP is associated with decreased HRV, predominantly as an immediate response within hours (Zhang et al., 2022). The study took into account the risk of bias in the included studies regarding confounding, selection bias, exposure assessment, outcome measurement, missing data, and selective reporting. An increase of 10 000 particles/cm^3 in UFP assessed by central outdoor measurements, showed decreases in the standard deviation of the normal-to-normal intervals (SDNN) by 4.0 % (95 % CI 7.1 %, -0.9 %) and root mean square of successive R-R interval differences (RMSSD) by 4.7 % (95 % CI 9.1 %, 0.0 %) within 6 h after exposure. This finding highlights that UFP may contribute to the onset of cardiovascular events through autonomic dysregulation.

2.2.2.2. Respiratory effects

RIVM study – Short-term exposure to aircraft UFP (Janssen et al., 2019):

- Panel study: Among primary school children, statistically significant associations were found between exposure to UFP and an increase in acute respiratory symptoms (shortness of breath and especially wheezing) and bronchodilator use. This was true both for total UFP, for UFP mainly from air traffic and for UFP mainly from road traffic. In addition, there was a significant association between UFP mainly from road traffic and reduced lung function in the morning (FEV1 measured at home). No consistent relationships with UFP exposure were found for lung function measurements at school and in the evening (at home) and for NO in exhaled air.
- Volunteer study: A 5–95th percentile increase in exposure to UFP (i.e. 125 400 particles/cm^3) was associated with a decrease in FVC of −73.8 mL (95 % CI −138.8 – −0.4). This effect was associated with particles < 20 nm. For other lung function parameters such as fractional exhaled nitric oxide (marker of asthma and airway inflammation) and oxygen saturation (parameter used to assess respiratory function), no statistically significant associations were observed with UFP (Lammers et al., 2020).
- Toxicological study: Human bronchial epithelial cells (Calu-3) exposed to low UFP doses (0.09 to 2.07 µg/cm²) showed that pro-inflammatory responses still can be activated at high cell viability (> 80 %) and low cytotoxicity. No significant differences in reactivity were observed between UFP exposure from aviation and road-traffic (He et al., 2020).

RIVM study – Long-term exposure to aircraft UFP (Janssen et al., 2022b):

- The mortality cohort study suggested no association with respiratory mortality, lung cancer and COPD mortality.
- The medication use cohort study suggested no association with asthma/COPD.
- The cross-sectional public health monitor (surveys) suggested no association with asthma/COPD.

Altogether, Janssen et al. (2022a) concluded that increased short-term exposure to UFPs is associated with **acute health effects** on the respiratory tract (adults and children). Acute exposure can increase medication use. No indications were found for effects of long-term exposure to UFP from aviation around Schiphol on respiratory system disorders.

In 1999, the Dutch RIVM performed a survey, a blood test, and a skin-prick test among 2500 primary school children (age 7-12) living in the area of Schiphol airport (van Vliet et al., 1999). The aims of this study included to check whether the exposed group showed differences in lung function and the prevalence of respiratory symptoms, and to verify if an association between exposure to air pollution (air and road traffic) and respiratory health could be established. While the study showed differences between the different populations near Schiphol for respiratory symptoms, lowered pulmonary function and antibodies in the blood (IgE), no trend could be established relating these conditions to the distance from the airport.

However, the average prevalence in all residential areas near Schiphol together was elevated, when compared to populations located far from the airport of busy road traffic. The components (NO_2, soot and benzene) for which Van Vliet et al. (1999) found an association with outdoor air, are more related to road traffic than air traffic.

Findings on long-term exposure similar to those of Janssen et al. (2022b) were inferred in a Canadian study by Weichenthal et al. (2017). This study among approximately 1.1 million residents of Toronto, did not observe clear evidence of positive associations between long-term exposure to ambient UFPs and lung cancer, adult-onset asthma, and chronic obstructive pulmonary disease (COPD) after adjustment for ambient fine particulate air pollution ($PM_{2.5}$), NO_2, and other individual/neighbourhood-level covariates .

Habre et al. (2018) conducted a randomised crossover study of 22 non-smoking adults with asthma, exposed during mild walking activity in a high UFP zone near the airport of Los Angeles. It was shown that increased acute systemic inflammation followed after exposure to airport-related UFPs, while health effects of roadway traffic exposure were distinct.

2.2.2.3. Adverse birth outcomes

RIVM study – Short-term exposure to UFP (Janssen et al., 2019):

- Not investigated

RIVM study – Long-term exposure to aircraft UFP (Janssen et al., 2022b):

- The perinatal registration study suggested no association with low birth weight and Apgar scores. A possible association was suggested with preterm birth and small gestational age. A probable association was suggested with congenital anomalies.

Altogether, Janssen et al. (2022a) concluded that the evidence is **indicative** (also from literature) that exposure to aircraft UFP during pregnancy is negatively associated with birth outcomes (preterm birth, low birth weight, birth defects).

A population-based study by Wing et al. (2020) studied preterm birth (PTB) among infants exposed *in utero* to UFP from aircraft emissions. Birth data were used of 174 186 mothers (including 15 134 PTBs) living less than 15 km from Los Angeles International Airport. *In utero* exposure was estimated using a validated geospatial UFP dispersion model. A significant association was found, the odds ratio (OR) per interquartile range (IQR, 9200 UFP particles/cm³) increase was 1.04 (95 % CI 1.02, 1.06). When comparing the 4th quartile of UFP exposure to the 1st quartile, the OR for PTB was 1.14 (95 % CI 1.08, 1.20), adjusting for maternal demographic characteristics, exposure to traffic-related air pollution, and airport-related noise. It was concluded that aircraft emissions play an etiologic role in PTBs, independent of noise and traffic-related air pollution. Later, Wing et al. (2022) studied the interaction of aircraft noise and vehicle traffic-related air pollution, affecting preterm birth (PTB). The results suggested a potential synergism between airport-related noise and Traffic Air Pollution exposures on increasing the risk of PTB in this metropolitan area.

Using a multi-scale modelling approach and concentrations-response functions, Yim et al. (2015) estimated the number of premature deaths due to populations exposure to aviation attributable PM2.5 (not UFP!) and ozone. It was stated that aviation emissions annually cause ca. 16 000 (90 % CI 8300 – 24 000) premature deaths.

2.2.2.4. Metabolic effects

RIVM study – Short-term exposure to UFP (Janssen et al., 2019):

- Not investigated

RIVM study – Long-term exposure to aircraft UFP (Janssen et al., 2022b):
- The mortality cohort study suggested no association with diabetes.
- The medication use cohort study suggested no association with diabetes.
- The cross-sectional public health monitor (surveys) suggested a clear association with diabetes (self-reported and medication).

Altogether, Janssen et al. (2022a) concluded that the evidence for effects of long-term exposure to aircraft UFP on diabetes is **insufficient**, based on the RIVM research and the existing results.

2.2.2.5. Neurodegenerative diseases

RIVM study – Short-term exposure to UFP (Janssen et al., 2019):
- Not investigated

RIVM study – Long-term exposure to aircraft UFP (Janssen et al., 2022b):
- The mortality cohort study suggested a possible association with Alzheimer's disease and an inverse association with neurogenerative disease as primary endpoint, and Parkinson's disease and dementia as secondary endpoints.
- The medicine use cohort study suggested no association with Parkinson's disease, but a clear association with medication for dementia.

Altogether, Janssen et al. (2022a) concluded that the evidence for effects of long-term exposure to aircraft UFP on neurodegenerative diseases is **insufficient**. There is insufficient information to assess whether or not there is a link.

2.2.2.6. Psychological complaints

RIVM study – Short-term exposure to UFP (Janssen et al., 2019):
- Not investigated

RIVM study – Long-term exposure to aircraft UFP (Janssen et al., 2022b):
- The medication use cohort study suggested no association with antidepressants (> 20 years) and an inverse association with antidepressant use between 6-19 years and ADHD medication between 16 and 19 years.
- The cross-sectional public health monitor suggested no association with psychological distress (including antidepressant medication)

Altogether, Janssen et al. (2022a) concluded that the evidence for effects of long-term exposure to aircraft UFP on psychological complaints is **insufficient**. There is insufficient information to assess whether or not there is a link.

2.2.2.7. Mortality & general health

RIVM study – Short-term exposure to UFP (Janssen et al., 2019):
- Not investigated

RIVM study – Long-term exposure to aircraft UFP (Janssen et al., 2022b):
- The mortality cohort study suggested no association with overall mortality.
- The cross-sectional public health monitor suggested no association with self-perceived health.
- The perinatal registration study suggested no association with infant mortality.

Altogether, Janssen et al. (2022a) concluded that no indications were found for an association between long-term exposure to aircraft UFP and total mortality, infant mortality and perceived general health.

Using a concentration-response function, Yim et al. (2013) estimated that 110 (90 % CI: 72–160) early deaths occur in the UK each year (based on 2005 data) due to UK airport emissions. While most studies on the health impact of aviation emissions mainly considered landing and take-off emissions near airports, Barrett et al. (2010) estimated the global mortality attributable to aircraft cruise emissions. Ca. 8000 premature mortalities were estimated per year, representing ca. 80 % of the total impact of aviation (including landing and take-off emissions) and 1 % of air quality-related premature mortalities from all sources.

2.2.2.8. Cancers

A large descriptive, population-based ecological study was executed by Visser et al. (2005) on the incidence of cancer in the area near Schiphol, using numbers of the regional cancer registry. Subsequently, these data were compared with national cancer incidences. The following observations were made:

- In total, 13 207 cancer cases were diagnosed in the study area between 1988-2003, which was close to the expected number (Standardised Incidence Ratio [SIR] 1.02). This trend remained continuous during the entire study period.
- The incidence of haematological malignancies was significantly increased (SIR 1.12, 95 % CI 1.05-1.19), mainly due to high rates of non-Hodgkin lymphoma (SIR 1.22, 95 % CI 1.12-1.33) and acute lymphoblastic leukaemia (SIR 1.34, 95 % CI 0.95-1.83). In contrast, Hodgkin lymphoma incidence was decreased (SIR 0.78, 95 % CI 0.58-1.04). The increased numbers were consistently elevated during the entire time interval, especially in males. The strongest increase existed in the core zone. The pattern of non-Hodgkin lymphoma in the Schiphol area was suggested to possibly indicate a relation with pollution as found in urban areas. However, it was concluded that a specific relationship to ambient air pollution caused by aircraft emissions is not obvious, based on the results of ambient air quality monitoring and source appointment. Other explanations related to e.g. intensive agricultural activities were ruled out.
- The incidence of respiratory cancers was significantly decreased (SIR 0.94, 95 % CI 0.90-0.99), due to low rates in males (SIR 0.89). It was suggested that this may be due to regional differences in smoking habits. Slightly increased incidences in females were linked to elevated smoking behaviour among women in moderately urbanised areas like the Schiphol surroundings, compared to rural areas. No link to aircraft pollution could therefore be found.
- Cancer incidence among children (74 cases < 15 years) was relatively high (SIR 1.26, 95 % CI 0.99-1.58) due to higher numbers of acute lymphoblastic leukaemia (23 cases, SIR 1.59, 95 % CI 1.01-2.39).
- Overall cancer incidence was slightly higher in the core zone of the study area than in the ring zone (rate ratio 1.05, 95 % CI 1.01-1.10) due to the higher incidence of respiratory, prostate and female genital organ cancers.
- It was concluded that the overall incidence of the Schiphol area was similar to the national incidence. The increase in haematological malignancies could not be explained by the higher air pollution near Schiphol (Visser et al., 2005).

A cohort study by Wu et al. (2021) studied the association between airport-related UFP and risk of incident malignant brain cancer (n=155) and meningioma (n=420) diagnosed during 16.4 years of follow-up among 75 936 men and women residing in Los Angeles County. Aircraft UFP exposure was estimated for participants living within a 53 km by 43 km grid around the Los Angeles International Airport, from 1993 to 2013. The study adjusted for sex, ethnicity, education and neighbourhood socioeconomic status. The following Hazard Ratios were found:

- Malignant brain cancer risk in all ethnic groups combined increased 12 % (HR 1.12, 95 % CI 0.98-1.27) per interquartile range (IQR) of airport-related UFP exposure (ca. 6700 particles/cm³).
- However, a more pronounced outcome was found when stratified per ethnicity. As African Americans had the highest exposure in the cohort, a significantly elevated HR of 1.32 (95 % CI 1.07-1.64) was found for this subgroup per IQR in UFP exposure.
- No association was found between the risk of meningioma and UFP exposure.

The results of the study of Wu et al. (2021) are in line with the observations of Weichenthal et al. (2020), who studied within-city spatial variations in ambient UFPs across Montreal and Toronto. The study identified 1400 incident brain tumors during the follow-up period among 1.9 million adults. Each 10,000/cm increase in UFPs was positively associated with brain tumor incidence (HR = 1.112, 95 % CI = 1.042, 1.188) after adjusting for $PM_{2.5}$, NO_2, and sociodemographic factors. Applying an indirect adjustment for cigarette smoking and body mass index strengthened this relation (HR = 1.133, 95 % CI = 1.032, 1.245). $PM_{2.5}$ and NO_2 were not associated with an increased incidence of brain tumors.

A large-scale epidemiological study on the incidence of cancers has not been conducted around Brussels airport. Such a study could provide more clarity on the Belgian situation.

2.2.2.9. Genotoxic effects (experimental and human biomonitoring studies)

In 1991, the Dutch RIVM performed Ames tests on some aerosol samples taken in the vicinity of Schiphol airport (van den Anker et al., 1991). These authors found no increased mutagenicity, which was in accordance with a parallel public health survey in the Schiphol region which found cancer mortalities similar to that of urban areas.

Cavallo et al. (2006) evaluated the genotoxic and oxidative effects of civil airport exposure, performing different tests on air samples, mainly focusing on PAHs (27 703 µg/m³ total PAHs in airport apron, 17 275 µg/m³ in airport building and 9494 µg/m³ in terminal/office area). They also compared biomarkers for genotoxic effects between airport personnel (n=41) and a control population (n=31). Tests included micronucleus and Fpg-modified comet assays (on lymphocytes and exfoliated buccal cells), chromosomal aberrations (CA) and sister chromatid exchange (SCE) analyses. For the comet assay, tail moment (Tail moment = tail length x % of DNA in the tail) values from Fpg-enzyme treated cells (Tmenz) and from untreated cells (TM) were used as parameters of oxidative and direct DNA damage. The following results were observed:

- Sister chromatid exchange (SCE): higher mean value of SCE frequency among the exposed group (4.6 vs. 3.8)
- Structural chromosomal aberrations (CA): more breaks (x 1.3, up to 2.0-fold) and more fragments (0.32 % vs 0.00 %) among the exposed group.
- Comet assay exfoliated buccal cells: TM 118.87 versus 68.20 (x 1.74; p < 0.001); TMenz 146.11 versus 78.32 (x 1.87, p < 0.001)
- Comet assay lymphocytes: TM 43.01 versus 36.01 (x 1.19, p = 0.136); TMenz 55.86 versus 43.98 (x 1.27; p = 0.003)
- Oxidative DNA damage was only found in cells from the exposed population (9.7 % exfoliated buccal cells; 14.6 % lymphocytes).

Bendtsen et al. (2019) evaluated the potential occupational exposure risk by analysing the particles from a non-commercial airfield and from the apron of a commercial airport. Toxicity of the collected particles was evaluated alongside NIST standard reference diesel exhaust particles (NIST2975) in terms of acute phase response, pulmonary inflammation, and genotoxicity after single intratracheal instillation in mice. Pulmonary exposure of mice to particles collected at two airports induced acute phase response, inflammation, and genotoxicity similar to standard diesel exhaust particles and carbon black nanoparticles, suggesting similar physicochemical properties and toxicity of jet engine particles and diesel exhaust particles.

2.3 Conclusions

The RIVM study found indicative evidence for an association between chronic aircraft UFP exposure and cardiovascular diseases and birth outcomes. Insufficient evidence was found for an association with neurological and metabolic effects. No indications were found for an association between chronic UFP exposure and respiratory diseases and general health (including mortality). On the other hand, acute respiratory effects were observed after short-term UFP exposure, while also indications exist for cardiovascular effects due to short-term exposure (Table 7, Janssen et al., 2022a).

Table 7. *Conclusions of the RIVM study with classification of the evidence for the association between adverse health effects and long-term and short-term exposure to aircraft UFPs. (Source: Janssen et al., 2022a)*

	Long-term exposure	**Short-term exposure**
Respiratory effects	No indications	Effects founds
Cardiovascular effects	Indicative evidence	Indications for effects
Adverse birth outcomes	Indicative evidence	/
Neurodegenerative diseases	Insufficient evidence	/
Psychological complaints	Insufficient evidence	/
Metabolic effects	Insufficient evidence	/
General health	No indications	/

Besides, genotoxic effects of aircraft UFP have been proven, while a recent cohort study by Wu et al. (2021) found an association between long-term exposure to aircraft UFP exposure and malignant brain cancer. This study confirmed earlier findings by Weichenthal et al. (2020).

In general, the outlined effects are in line with the wider literature evidence on the effects of UFP (not source specific) on human health (e.g. Gezondheidsraad, 2021). The results of the RIVM study give an idea of the potential effects of aircraft UFP but cannot simply be extrapolated from Schiphol to Brussels Airport, as the geographical context is very different (i.e. much higher population density in the immediate vicinity of Brussels Airport than in Schiphol). Moreover, it is not straightforward to distinguish between health effects of UFP that come from aviation or road traffic in the area. **A specific study of the context around Brussels Airport is needed.**

Various other pollutants are not specifically measured around airports or related to airport emissions, which does not mean they are not present in harmful concentrations or could not interact, as exposure is multiple (various air pollutants and noise) (see Bendtsen et al., 2021).

The question could be raised whether the focus on aircraft-induced air pollution (and noise pollution) should not be widened to other aspects of airport activity. Air pollution from aircraft is indeed a problem of its own, but the impact of the airport on ambient air quality is much larger because more flights and especially more cargo activity will result in a significant rise in road traffic, both at night and during the day, with heavy trucks in a densely populated area that already faces saturated traffic conditions and the concomitant noise and air pollution. The same holds for noise pollution, especially with respect to Brussels Airport's ambition to more than double cargo traffic by 2032. Based on the EIA, it is to be expected that in 2032, the number of trucks will increase by 43 % compared to 2019. Recent data and prospects can be obtained from the EIA report that became available to the public in December 2023.

3 Windows of exposure: daytime, early morning and night flights

3.1 Aircraft noise

With regard to the second question submitted to the SHC, namely:

"Are there any differences in the effects of daytime, early morning and night flights?"

The discussions in the previous sections, which were based on an extensive review of the literature, have led the SHC to conclude that not only the average noise levels, but more so the number of flyovers where the noise level exceeds a certain threshold have a significant impact on the type and severity of the health effects induced by aircraft noise (e.g., Nassur et al., 2019a; Yang et al., 2022).

However, the impact of aircraft noise on health varies depending on the time of day when exposure occurs. Notably, considering the critical role of sufficiently good quality **sleep** in both physical and mental well-being, night-time exposure to aircraft noise tends to have the most severe effects. (cf. section 1.4 above). Both self-reports and polysomnographic studies have shown that night-time exposure to aircraft noise results in a higher probability of awakening, more time to sleep onset, and more sleep disturbance (e.g. Passchier-Vermeer, 2002; Basner & McGuire, 2018; WHO 2018, Nassur et al., 2019a). For example, in a study examining objective parameters of sleep quality through wrist actigraphy among a population sample residing near two French airports, it was found that higher levels of aircraft noise and increased frequency of aircraft noise events correlated with prolonged sleep onset latency (SOL), extended periods of wakefulness after sleep onset (WASO) and reduced overall sleep efficiency (SE) (Nassur et al., 2019 b).

In addition to L_{night}, it should be mentioned that frequency of exposure has a dominant influence on polysomnographically objectified sleep quality. To properly characterise exposure frequency, more qualified single noise event indicators (L_{Amax} or SEL) should be used.

Also, as discussed in section 1.4.2 above, sleep is a dynamic process in which the brain goes through several rounds of unidentical cycles, each of which can again be subdivided into a set of characteristic stages. These stages contribute to varying degrees to the recuperative power of sleep, to which the deep sleep stage is critical. As the night progresses, the time spent in deep sleep decreases. As a result, flights occurring in the early morning have a significant effect on the perception of sleep quality, as sleepers are more easily woken up, but also on objective sleep parameters. The NORAH study showed that whilst the night-flight ban in Frankfurt resulted in fewer aircraft-noise-induced awakenings until the early morning (the night flight ban ends at 5 am in Frankfurt), the higher concentration of flights in the early morning (5 to 7 am) corresponded with a reverse effect on sleep quality perception (Müller et al., 2015) . Whilst flights in the evening did not appear to influence sleep onset latency in Frankfurt, the latter increased significantly around Cologne-Bonn airport, which did not have a night-flight ban (Müller et al., 2017).

Of course, flight restrictions are in place for Brussels Airport during night-time hours, which span from 23:00 to 05:59 local time. At the time of writing, these measures encompass various regulations, including limiting the total number of night flights to 16 000 per year, with a maximum of 5 000 take-offs allowed[20]. Additionally, there are specific bans on take-offs between 1:00 and 6:00 am on Fridays, as well as from 00:00 am to 6:00 am on Saturdays and Sundays, and aircraft noise is managed through a quota count system (ENVISA, 2022: 187). Still, it is important to note that, according to the definition of the federal government, the night-

[20] Ministerial decree of 21 January 2009 amending the Ministerial Decree of 3 May 2004 on noise management at Brussels National Airport

time period spans from 11 pm to 6 am. However, the acoustic night, i.e., the period during which the L_{night} level is measured, extends from 11 pm to 7 am. As observed previously (cf. section 1.4.3 above), individuals typically spend more time in lighter sleep stages during the latter part of the night. Yet these stages are more easily disrupted by external stimuli, making early morning flights more likely to affect sleep quality. In 2019, there were 10 029 flights during this one-hour window (7 804 in 2022) (cf. section I above). Consequently, the SHC emphasises the importance of including the time between 6 and 7 am in the night-time period.

Similarly, variations in **annoyance** depending on the time of the day have been identified (Hume et al., 2003; Hoeger, 2004; Baudin et al., 2021; Bartels et al., 2022). Therefore, unsurprisingly, rising nocturnal flyover frequencies have been associated with a statistically significant increase in the portion of annoyed residents (e.g. Quehl, 2021). Airports such as Leipzig/Halle, which does not have a ban on night flights, are a case in point. Leipzig/Halle airport had 80 000 aircraft movements in 2019, about half of which occurred in the L_{night} period (mainly airfreight traffic). Here, aircraft noise exposure was associated with an OR of high annoyance of 12.7 (95 % CI: 9.37–17.10 per 10 dB(A) L_{den}) and 19.71 (95 % CI: 11.65–33.35 per 10 dB(A) L_{night}) for L_{den} and L_{night}, respectively (Starke et al., 2023).

A study by Schmitt et al. (2014) has revealed that night-time aircraft noise significantly affects endothelial function in patients with or at risk for cardiovascular disease. In addition, a mortality study (Saucy et al., 2021) with high precision aircraft noise modelling and a case-crossover design from the Swiss National Cohort around Zürich Airport between 2000 and 2015 revealed that exposure levels 2h preceding death were significantly associated with night-time (11 pm–7 am) mortality for all causes of CVD (OR 1.44, 95 % CI 1.03–2.04) for the highest exposure group (L_{Aeq} > 50 dB(A) vs < 20 dB(A))]. The odds of night-time cardiovascular mortality (all causes) were significantly increased for 2h-L_{Aeq} values above 40 dB(A).

Table 8: *Associations between night-time mortality from cardiovascular cause and noise exposure groups 2 hours preceding death (Source: Saucy et al. 2021: 839)*

Exposure groups	All cardiovascular diseases			Ischaemic heart diseases			Myocardial infarction			Heart failure		
	n	**OR**	**95% CI**	***n***	**OR**	**95% CI**	***n***	**OR**	**95% CI**	***n***	**OR**	**95% CI**
<20 dB	4245	1		1797	1		527	1		229	1	
20–30 dB	824	1.08	(0.92–1.26)	360	1.15	(0.90–1.47)	101	1.11	(0.67–1.79)	69	1.11	(0.63–1.99)
30–40 dB	1169	1.23	(1.00–1.51)	513	1.1	(0.80–1.51)	156	0.86	(0.45–1.64)	108	**2.08**	**(1.01–4.29)**
40–50 dB	1157	**1.33**	**(1.05–1.67)**	479	1.13	(0.78–1.64)	152	0.93	(0.46–1.88)	74	2.07	(0.93–4.61)
>50 dB	246	**1.44**	**(1.03–2.04)**	117	1.64	(0.96–2.79)	35	1.62	(0.62–4.25)	16	3.09	(0.94–10.23)
Trend	P for trend = 0.01			P for trend = 0.18			P for trend = 0.57			P for trend = 0.05		

Statistically significant results at level alpha = 5% are marked in bold, adjusted for NO_2, temperature, precipitation, and holiday.

A significant increase in odds of mortality due to heart failure was observed when exposure levels exceeded above five events above the threshold of 55 dB(A) (NAT_{55}) within the 2h window prior to death. However, for deaths occurring during daytime hours, there was no consistent indication of increased risk (Saucy et al., 2021).

Another study at Heathrow airport (Itzkowitz et al., 2023) confirmed the results. This study found small associations between aircraft noise and cardiovascular disease admissions mainly related to late evening and early night-time exposures. An increased risk for all CVD admissions was estimated per 10 dB(A) increment in noise during the previous evening (Leve OR = 1.007, 95 % CI 0.999–1.015), particularly from 10-11 pm (OR = 1.007, 95 % CI 1.000–

1.013), and the early morning hours 4.30 – 6 am (OR = 1.012, 95 % CI 1.002–1.021), but no significant associations with day-time noise.

Benz et al. (2022) reported a study showing a stronger relationship between the intermittency ratio and the risk of myocardial infarction than continuous noise levels of the same average level, with the most problematic time of day being between 5 and 6 in the morning.

The aforementioned considerations underscore the critical significance of night-time and even early-morning exposure. However, it's essential to note that daytime exposure to aircraft noise has also been linked to health effects, including high annoyance, and learning difficulties in school children (see discussions in sections 1.3.2 and 1.5.1 above).

Another factor that needs to be taken into account is the fact that **gradients in temperature and wind affect the intensity of the noise people are exposed to** (IST, 2012). Indeed, in sunny and windless weather conditions, the temperature decreases sharply with altitude. This causes the sound to deflect upwards and resulting in lower noise levels on the ground. Conversely, as the temperature gradient reverses at night-time, temperatures will be cooler near the ground, especially if the weather is clear and the wind is calm. This will cause the sound to deflect towards the ground, with higher noise levels as a result. Wind has a similar effect: an upwards wind gradient decreases the noise levels on the ground, whilst a downwards wind gradient increases it. The combination of these factors can result in fluctuations in noise levels by 10 to even 20 dB(A) at 1 km from a major noise source (IST, 2012).

3.2 Aircraft pollutant emissions

The concentration of air pollutants in the atmospheric boundary layer (lower troposphere) is influenced by both the weather conditions and the time of the day (e.g. Sorbjan, 2003; Pérez et al., 2020; UCAR, 2024). The dispersal of pollutants is forced by the unstable state of the air due to advection (wind: horizontal transport although mediated by the shape of the built environment) and convection, which causes atmospheric turbulence.

A stable state occurs when the air near the ground is colder than the higher air. This creates a temperature inversion where cold air is trapped near the ground by a layer of warmer air. As a result, the dispersion and dilution of pollutants are limited. Pollutants can remain trapped close to the ground, leading to an accumulation of pollution around the source of emission. At night, the temperature inversion is usually formed near the surface (Sorbjan, 2003). Strong stability is observed shortly after sunrise (early morning) and shortly before sunset (late afternoon, early evening). For example, a study in Los Angeles by Choi et al. (2016) found that the limited dispersion capacity and the stable surface layer in the morning led to higher UFP concentrations than in the afternoon. It is therefore not advisable to concentrate the emission of pollutants (such as UFP) at these times of the day.

4 Methodology of studies on health effects

The third question put to the SHC was the following: *"Is there any evolution in the assessment of these effects in the international scientific literature, and have any good studies been conducted on this subject in the vicinity of comparable airports in Western Europe whose methodology could be useful in Belgium?* "

Most studies focus on noise exposure. Air pollution, however, should not be neglected. Moreover, few studies address the combined effect of air pollution and noise exposure.

4.1 Aircraft noise

The issue of the health effects of aircraft noise had already been addressed by the SHC in 2011 in advisory report no. 8603 on the environmental effects of traffic on health, as well as in 2013 in advisory report no. 8738 on windfarms. The SHC found that the main effects of (traffic, including aircraft) noise were annoyance, sleep disturbance, effects on the cardiovascular system and cognitive effects.

At the time, the effects of aircraft noise on cognitive disorders, annoyance and hypertension had been explored at European level, using a combination of methods in several major European projects, including

- 5A - Attitudes to Aircraft Annoyance Around Airports[21], conducted on the basis of surveys between 2001 and 2003, which established a degree of correspondence, but not exact coincidence, between standard noise measurements and aircraft annoyance for Manchester, Lyon and Bucharest airports.
- HYENA - Hypertension and Exposure to Noise near Airports. This cross-sectional study, funded by the European Community, assessed the impact of aircraft and road traffic noise on cardiovascular health around 7 major European airports (Athens, Greece; Milan/Malpensa, Italy; Amsterdam/Schiphol, Netherlands; Stockholm/Arlanda and Broma, Sweden; Berlin/Tegel, Germany; London/Heathrow, United Kingdom). The data were collected in 2004-2006, but regular follow-ups are conducted. A positive correlation was found between aircraft noise and the consumption of anxiolytics and antihypertensive drugs.
- RANCH - Road Traffic and Aircraft Noise Exposure and Children's Cognition and Health. This study, conducted in 2002-2003, focused specifically on children and found significant effects of environmental noise on reading comprehension and annoyance (Stansfeld, et al. 2005).

Since then, many more reports and studies have associated aircraft noise with (high) annoyance and sleep disturbance as well as with cognitive impairment, hypertension, and cardiovascular disease (see above). These studies have already been referred to in the discussion above. Below is a brief (non-exhaustive) overview of some of the most significant projects that have been conducted in Europe.

Most importantly, there are of course the revised WHO guidelines for environmental noise in Europe[22], which were issued in 2018, and which were already extensively referred to above. Compared to the previous guidelines from 2009, the new guidelines are based on a GRADE-approach, the recommendations formulated according to the source of the noise (air, road, rail), new noise sources (windfarms, etc.) are taken into account and two noise indicators are used (Ld_{en} and L_{night}). Various health effects were selected to be considered in the systematic review and were classified as either "critical' or "'important but not critical" for the development of recommendations on environmental noise (mostly depending on the strength of the

[21] https://www.eurocontrol.int/sites/default/files/library/026_Attitudes_to_Aircraft_Annoyance_Around_Airports.pdf (accessed on 24/4/2024)
[22] https://www.who.int/europe/publications/i/item/9789289053563 (accessed on 24/4/2024)

evidence). As each effect can be monitored in terms of several different indicators, specific indicators were prioritised for each type of critical effect on the basis of their representativeness and validity. The recommended exposure levels referred to above were developed on this basis.

In France, ANSES issued a report entitled "*The extra auditory health impacts of environmental noise*" in 2013, whilst in 2020[23], they followed with a report that focused more specifically on the health impacts linked to aircraft noise and provided a summary of all major international studies published since 2012.

The DEBATS programme[24] (*Discussion sur les effets du bruit des aéronefs touchant la santé*) was launched in 2012 and is supervised by the French Airport Pollution Control Authority (ACNUSA). It is the first large-scale research programme in France that has assessed the potential effects of aircraft noise on sleep in the vicinity of 3 airports (Paris-Charles-De-Gaulle, Lyon-Saint-Exupéry, Toulouse-Blagnac). Gustave Eiffel University conducts the research, which has been published on the DEBATS-website as well as in scientific journals since 2016.

Three types of complementary methodological approaches are used:

- an *ecological study* at municipal level, which aims to relate aggregate health indicators to the weighted average level of noise exposure. These health indicators are based on drug prescriptions, medical consultations, hospitalisations, time off work, and mortality data.
- an *individual longitudinal stud*y: individuals (1244 participants) were followed for at least 4 years with repeated measurements of their state of health. Data for this study come from a questionnaire (effects on sleep and the cardiovascular system, anxiety depressive disorders and perceived annoyance) as well as measurements of blood pressure, heart rate and salivary cortisol concentration (marker of stress states).
- A clinical *sleep study* conducted among participants (112 participants) of the longitudinal study aimed at obtaining a detailed and specific description of the acute effects of aircraft noise on the quality of sleep. On the one hand, two types of acoustic measurements were carried out among participants: measurements taken with sound level meters installed in the participant's home, inside the bedroom and on the exterior wall of the building for a period of 7 days, and a measurement of individual exposure over 24 hours using a noise dosimeter. In addition, the quality of the participant's sleep was assessed using actimetric recordings combined with a sleep diary and an electrocardiogram. This study was conducted in cooperation with **Bruitparif,** which was involved in the instrumentations at the participants' homes (acoustic measurements, actimetric -limb movement- measurements, heart rates).

The results of the DEBATS programme are in line with the findings of previous studies carried out abroad, as they provide additional support for the finding that the main adverse effect associated with aircraft noise is (high) annoyance and a severely impaired sleep quantity and quality. In addition, aircraft noise is strongly suspected to be associated with effects on the endocrine and cardiovascular system, perceived health status, psychological health, and cognitive effects. The results of the research programme are regularly published on the DEBATS website[25] and are used by organisations such as ACNUSA to fulfil their mission objectives.

Due to the profound impact of aircraft noise on both subjective and objective sleep parameters, safeguarding residents near airports from aircraft noise-induced sleep disturbance remains of paramount concern. In France, numerous regulations have been implemented to address this

[23] https://www.anses.fr/fr/system/files/AP2020SA0053Ra.pdf (accessed on 24/4/2024)
[24] http://debats-avions.ifsttar.fr/index.php (accessed on 24/4/2024)
[25] http://debats-avions.ifsttar.fr/publications.php (accessed on 24/4/2024)

issue: some airports have instituted curfews, while others have banned the noisiest aircraft from operating during all or specific hours of the night. In its 2021 annual report, ACNUSA recommended the establishment of a national observatory to monitor night-time movements at major French airports[26].

In Germany, the **NORAH-study** (Noise-Related Annoyance, Cognition, and Health) was conducted over 3 years (2011, 2012, 2013) to gather comprehensive and up-to-date scientific information on the effects of traffic noise on the health and quality of life of affected populations. Led by nine independent scientific institutions, the study focused on assessing the health effects of aircraft noise around Frankfurt airport before and after the opening of a new runway designed to accommodate an additional 200 000 flights per year, alongside the implementation of a night flight ban spanning 6 hours from 11 PM to 5 AM. More specifically, the study looked at noise-induced annoyance, sleep disturbance, cardiovascular diseases, cognitive impairment and mental health, as well as overall well-being and health related quality of life.

a) *Noise annoyance* was mainly investigated by means of online and telephone surveys. The study further explored the potential impact of implementing the night flight ban on aircraft noise-induced annoyance. It looked at the combined effects of aircraft and railway noise, as well as aircraft and traffic noise, while analysing exposure-response relationships. This involved calculating the noise exposure from aircraft, railway and road traffic for each participant at their home in the year prior to the survey.

b) *Sleep disturbance* was assessed through objective measurements, by means of polysomnography and ECG + actigraphy, and subjective evaluations via questionnaires. Continuous monitoring of noise pressure levels and individual noise events at the sleeper's ear allowed for linking each noise incident to polysomnography outcomes. The study compared the results of two bedtime groups: one exposed to aircraft noise in the evening and in the early morning, and another exposed solely in the early morning. Additionally, comparisons were drawn with a field study conducted near Cologne-Bonn airport, where no night-flight ban was enforced.

c) As regards the effects on *cardiovascular disease*, the study used data from three health insurance funds and linked them to current and past noise exposure.

d) The investigation into *children's cognition and health-related quality of life* focused on three key indicators: reading and related abilities, self-reported quality of life and self-reported annoyance. Specifically targeting second graders (aged 7-9) attending schools around Frankfurt/Main airport, the study calculated noise exposure levels at both school and home for each child using radar data from the Flight Track and Aircraft Noise Monitoring System (FANOMOS) spanning a period of 12 months prior to data collection. Schools were categorised into groups based on prevailing sound levels during the day, with particular care taken to ensure comparability across groups in terms of factors such as migration background and socio-economic status. Measurements were gathered through a combination of in-school tests (reading, non-verbal intelligence etc.) and questionnaires.

Lastly, there is the ANIMA project platform[27], which is a large consortium gathering 22 partners throughout Europe – airports, aviation research centres, universities, SMEs and NGOs from 11 countries. It is dedicated to research into the effects of aircraft noise and offers a

[26] https://www.acnusa.fr/sites/default/files/2021-12/Rapport%20annuel%202021.pdf (accessed on 24/4/2024)
[27] https://anima-project.eu/noise-platform/main-page (accessed on 24/4/2024)

comprehensive overview of the relevant regulations, noise management strategies and other tools aimed at assisting airports and authorities in drawing up noise maps and arousal indices to test the ramifications of different scenarios involving diverse fleet and flight configurations. Moreover, the platform grants access to an enhanced Aviation Noise Research Roadmap, designed to aid decision-makers in defining future policy and research objectives. This resource encompasses scientific publications summarising key findings on addressing the impact of aircraft noise, alongside other tools geared towards fostering community involvement and promoting transparent working relationships among all stakeholders.

4.2 Air pollutants

Surprisingly, the impact of jet emissions on the health of residents living near international airports has not been extensively studied in the current international literature. A promising, pioneering study was recently conducted near Schiphol Airport by the Dutch RIVM (2022), using a multidisciplinary approach. Their methodology was threefold:

a) Measuring and modelling long-term UFP concentrations. The dispersion model was found to be suitable for application in epidemiological studies on long-term exposure.

b) Studies on health effects of short-term UFP exposures. Three studies were undertaken:
- A panel study with primary school children in residential areas at real-life UFP concentrations.
- A volunteer study with healthy adults exposed to high UFP concentrations in a mobile lab next to the airport (measurements on lung function, exhaled NO, ECG, blood pressure, oxygen saturation).
- A toxicological study (*in vitro*) with human bronchial epithelial cells.

c) Studies on health effects of long-term UFP exposures. Modelled UFP concentrations were linked to data of the exposed residents, using existing health registries and surveys.

Statistical analyses adjusted for the effects of noise in order not to overestimate the effects of UFP. For further details, see section 2.2.2. Based on a similar methodology, an interesting comparative study around Brussels Airport could be set up.

5 Impact on healthcare budgets and organisation

The fourth question was: *What impact do these effects have on healthcare budgets and organisation?*

In terms of healthcare organisation, a reduction in pollution and noise around Brussels airport can be expected to reduce the burden on morbidity and mortality, but certainly also on general practitioners and hospitals (severe pathologies). Based on balanced dose-response curves and disability-adjusted life years (DALYs) for both noise and UFP, the "Federaal Kenniscentrum – Centre federal d'expertise" (KCE) should quantify the impact on the healthcare budget. This calculation is beyond the remit of the Superior Health Council and this report can only document related information obtained in the literature or through the hearings mentioned above.

In general, each study should follow the same methodology: (1) assess the population exposed, (2) assess the fraction of that population that will suffer health effects, (3) evaluate the number of years of good health lost (disease and mortality) and (4) assign a monetary value to each lost year. Each of these steps may be criticised and could be refined through new investigations, but this will not remove the fact that, overall, each study assigns a considerable health economic cost to air traffic near airports.

In this context, the SHC notes that the exposure-response functions used for these calculations are based on systematic reviews of evidence such as those mentioned in the 2018 WHO report, where the quality of the evidence of the systematic reviews was assessed according to the GRADE guidelines, taking into account the study design, the number of participants and the included confounders. Whilst most studies did consider a set of confounders, there were cases in which this information was not available, thus potentially weakening the interpretation. In noise annoyance studies, non-acoustic factors[28] may account for up to 33 % of the variance. More specifically, the WHO experts graded the association of L_{night} with sleep disturbance and L_{den} with high annoyance, reading skills and oral comprehension in children as moderate quality. Similar conclusions were drawn from the systematic review by Basner & McGuire (2018).

5.1 Aircraft noise

There are studies on the health costs of noise exposure in terms of financial impact, whilst other studies examine the effects of exposure reduction measures.

In France, the total social cost of noise is estimated at 147 billion euros each year, based on existing data and studies (ADEME report). 97.8 billion euros are transport-related, of which 6.1 billion are due to aircraft noise.

In Belgium, a short study commissioned by "Bond Beter Leefmilieu" was conducted in 2023 by a French consulting bureau, ENVISA, to assess the health economic impact of aircraft noise on those living in the vicinity of Brussels airport. The authors used the same methodology as that used for a study conducted in 2021 by Bruitparif in Île de France (Social cost of aircraft noise in Île de France), and their results are in line with those of the latter. Using 2019 noise contours for L_{den} 45 dB(A) and L_{night} 40 dB(A), according to WHO 2018 guideline recommendations, it was calculated that 220 000 people are highly annoyed, which amounts to 4 380 DALYs and a health-economic cost of 578 million euros per year. High sleep disturbance in 109 999 people corresponded to 7 630 DALYs and an associated cost of 1

[28] Possible non-acoustic confounders in noise studies may be: gender, age, education, subjective noise sensitivity, extroversion/introversion, general stress score, co-morbidity, length of residence, duration of stay at dwelling in the day, window orientation of a bedroom or living room towards the street, personal evaluation of the source, attitudes towards the noise source, coping capacity with respect to noise, perception of malfeasance by the authorities responsible, body mass index, smoking habits.

billion euro per year. In addition, noise exposure puts 2 000 people at an increased risk of ischemic heart disease (very low-quality evidence base, WHO 2018) and 51 000 people at an increased risk of hypertension (low quality evidence base, WHO 2018), which may correspond to 6 800 DALYs and a potential health economic cost of almost 900 million euros per year.

5.2 Air pollutants

No data are available at the moment. It will not be possible to make this calculation until the exposure and disease burden on the residents living around Brussels Airport have been better studied. It is to be expected that only molecular epidemiological studies, in which also physiological functions are measured and in which associations with specific internal exposures are studied, can yield trustworthy insights. However, it should be possible to calculate the health economic cost of road traffic air pollution caused by trucks from and towards the airport.

IV CONCLUSIONS AND RECOMMENDATIONS

The fifth question put to the SHC was the following: "*What are the policy recommendations on this issue?*"

1 Reducing aircraft noise exposure

1.1 Ban on night flights

Given the substantial evidence showing (severe) **negative health effects**, which are primarily related to sleep disturbance, the SHC believes that a complete **ban on night flights** between 11 pm and 7 am is most desirable from a health perspective to protect the well-being of the approximately 163 518 residents within the L_{night} > 45 dB(A) noise contours of 2019. This measure should at least allow those living near the airport to benefit from **7 hours, ideally 8 hours, of sleep undisturbed by aircraft noise**. In addition, particular care should be taken to avoid a high concentration of flights in the shoulder hours early in the morning and late in the evening.

1.2 Land use planning

It is also important to recall that the Regions are responsible for land use planning. It follows that both Brussels-Capital Region and Flanders should put a stop to further urbanisation for residential purposes in the affected areas, in contrast with current practice.

1.3 Flight paths

The flight paths should be aligned in such a way that no one experiences an unacceptable nuisance in terms of the number of exceedances of the 60 dB(A) $L_{A,max}$ threshold, especially at night. In keeping with this concept (i.e. the prime importance of both peak intensity ($L_{A,max}$/SEL) and the number of exposures), the herewith related number of sleep-disturbed people and the number of annoyed people should be kept as low as possible. Not only should no one be subjected to an unacceptable level of exposure, but care should be taken to keep the number of highly annoyed people as low as possible.

1.4 No further increase in flight numbers

An expansion of the airport with the aim of achieving an increase in flight numbers is not acceptable given the current high burden on the neighbouring residents in terms of air pollution and noise exposure.

1.5 Limited exposure of schoolchildren

In light of the growing body of evidence that chronic aircraft noise impairs children's cognition and learning, the SHC believes that both L_{Aeq} and the number of daily overflights exceeding the 60 dB(A)-threshold that **school children** are exposed to should be reduced. It is doubtful whether soundproofing schools would contribute towards reducing the noise children are exposed to, whilst implementing this measure would entail that particular care should be taken to ensure sufficient ventilation (see SHC advisory report no. 9616 of 2021).

1.6 Unrealistic soundproofing of bedrooms

The same holds for the soundproofing of bedrooms: it is unrealistic and cannot be justified, among other things because the lack of ventilation results in the same problems as in classrooms. Noise from outside enters through the vents, the ventilation itself is noisy, and lack of ventilation results in a considerable rise in indoor air pollution, as well as a thorough disruption of the bedroom biotope (humidity, temperature) – a problem that will become increasingly serious with global warming – as shown by numerous studies (Mishra et al., 2018, Xu et al., 2021, Basner et al., 2023).

1.7 Integrated indicators

There is considerable evidence for the adverse health effects of aircraft noise, but the indicators used to quantify the noise exposure underestimate both the impact of the noise and the number of people affected. To associate exposure with different types of health effects (annoyance, sleep disturbance, cardiovascular and cognitive effects, etc.), a set of integrated indicators should be introduced, allowing the collection of reliable data that should be made public. The most important indicator for assessing the impact of night and day flights is the frequency with which the maximum level reached by each flight exceeds 60dB(A) $L_{A,max}$ and the extent to which this threshold is exceeded. Yearly averaged acoustic levels (L_{den}, L_{night}, L_{Aeq}) are widely used in policy making and follow-up as well as in communication between stakeholders and residents. The working group insists on the fact that, from the point of view of the health impact of noise, the number of times a given event-related noise level is exceeded during a given time period is much more relevant than average acoustic energy levels. This means that, whilst a reduction in average noise levels (e.g. L_{den}) would be welcome, it could not be used as an excuse for increasing flight frequency. In fact, a decrease in L_{den} and/or L_{night} at the regional or at the community level may easily be accompanied by a worsening impact on health, because it allows for more frequent flyovers e.g. when a few noisy aircraft are replaced by many more less noisy aircraft. As truly silent aircraft are not a realistic option in the near future, a high frequency of flyovers leads to a worst case scenario for sleep disturbance.

1.8 Cooperation among Regions

Given the location of Brussels Airport, the different Regions should cooperate and agree on a common set of indicators, a common set of thresholds for health protection and the enforcement thereof. Indeed, these indicators form the basis for a peaceful, constructive, and collaborative debate among stakeholders and would provide a systematic tool to measure progress and perform further risk assessments.

1.9 Updating thresholds

Currently, the noise exposure levels that are considered thresholds for adverse health effects by the WHO (45 dB(A) L_{den}, 40 dB(A) L_{night}) are below the levels used for reporting and risk assessment. Therefore, the SHC advises updating the regional thresholds to the WHO levels for reporting and risk management and, at the same time, using the most recent dose-effect relations, while also complying with European legislation (Commission Directive (EU)2020/367, amending Annex III to Directive 2002/49/EC).

2 Reducing air pollution

The SHC recommends that measures should be taken to **reduce exposure to UFP** in residential areas near the runways. UFP concentrations should be monitored more continuously in both the Flemish Region and Brussels Capital. Besides UFP, other emissions and/or fractions should be studied as well (e.g., $PM_{2.5}$, PAHs, VOCs, OPEs, NOx). Permanent monitoring of these emissions should be implemented at Brussels Airport. The existing data on UFP show that residents living close to the runways and further along the north-east axis (gradually decreasing) are significantly exposed. *Airparif*, situated in the Paris region in France, closely monitors air quality around airports such as Charles de Gaulle and Orly, employing specialised monitoring stations strategically placed to capture emissions from aircraft take-offs and landings, as well as ground-based activities. Through sophisticated air quality modelling techniques, *Airparif* assesses the dispersion of pollutants emitted by airport operations, providing accurate insights into the concentration levels of pollutants such as nitrogen oxides (NO_x), volatile organic compounds (VOCs), and particulate matter (PM). Through clear and accessible communication channels, including websites, public reports, and community outreach initiatives, *AirParif* disseminate information about air quality monitoring results, pollution sources, and potential health impacts to a diverse audience. The Belgian Interregional Environment Agency (IRCEL-CELINE) provides an online platform for consulting air quality parameters in Belgium, but there is no specific communication about Brussels Airport. Consequently, such an initiative would be welcome near Brussels Airport.

It is important that in the early morning and evening, when the air is most stable, **emissions should definitely not increase any further** because the measured peak levels of UFP are already of concern in Diegem and Steenokkerzeel.

3 Improving scientific knowledge

The existing data considering the frequency of flyovers exceeding a given threshold should be analysed, mapped (geographically) and a study should be initiated to link these data with the available health information (e.g., use of medicines against hypertension and depression, incidence of stroke, myocardial infarction, heart failure, regional mortality). However, conventional epidemiological studies lack the necessary sensitivity and distinctive capability to measure the impact of one environmental factor (the airport-associated activities) on the incidence of chronic diseases in a complex environmental situation such as the region around Brussels. A molecular-epidemiological approach may be more precise and provide a much clearer insight into the extent to which nearby residents of the airport suffer from increased internal exposures and associated health effects in relation to airport operations. Longitudinal and molecular epidemiological studies will provide more accurate information on the potential health impacts of reducing aircraft noise and air pollution around Brussels Airport. It is important to include influencing factors other than noise in these studies such as socio-economical and psychological factors. Moreover, it is known that socio-economic inequity in noise distribution exists and should be documented as well.

- It would be interesting to use available cancer registry data and to set up an epidemiological study to determine whether the incidence of cancer (including brain cancers, although the population size is possibly too small) is higher in the vicinity of the airport than in the rest of the country and whether cancer incidence is associated with higher levels of aircraft noise and air pollution in the region.

- Sleep research is mainly performed with obtrusive equipment. Recently, less invasive protocols have enabled the measurement of short-term biological effects on sleep in real-life conditions. Larger studies merging real-life noise exposure and short-term biological responses are both feasible and affordable, which will increase knowledge significantly. Several studies are ongoing in the US and the UK, but results are not available yet. Similar studies can be deployed in Belgium.

- Similar to the RIVM approach, the SHC recommends investigating the effects of short-term exposure to UFP on lung function, and UFP and noise on inflammatory parameters in children and adults.

- **However, given the existing conclusive evidence for the adverse health effects of aircraft noise and emissions, the implementation of measures should not be delayed while new scientific studies are conducted.**

4 Communication

The SHC recommends investing in consultation and improving residents' trust in the authorities and the airport management. **Effective communication is of the utmost importance**. The measures taken should include **transparency** of the decision process, the implementation of **fair procedures** in which all stakeholders are represented, as well as **openness about the distribution of costs and benefits**. During the process of decision-making, residents should, in a timely way, receive truthful, exhaustive, and clear information on the scope, duration, and levels of the noise and pollutants around Brussels Airport.

5 Towards a greening of air transport:

As demonstrated in the chapters above, noise and air pollution caused by aviation have a significant direct and indirect impact on both human and environmental health. Besides, CO_2 and non-CO_2 greenhouse gasses emitted contribute to climate change. While all these aspects should be considered, a study by Boussauw & Vanoutrive (2019) on these issues near Brussels Airport concluded the following:

1. The noise impact of aviation is recognised and mainly described in an institutionalised format;
2. The impact of aviation on local air quality is ignored;
3. The communication on climate impact shows little correspondence or concern with the actual effects.

Between 1960 and 2018, global CO_2 emissions increased from 6.8 to 1034 Tg CO_2/yr (Lee et al., 2021). Contrail cirrus clouds yield the largest positive net warming effective radiative forcing, followed by CO_2 and NO_x emissions. In 2011, the contribution of global aviation was calculated to be 3.5 % of the net anthropogenic effective radiative forcing (Lee et al., 2021). While popular debate often points to short-haul flights (< 500 km) as the "culprit", they are not the main source of greenhouse gas emissions. With 27.9 % of flights, they represent only 5.9 % of fuel burned, while 6.2 % flights longer than 4 000 km account for a total of 47 % of fuel burned (Dobruszkes et al., 2022). These authors therefore concluded that policy initiatives that target longer flights are urgently needed.

Besides local measures to reduce noise, air pollution and emissions of greenhouse gasses, reduction strategies should also be considered in an international framework to achieve a more sustainable transport and mobility strategy. It is an important strategic mission for the future: too often, sustainability remains a word added to a policy document, accompanied by some marginal changes, which is enough to allow policymakers to claim that sustainability has been considered (McManners, 2016).

The SHC expert group has highlighted the important negative effects of air transport on public health. Improvements can be obtained by following the various policy recommendations contained in this report, but these potential improvements will be voided if air traffic continues to grow. Before the COVID-19 crisis, global air transport demand was estimated to triple between 2020 and 2050 (Gössling & Humpe, 2020). In its last EIA (Environmental Impact Assessment, *Milieueffectenrapport*), Brussels Airport indeed foresees increasing the total amount of passengers from ca. 26 million before COVID-19 to ca. 32 million in 2032. It envisages that this growth is possible with a similar number of aircraft movements and the current legal limit of 16 000-night flights. This SHC report considers that the tolerable capacity of the highly urbanised environment around Brussels Airport has already been exceeded for air pollutants and (night-time) noise. Therefore, the most significant reduction in the health impact from air transport will indeed come from a global reduction in air traffic. As a society, we should reflect on our (recent) dependency on immediate goods delivery processes and on the value we place on frequently flying to near or far destinations for business or leisure. The greening of air transport will essentially depend on our collective ability to reduce air traffic.

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VI SAMENSTELLING VAN DE WERKGROEP

De samenstelling van het Bureau en het College alsook de lijst met de bij KB benoemde experten is beschikbaar op de website van de HGR: wie zijn we?.

Al de experten hebben ***op persoonlijke titel*** aan de werkgroep deelgenomen. Hun algemene belangenverklaringen alsook die van de leden van het Bureau en het College kunnen worden geraadpleegd op de website van de HGR (belangenconflicten).

De volgende experten hebben hun medewerking en goedkeuring verleend bij het opstellen van het advies. Het voorzitterschap werd waargenomen door **Greet SCHOETERS** en **Jean-Louis MIGEOT** en het wetenschappelijk secretariaat door Stijn EVERAERT, Evelyn HANTSON en Stijn BOODTS.

ADANG Dirk	Menselijke gezondheid, elektromagnetische straling & milieugezondheid	UHasselt
BOULAND Cathérine	Milieugezondheid	ULB
COLE Pierre	Psychiatrie	Tivoli La Louvière
DOBRUSZKES Frédéric	Transportgeografie	ULB
FALZONE Claudia	Atmosferische vervuiling & milieumetrologie	ULiège
HENS Luc	Menselijke ecologie	VUB
INT PANIS Luc	Milieueffecten van transport en mobiliteit	UHasselt
MIGEOT Jean-Louis	Akoestiek	ULB
ROMAIN Anne-Claude	Atmosferische vervuiling & milieumetrologie	ULiège
SCHOETERS Greet	Milieugezondheid & toxicologie	UAntwerpen
VAN LAREBEKE Nic	Milieugezondheid & cancerologie	UGent, VUB
VERBRAECKEN Johan	Pneumologie & somnologie	UZA
KOLH Philippe	Cardiologie	ULiège
MASQUELIER Etienne	Fysische geneeskunde & pijnbestrijding	UCL

De volgende experten werden gehoord maar waren niet betrokken bij de goedkeuring van het advies.

DEGGOUJ Naïma	Otorinolaryngologie	UCL
DEKONINCK Luc	Akoestiek	UGent
GLORIEUX Christ	Akoestiek	KULeuven

De volgende instituten/verenigingen/etc. werden gehoord:

LEFEBVRE Wouter	Onderzoeker	VITO
PETERS Jan	Onderzoeker	VITO
SCHRECKENBERG Dirk	Onderzoeker en directeur	ZEUS
JANSSEN Nicole	Senior onderzoeker	RIVM
GABOULEAUD Philippe	Secretaris-Generaal	ACNUSA
BRION DUCOUX Agnès	Voormalig lid van het bestuur	ASNUSA

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