AIR QUALITY EXPERT GROUP

Indoor Air Quality

Prepared for:

Department for Environment, Food and Rural Affairs;
Scottish Government; Welsh Government;
and Department of Agriculture, Environment and Rural Affairs in Northern Ireland

This is a report from the Air Quality Expert Group to the Department for Environment, Food and Rural Affairs; Scottish Government; Welsh Government; and Department of Agriculture, Environment and Rural Affairs in Northern Ireland, on indoor air quality in the UK. The information contained within this report represents a review of the understanding and evidence available at the time of writing.

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Executive Summary

People spend a substantial fraction of their lives indoors (often 80-90%) and so these locations can represent a significant fraction of exposure to air pollution. Indoor air quality is a complex phenomenon but has been studied far less than air quality outdoors. In the absence of indoor sources of pollution, indoor air quality is determined by ingress of outdoor air, balanced with pollutant loss processes such as deposition to surfaces and through ventilation. In reality most enclosed spaces have a wide range of indoor emissions including from buildings materials, furnishings, the use of combustion appliances such as gas and solid fuel cookers, boilers and stoves, the consumption of solvent-containing products, and the use of consumer products (e.g. cleaning and personal care products). Individuals themselves are a source of emissions that include CO_2, human bio-effluents and biological aerosols such as viruses. Some factors are outside an occupant's control, such as building fabric or ventilation in public spaces or the workplace, however individual behaviour and activities are a significant determinant of indoor air pollutant concentrations. Consequently, a person's actions can directly influence the concentrations they experience. This contrasts with outdoors where concentrations are to a large degree controlled through the aggregation of collective societal emissions. Since dispersion is much more limited indoors, compared to outdoors, even modest emissions indoors can result in high indoor concentrations.

There is extensive qualitative information on how individual processes, materials and activities can lead to emissions indoors, including the detailed chemical speciation of the pollutants released. Many of the key outdoor pollutants are found to be important indoors, such as particulate matter ($PM_{2.5}$), nitrogen oxides (NO_x), and carbon monoxide (CO), although there is limited evidence of whether the toxicity of PM indoors differs from outdoors. There are aspects of pollution found indoors that are notably different to outdoors. Mould and damp can lead to elevated concentrations of biological aerosols when compared to those found typically outdoors. The indoor environment can accumulate much higher concentrations of volatile organic compounds (VOCs) than are found outdoors in the UK, due to their release from construction and furnishing materials and use of cleaning and personal care products.

Whilst comprehensive inventories exist that list the myriad different chemicals that are emitted indoors, AQEG found only limited information that places those emissions on a quantitative footing, e.g., expressing emissions of a pollutant as mass per unit of activity, person, or consumption. In this report, the National Atmospheric Emissions Inventory (NAEI) has been a key source of information on indoor emissions in the UK. Although the NAEI is not designed specifically for the purposes of evaluating indoor air quality, sources of pollution arising from buildings are significant to outdoor air quality and their emissions at a national scale are captured and reported as part of transboundary emissions obligations. Particularly notable are emissions of VOCs, of which >14% occur indoors according to the NAEI (for contrast only 0.1% of NOx and 0.7% of $PM_{2.5}$ emissions occur indoors). Major sources include aerosol propellants and decorating products such as paints and varnishes.

A complex mix of ventilation and product emission regulations and guidelines have an impact on indoor air quality, but these are not always well-integrated with one another or used to their best effect. Standards for acceptable ventilation rates are included in Buildings

Regulations in the UK; high VOC content products such as paints have been regulated through EU Directives, and numerous labelling schemes exist for construction products across Europe, but not in the UK. Less well defined are standards for acceptable concentrations of air pollutant indoors. Advisory health-based guideline values on selected indoor air pollutants issued by WHO and UKHSA (formerly PHE) do not have any statutory underpinning. In the workplace there are limits on occupational exposure to a range of airborne chemicals. These assume that the time spent in these settings is limited and those exposed are healthy adults, so they are set at high time-weighted concentrations. Occupational indoor air quality standards are likely not appropriate for a wider population that includes children, elderly and vulnerable individuals.

A major area of uncertainty identified relates to current concentrations of indoor air pollution in UK homes and their trends over time. Most AQEG reports on outdoor air quality can draw on extensive observational data collected through national, local authority and research networks, on many different pollutants, and often over multi-decadal periods. No such datasets exists for indoor air quality in the UK. Instead, the only quantitative evidence on indoor air quality comes from individual research studies in specific indoor micro-environments (e.g., homes, schools, transport, rail stations, shops etc.) with fragmented and inconsistent pollutant speciation. Most research studies report information for only a small number of pollutants over a short period of sampling, providing only a snapshot of concentrations and with limited data on occupant activities.

Since it is impossible to measure everywhere at once, outdoor air quality management assumes that given suitable criteria, representative assessments of concentrations can be made from a limited number of representative monitoring locations such as roadside, urban background and rural. It is however challenging to characterise a ‘representative’ indoor space that can be used as a reference point or a baseline against which other locations can be compared. A consequence is that it is impossible to generate a holistic and quantitative picture of current concentrations in UK buildings, or how this may have changed over time. It is also challenging to use measurements to evaluate those processes that determine indoor air quality, or to draw general or widely applicable conclusions on the effectiveness of interventions. Compared to outdoors, conducting detailed observations in homes is practically difficult and resource intensive, and each experiment runs the risk of being unrepresentative of other indoor locations. What emerges from research measurements of indoor air is the exceptional heterogeneity of chemicals found, and with a far greater range of concentrations than are encountered in typical ambient outdoor air in the UK.

There are currently rather limited capabilities to model and predict indoor concentrations (or personal exposure). For outdoor air, there is comprehensive model infrastructure to estimate concentrations of pollution at any given point in space or time, through combining emissions data, chemical mechanisms and meteorological fields. Outdoor models are routinely tested against observations to evaluate their performance, and in some cases, observations are used to improve model forecasts. The indoor environment lacks this same degree of predictive capability, in large part because of the uncertainty in potential contributing emission sources. This compromises attempts to estimate exposure and health effects, or the use of models to evaluate potential interventions. There are detailed chemical mechanisms that describe indoor gas and particle-phase reactions, developed as extensions of schemes used for outdoor models. However, these models frequently lack building and occupant-specific emission rates into indoor spaces, or parameterisation of ventilation, temperature, relative humidity, lighting, and air exchange of individual buildings. The key role played by occupant behaviours in controlling factors such as ventilation and frequency of use of consumer products that emit, means that identical homes can often experience widely differing levels of indoor air quality.

Determining whether indoor or outdoor air quality is the greater contributor to overall exposure is not straightforward. For any individual, it will depend uniquely on time spent in each environment, their home, its location, ventilation, choices of activity indoors and crucially which pollutant is being considered. The home is also not the only indoor environment that people experience. Elevated concentrations of pollution have been reported in studies of air quality in UK schools and hospitals, of particular significance given they are occupied by more vulnerable groups. Transport micro-environments are also a significant route of exposure, inside cars, buses and trains and transport hub buildings. Looking across the literature, peak indoor reported concentrations of $PM_{2.5}$ can often be higher than those that are experienced outdoors. For NO_2 the picture is mixed; outdoors at the roadside concentrations are often higher than are typically reported indoors, except when there is unextracted gas cooking. For biological aerosols, carbon monoxide and many VOCs, literature reported indoor concentrations in the UK are often significantly higher than outdoors.

There are numerous interventions that would likely improve indoor air quality including eliminating emissions from highly polluting sources such as solid fuel burners, improving building quality, and the development of lower emission product standards with accompanying labelling. Reducing emissions from these sources would also benefit outdoor air quality as well.

Anticipated improvements in outdoor air quality, as set out in the Clean Air Strategy, should also feed through into better indoor air quality, since air exchange will remain a key factor in determining indoor concentrations. However, it should be noted that in some urban and road-side locations ozone concentrations are likely to increase and that if brought in to buildings nearby could increase rates of indoor air chemistry. There are direct opportunities to further improve indoor quality through increased ventilation in buildings (including homes, commercial and public spaces), an issue which has increased significantly in public prominence during the COVID-19 pandemic. The enclosed nature of indoor spaces makes them amenable to air quality improvement through active air filtration systems for particulate matter, although these may incur tradeoffs that include capital / operational costs and long-term changes in exposure to bioaerosols that may have uncertain impacts on health. Caution is noted regarding other air cleaning technologies such as those using UV light, ozone, peroxyl radicals or ionizing reactions, which have the potential to be detrimental to indoor air quality through the creation of harmful secondary pollutants.

Evidence gaps and recommendations

Emissions and sources.

1. The myriad of different sources of pollution that exist indoors makes estimating emissions a complex task. Whilst there are comprehensive chemical and biological descriptions of the *types* of pollutants released from different sources and activities, estimates of absolute *amounts* are often inconsistent or not known. The size of indoor emissions from building and construction materials, use of cleaning and personal care products, cooking, solid fuel burning and so on are very variable and depend on factors such as materials used, type of appliance and crucially, occupant behaviour. Significant additional efforts are required to place indoor emissions on a more quantitative footing, including the development of activity-based emissions inventories that could be used to underpin modelling of indoor concentrations and the effectiveness of interventions.
2. Twelve emission sources within the NAEI, including use of cosmetics, toiletries, household products, paints, gas cookers and cigarette smoking are identified as occurring at least partially in indoor settings. However further activity surveys are required to confirm the fraction of emissions occurring indoors and between different indoor environments. Many key sources of emissions indoors are not included in the NAEI including from the fabric of buildings and furnishings, cooking processes, and sources of airborne biological matter. There are no methodological approaches, emission factors or activity data to enable detailed emission inventories to be developed for such indoor sources. Moreover health-based exposure limits have not been established for some pollutants found indoors, including biological aerosols. Further measurements and modelling of emission factors, and the parameters affecting them, are needed for a range of key processes occurring indoors and that affect indoor air quality.
3. Based on the current NAEI speciation approximately 50% of the VOCs emitted indoors in residential buildings are as ethanol and *n* and *iso* butane, from use of cosmetic, toiletry and aerosols and some other household products. Emissions of VOCs in non-residential buildings are an area of further uncertainty. Research is needed to characterise both the speciation and absolute emission rates of VOCs from indoor sources and those activities releasing reactive chemical species such as terpenoids in fragranced products. These have the potential to generate secondary air pollutants indoors through reaction with ozone, some of which are harmful to health. However, these secondary species may only be present in small amounts and are not currently covered by the NAEI. More generally the transformation of indoor chemical emissions into potentially more harmful secondary pollutants is an area of uncertainty that could be addressed through investment in research into both laboratory studies and modelling.
4. There is a complex landscape of indoor air guidelines and source-specific regulation for emissions, predominately focused on VOCs, semi-volatiles and fire retardants from buildings and related construction products (including wood, glues, carpets, and

furniture). The current regulatory framework is fragmented and is designed around product sectors, and there is no holistic management of sources of emissions indoors, or consideration of how they behave in a realistic environment rather than in testing chambers. Paints and varnishes are one of the few indoor sources of air pollution where there is a widely recognisable UK product labelling scheme to inform consumers of possible indoor impacts (and mitigating actions). Most other VOC-emitting products carry no labelling and there is very limited public recognition of such products as a source. Wider use of product labelling in sectors such as buildings materials, furnishings, and consumer products may be effective in driving innovation and reduction in VOC content. It may also raise awareness with the public and allow for discrimination between low and high emitting products. The use of improved product labelling is cited in the Clean Air Strategy (2019) as a possible means to reduce overall national emissions of VOCs to meet international emission ceiling obligations.

Distribution, trends and effects of indoor air pollution

5. Very limited evidence exists for historical trends in indoor air pollution, and there are very few UK-specific measurements with which to assess its significance relative to outdoor air pollution. The challenge is compounded by the enormous heterogeneity in the indoor environment and concentrations encountered. Several long-term societal trends are however likely to be driving the balance of exposure to air pollution towards indoors. Whilst outdoor air quality has improved significantly over the last 30 years, improved energy efficiency has led to increased airtightness levels and lower building ventilation rates, potentially deteriorating indoor air quality. Changes in lifestyle have led to more time spent in enclosed spaces (spread across transport, homes, workplace, schools and so on). There is an urgent need for more quantitative evidence on the balance of air pollution exposure from outdoors vs indoors, which at its heart will require a greater emphasis on measurements indoors (see recommendation 9).
6. Significant exposure to air pollution can be encountered in enclosed spaces during travel, however most studies on concentrations in travel modes were carried out some years ago and do not reflect recent large decreases in exhaust emissions arising from progressive tailpipe emissions standards. In many older studies, outdoor concentrations were higher than they are today, and the differences between being outdoors in traffic environments versus inside were also far greater. The conclusions that might be drawn on whether air pollution was higher or lower during walking or cycling versus travelling for example in a car, could well be different if these experiments were repeated today. There is an urgent need for more contemporaneous data that can inform the public on likely exposure to pollution using different travel modes.
7. There are likely to be further changes forced externally that may alter the future balance of exposure to air pollution and increase the significance of indoor environments. Climate change adaptation and mitigation policies and net zero technologies may lead to a range of impacts that could alter airtightness, temperature

and humidity indoors, which in turn may lead to changed behaviours around ventilation and building management. Indirect climate change effects such as increases in flooding, may degrade indoor air quality as a consequence of increased prevalence of moulds and damp, in turn increasing concentrations of biological aerosols. Changes in lifestyle and working patterns may also impact on indoor air quality (and more broadly the spread of respiratory pathogens), for example increasing time spent at home, or in shared co-working environments. It is critical that the possible impacts of future technological or behavioural changes on indoor air quality are evaluated routinely in policymaking and cross-government mechanisms developed further to consider these effects.

8. The effects of indoor air quality on health were not covered in this review, however little is known about the synergistic effects of environments that lead to simultaneous exposure to both chemical and biological materials. The evidence base most frequently reports individual pollutants and effects in isolation, and the relative impacts of each pollutant type are difficult to rank. For particulate matter, it is unclear whether particles generated indoors are more or less toxic than those generated outdoors, a critical uncertainty when ascertaining health impacts or mitigations like ventilation. Little is known about how emissions and subsequent exposure indoors to combinations of organic, inorganic and biological aerosols may influence issues such as asthma and paediatric admission. This would be an area where future joint working with the Committee on the Medical Effects of Air Pollutants (COMEAP) could be productive. As for outdoor air pollution, it is clear that concentrations of some air pollutants found in homes can be linked to socioeconomic co-factors, although the UK-specific data on this is sparse. Increasing the range of observations that are made may help establish the scale of these inequalities (see Recommendation 9)

The Measurement of Indoor Air Pollution

9. The indoor environment is highly heterogenous and cannot be monitored using the same research or regulatory strategies that are used outdoors. Throughout this review a lack of UK measurement data has hampered AQEG in drawing conclusions on a range of issues. There is an urgent need to establish a national baseline assessment of indoor air quality across the UK in both heating and non-heating seasons. This requires consideration of which properties are most important to measure and capture a representative range of homes/building stock. It should also account for socioeconomic diversity, and exposure in key shared spaces such as schools, hospitals or offices. It is vital that a set of high quality, calibrated data are available for assessment of pollutant distributions over representative environments, covering both organic and inorganic pollutants, and capturing variability over daily to multi-year timescales.

10. Instrumentation that is designed and calibrated for outdoor air pollution may not necessarily be optimised for indoor environments. There are differences that include confounders such as relative humidity, different pollutant mixtures (including high and very rapidly changing concentrations) that interfere with the measurement process or, in the case of particles, different chemical and physical properties. Whilst there

has been considerable progress in improving lower-cost compact instruments for indoor air quality assessment, further research is needed to ensure data quality from devices of this kind meet the necessary data quality standards. Nonetheless there are significant opportunities for affordable compact devices to play a role in providing real-time information on indoor air quality trends, for supporting epidemiological studies that account for health impacts of indoor pollution (for example using wearable sensors) and that enable public engagement in measurements. The use of CO_2 monitors in response to concerns around ventilation and spread of SAR-CoV-2 in aerosol form may help raise awareness of indoor air quality more generally.

11. New strategies are needed for research-led measurements that develop better process-level understanding of indoor air quality, and that can help quantify the effectiveness of technical or behavioural interventions. Studies of outdoor air pollution processes have somewhat fewer physical constraints than experiments conducted in small enclosed indoor environments. New research infrastructure, such as test homes, and appliance and emission testing facilities are needed that can support detailed quantification of effects. To accompany new physical infrastructure there should be parallel development of model capabilities that can represent indoor air quality processes. There is a need for support for multidisciplinary research that can draw researchers from across a range of social, behavioural, engineering, physical and bioscience disciplines. Investment is needed to assess not only the performance of new buildings and appliances, but crucially how retrofits and adaptations of existing infrastructure impact on indoor air quality.

12. Whilst this report considers primarily the sources and character of airborne pollutants indoors that may be relevant to human inhalation, it is important to recognise that within buildings other reservoirs exist, along with alternative pathways for exposure. Air pollution generated indoors may end up accumulating on surfaces or deposited to flooring for example. Dermal exposure is then possible, as is the ingestion of particulate matter, something that is potentially significant young children who may spend time on carpets and flooring.

Interventions and actions to improve indoor air quality

13. Many personal actions and behavioural changes can have a significant impact on improving air quality indoors (Chapter 7), although it is important to note that individual agency may be limited for some vulnerable individuals and certain socioeconomic and demographic groups. Differences will also exist between home-owners and tenants. The most effective individual actions to improve indoor air quality are often rather simple – increasing ventilation and reducing emissions at source. As has been raised in previous reviews, public knowledge of indoor pollution and its effects are relatively limited. There is clearly more that can be done in the sphere of communication of the science, health impacts and of potential mitigating actions, and there is likely a role for government departments and health agencies in that. It is impossible to ignore however that many of the sources of air pollution indoors are linked to activities that individuals and families may value and enjoy (cooking, cleaning, fires, candles, fragrance to name only a few); understanding risk

perceptions and motivations are important for developing effective policies and interventions.

14. Regulations and guidelines affecting indoor air quality derive from a wide range of sources. These can be poorly understood and not effectively implemented. A more consistent application of existing regulations and schemes is likely to benefit indoor air quality. There are also a range of further interventions where innovation from industry, and if necessary additional regulation, would be expected to lead to improvements in indoor air quality. The net zero transition includes decarbonisation of home heating and gas cooking, a climate intervention that would be beneficial also for indoor air quality. Further limiting the VOC and SVOC content in frequently used consumable products, or in the products and materials that make up the fabric or furnishing of a building would also deliver improvements in indoor air quality. Reducing emissions at source may be particularly impactful in new buildings or retrofitted houses working to the latest low energy requirements and with reduced ventilation rates.

15. Not all exposure to pollution indoors occurs in private homes. The management and regulation of air pollution in commercial and public spaces (where individuals have no control over the air they breathe) is complex but leans heavily on occupational rather than public health standards. Whilst recommendations from WHO and UKHSA (formerly PHE) for indoor air quality exist (at broadly similar concentrations to outdoors), they do not have the same enforceability. Instead, actionable limits on concentrations are determined by occupational exposures, and assume the individual exposed is a healthy worker. These standards may not necessarily protect children, the elderly or otherwise vulnerable, should they spend long periods in those buildings. Given the increasingly demanding targets being set in the Environment Act (2021) for outdoor air quality, the lack of similarly ambitious standards for public spaces appears a significant inconsistency. Evaluating the costs and benefits of enforceable indoor air quality standards appropriate to broader protection of public health in public places is recommended.

16. Delivering a net zero greenhouse gas budget in 2050 is likely to have significant impacts on the built environment. Although not evaluated in detail, energy efficiency improvements in buildings have the potential to reduce ventilation and potentially degrade indoor air quality if they are not implemented well. Engineering solutions that deliver both energy efficiency and good air quality are well-proven, however guidance and regulations should be strengthened to require a holistic consideration of both issues starting from the design stage. The decarbonisation of homes is likely to improve indoor air quality by removing substantial NO_x and $PM_{2.5}$ sources such as gas cooking, gas and solid fuel fires and gas boilers. AQEG would recommend that the possible changes in indoor concentrations, and consequential health benefits, from home decarbonisation are empirically evaluated since this may strengthen the economic case for investment.

17. As noted in previous recommendations, many of the challenges around indoor air quality stem from deficiencies in the evidence system and the lack of recognition of its potential importance outside expert communities. As an issue that has had no

obvious single owner in either government or the research funders this is perhaps unsurprising. It is noted that the Cross Government Working Level Group on Indoor Air Quality has been set up. It will be important that this group, or an equivalent, remains active and is effective at raising the profile of indoor air pollution across government. Numerous other recommendations will require this group to act to ensure that indoor air pollution is appropriately considered in future policy decisions and the prioritisation of research funding.

Contents

A short technical primer:

What are the key principles that control indoor air quality?

The concentration of an air pollutant indoors depends on several factors, principal amongst these are:

i) the magnitude and location of indoor emission sources that release the pollutant, and sinks that remove the pollutant,
ii) the indoor airflow and turbulence which disperses the pollutant,
iii) the rate of air exchange between outdoors and indoors,
iv) the concentration of that pollutant in the outdoor air that is brought inside, and
v) the dimensions of the indoor space.

Knowledge of these five factors along with equations governing the airflow, pollutant transport and chemical transformation of pollutants allows for a numerical estimation of the pollutant's concentration, including its spatial and temporal variation.

A simplified approach to describing indoor air quality for a specific pollutant is to use a box or zonal model. This can show the dependencies of a pollutant's concentration indoors to the factors described above. This widely used method assumes that the pollutant is well-mixed indoors, which is approximately the case when the indoor - outdoor air exchange rate, indoor emissions and outdoor concentrations are varying only slowly. Assuming that indoor removal (for example surface deposition, chemical reactions and indoor filtration) is proportional to the indoor pollutant concentration, then, from conservation of mass, an approximate steady state relationship for the indoor concentration (C_i) is obtained as:

$$C_i \approx \frac{(q_i + q_o)}{(\alpha_e + \alpha_s)}$$

Here q_i is the indoor source emission rate; q_o is the source strength of pollution entering the room from outdoors equal to $(1-f)\alpha_e C_0$ where C_0 is the external concentration at the outdoor air inlet(s) with filtration factor f ; α_e is the rate of exchange of outdoor air and α_s represents the rate of loss indoors so that the loss rate is $\alpha_s C_i$.

The above expression can be formulated in terms of the indoor concentration/outdoor concentration ratio (I/O) as:

$$\frac{C_i}{C_o} \approx \left(1 - f\right) \frac{1 + q_i/[(1 - f)\alpha_e C_o]}{1 + \alpha_s/\alpha_e}$$

In the **Appendix 1** we present a fuller derivation and give examples of the application of these simple formulae to some of the data presented in the main report. These examples illustrate the dependencies between indoor concentrations of VOCs, PM, NO_2 and CO_2 and indoor emission rates, outdoor concentrations, indoor-outdoor air exchange rates and indoor sinks.

Chapter 1 – Indoor sources of air pollution

The indoor environment is subject to myriad different emissions of both chemicals and biological material. Emissions to air occur from virtually all aspects of lives lived indoors. These include emissions from the fabric of buildings themselves, through to sporadic emissions from activities such as cooking, cleaning, the use of heating systems, fires and solid fuel stoves. Buildings and their occupants are also a rich source of bioaerosols, including respiratory viruses and bacteria, from microbes that grow on indoor materials and aeroallergens for example from animals kept indoors. A complete inventory of all the different types of emissions that occur indoors is however an almost impossible task to ever fully complete. In this section some of the major categories of emissions of relevance to indoor air quality are described along with some of the key pollutants. As with outdoor air, some pollutants are released from multiple different activities, for example volatile organic compounds (VOCs) are emitted indoors from decorative products, from furnishings and glues, as well as from hundreds of different household products and from microbes that dwell indoors. Similarly, particulate matter (PM) is released from multiple activities indoors, including combustion and cooking as well as having secondary sources via atmospheric oxidation.

1.1 Chemical emissions from the fabric of buildings

VOCs may be emitted over periods of months or years from a wide range of construction and finishing products that are used in a building. Examples include concrete and masonry surface treatments, timber preservation and coatings, adhesives, sealants, paints and coatings, damp-proofing emulsions and membranes, wall coverings, floor coverings and fungicide washes. In addition, the volatile organic compound (VOC) formaldehyde may be emitted over significant periods of time from resins, phenol-formaldehyde and urea formaldehyde (UF) from wood-based products such as particleboard in furniture, urea-formaldehyde based lacquers and foam cavity wall insulation. Examples of building sources of emissions are given in Table 1.1.

The extent and longevity of the emission of VOCs, including formaldehyde, from building products into indoor spaces are functions of the exposure to the air of the source and its strength, and for some sources the nature of their surface chemistry. The concentration of an emitted chemical in indoor air will depend on temperature and the amount to which the air is ventilated, and in some cases cleaned or treated. Irrespective of this, the most effective way to control such indoor air pollution is to control the source of the pollutant(s). The need for such source control is indicated in the Building Regulations Part F (see Section 2.1.6). For source control, please see Section 7.1.

1.2 Space heating using natural gas

In the UK domestic boilers primarily operate on natural gas (~ 93% methane plus some ethane and propane). The National Atmospheric Emissions Inventory (NAEI) estimates that around 95% of emissions from domestic gas consumption are from use of domestic boilers with the remaining 5% from use of gas fires and gas cookers. The principal pollutants of concern once natural gas is combusted in boilers are nitrogen oxides (NO_x) and carbon monoxide (CO), but the combustion of natural gas can also lead to high particle number concentrations. Space heating boilers are typically located within a house or garage and vent combustion gases outside. Given the proximity of the exhaust from domestic boilers to buildings, there is the potential for ingress of some of those exhaust gases back into the indoor environment, although the evidence for the importance of this route of exposure is limited. There are requirements within Building Regulation that address this issue and that provide guidance on the acceptable location of gas flue outlets.

The principal issue related to indoor exposure to pollutants from space heating is related to boiler malfunction which may lead to asphyxiation from exposure to extreme concentrations of CO; this leads to about 20 deaths in England and Wales each year (HSE, 2021). While such poisoning events are comparatively rare, there will likely be an influence of less extreme boiler malfunction which may result in prolonged and increased exposure to CO and other combustion gases. However, there is a lack of evidence for wider exposure to lower concentrations and therefore it is difficult to evaluate its importance in an air quality context.

1.3 Emissions from the use of consumer and domestic products

The use of building and construction materials as well as consumer products in the home is a significant source of volatile and semi-volatile organic compounds (VOCs) (e.g. Shrubsole *et al*., 2019) and also some inorganic emissions, such as hydrochloric acid (HCl) and ammonia (NH_3) from cleaning products (Rösch *et al.* 2014; Ampollini *et al*., 2019). The range of commonplace materials that release VOCs is exceptionally broad and includes personal care and cleaning products, inks, glues, insecticides, room fragrance, adhesives and many decorative products (Trantallidi *et al.* 2015).

Halios *et al*. (submitted 2021) reviewed the emissions, concentrations, and health effects of Volatile Organic Compounds (VOCs) in European residences. A summary of sources and health effects for the VOCs identified as frequently occurring and health-relevant are presented in Table 1.1. Many of the VOCs listed in Table 1.1 are released from multiple different indoor sources, for examples aldehydes and ketones can off-gas from buildings materials and be released from microbial sources. Microbes produce VOCs as by-products of their metabolism which can also lead to the release of alcohols, esters, terpenes, and heteroatom containing VOCs. Sometimes VOC emissions are short and sporadic, for example from the use of a cleaning product, sometimes persistent, from outgassing from materials.

Table 1.1: Sources of VOCs Released Indoors and potential health effects (Halios et al., submitted).

Volatile Organic Compound		Sources	Health effect
Aldehydes	Formaldehyde	Particleboard, MDF, plywood, chipboard, composite board, gypsum board, ceiling tiles, sound insulators, polyurethane adhesive mastic, vinyl and ingrain wallpaper, expanding foam, glue for wallpaper, sealing plaster, finishing plaster, wallpaper paste, latex and dispersion paint, machine wash liquids/detergents, paints and coating or adhesives, furniture and carpet, fragranced and unfragranced jarred candles, burning stick incense, shampoo, shower gel, body lotion, facial moisturiser, hair styling gel, deodorant, hair conditioner, typical wood stoves, kerosene space heaters, ethanol fireplaces. Used in adhesives and sealants, coating products, fillers, putties, plasters, modelling clay, inks and toners, polymers, fuels, biocides (e.g. disinfectants, pest control products), polishes and waxes, washing & cleaning products and cosmetics and personal care products.	Respiratory Carcinogenic Irritation
	Acetaldehyde	Concrete/screed with and without PVC covering, MDF, chipboard, plywood, composite board, expanding foam, finishing plaster, ceiling tiles, gypsum, plaster, vinyl and ingrain wallpaper, polyurethane adhesive mastic, wallpaper paste, latex and dispersion paints, fragranced and un fragranced jarred candles, burning stick incense, typical domestic wood stoves, kerosene space heaters, ethanol fireplaces	Respiratory Carcinogenic Irritation
Aromatic hydrocarbons	Benzene	Gypsum board, commercially available floor coverings (made of PVC or with polypropylene or polyamide fibres), carpet glue, scatter rugs, solvent-based cleaning and painting products (acrylic and water-based paints, matt emulsion), burning fragrance jarred candles and burning stick incense, kerosene space heaters, fireplaces with liquids, wood-burning fireplaces.	Cardiovascular Neurological, Carcinogenic Irritation
	o, m, p xylenes	Solvent-based cleaning and painting products, solvent-based and water-based interior coating, commercially available candles, machine wash liquids/detergents, paints and coating or adhesives. Used in lubricants and greases, polishes and waxes, adhesives and sealants, antifreeze products	Respiratory Cardiovascular Neurological Irritation

		and biocides (e.g. disinfectants, pest control products), and close systems (e.g. cooling liquids in refrigerators, oil-based electric heaters).	
	Ethylbenzene	Materials for floor coverings (PVC, Linoleum, Rubber, Poly- olefine), Gypsum board, polyurethane foam, polyurethane adhesive mastic, solvent-Based cleaning and painting products, solvent and water based interior coating, burning stick incense, candles.	Respiratory, Neurological Carcinogenic Irritation
	Toluene	Materials for floor coverings (PVC, Linoleum, Rubber, Polyolefin), gypsum board, polishes, nail polish, synthetic fragrances, paint, paint thinner, adhesives, anti-freeze products, non-metal surface treatment products, inks and toners, biocides (e.g. disinfectants, pest control products), textile treatment products and dyes, leather treatment products, machine wash liquids/detergents, burning stick incense and candles. Used in closed systems like cooling liquids in refrigerators, oil-based electric heaters, carpets, general furnishing.	Respiratory, Cardiovascular Neurological Irritation
	Styrene	Wooden flooring, materials for floor coverings (PVC, linoleum, rubber, polyolefin), polyurethane foam and adhesive mastic, rubber and epoxy adhesives, medium density board, carpet (Nylon and polypropylene w SBR adhesive), polystyrene foam, solvent-based cleaning and painting products, solvent and water - based interior coating, machine wash liquids/detergents, burning fragranced and unfragranced paraffin wax jar candles, burning incense stick, paints and coating or adhesives. Used in fillers, putties, plasters, modelling clay and coating products.	Respiratory, Neurological Carcinogenic Irritation
	Trimethylbenzene [1,2,4- and 1,3,5-Trimethylbenzene	Materials for flooring coverings (PVC, linoleum, rubber, polyolefin)	Respiratory, Neurological, Irritation
Ketones	Acetone	Solid wood (pine, oak, beech), plywood, composite board (MDF, chipboard, OSB), fireboards from coriander biorefinery, glue for wallpaper, finishing plaster, linoleum, silicone, expanding foam, ceiling tile, gypsum board, veneered particle board (UV curing lacquer), surface sprays, glues, burning stick incense, electric air fresheners, kerosene space heaters, ethanol fireplaces, cleaning agent, cosmetics, flea sprays	Respiratory, Cardiovascular Neurological, Irritation

Terpenes	α-pinene	MDF, chipboard, veneered particle boards, adhesives for flooring installation, nylon carpets, PVC, solvent-based interior coatings, passive diffusers, burning wood-sticks, automatic sprays, electric air fresheners, perfumes, cleaning products and deodorants.	Irritation
	Limonene	MDF, chipboard, adhesive for flooring installation, veneered particle boards, paints, multipurpose coating products, solvent and water-based interior coatings (polishes and waxes), biocides (e.g. disinfectants, pest control products), shampoos, shower gels, moisturizers, conditioners, passive diffuser, electric evaporator, burning wood-stick, automatic spray, cleaning agents.	Irritation
Polycyclic aromatic hydrocarbons	Naphthalene	Materials for floor coverings (PVC, Linoleum, Rubber, Polyolefin), Insecticide or pest repellent, fragrance jarred candles, wax candles, anti-mosquito incense sticks, solvent-based cleaning and painting products.	Respiratory Neurological Carcinogenic
Chlorinated hydrocarbons	Tetrachloro-ethylene	Paint removers, cleaners, glues, suede protectors	Respiratory Neurological Carcinogenic
	Trichloroethylene	Metal cleaning and degreasing agents, dry cleaning, breakdown product from Tetrachloroethylene	Respiratory, Cardiovascular Neurological Carcinogenic irritation

Emissions from consumer products are dependent in large part on consumption and usage patterns, and this varies very widely indoors (Wu *et al.* 2010). As a result there is considerable variation (orders of magnitude) in indoor concentrations of VOCs between homes even of identical construction. An example in UK homes is given in Wang *et al. (*2017) where limonene in indoor air varied over three orders of magnitude between identical modern brick-built homes. Estimating indoor emissions of VOCs depends on knowledge of the range of product consumption patterns and these are relatively poorly documented in research literature (although these data do exist commercially, for example held by manufacturers and supermarkets). Some VOC-emitting products are used in most UK homes on most days, whilst others see only very infrequent use. Publicly available use patterns can be found in databases used for modelling human exposure to chemicals, however there is relatively little data available specifically for the UK. One of the largest publicly available exposure factor databases collated from various studies is the US EPA's exposure factor database chapter for consumer products, but there may be differences in use patterns between the US and UK (US EPA, 2015).

An example of the variability in VOC product usage and resulting indoor VOC emissions is taken from Heeley-Hill *et al.* (2020). This studied VOC indoor concentrations in parallel to collecting statistics on UK household behaviours. Figure 1.1 shows the cumulative frequency of use of various VOC-containing products over 3-day survey periods and the resulting concentrations of VOCs observed in those 60 UK homes. Overall, the most abundant VOC found indoors in the UK in this study was *n*-butane, which is the main VOC used as an aerosol propellant (used for example in antiperspirant deodorants, cleaning sprays, insecticides and room fragrance). The most frequently used products were deodorants, perfumes and cleaning sprays. In Heeley-Hill *et al.*, median indoor concentrations of *n*-butane were around 100 times greater than measured in matching outdoor air samples.

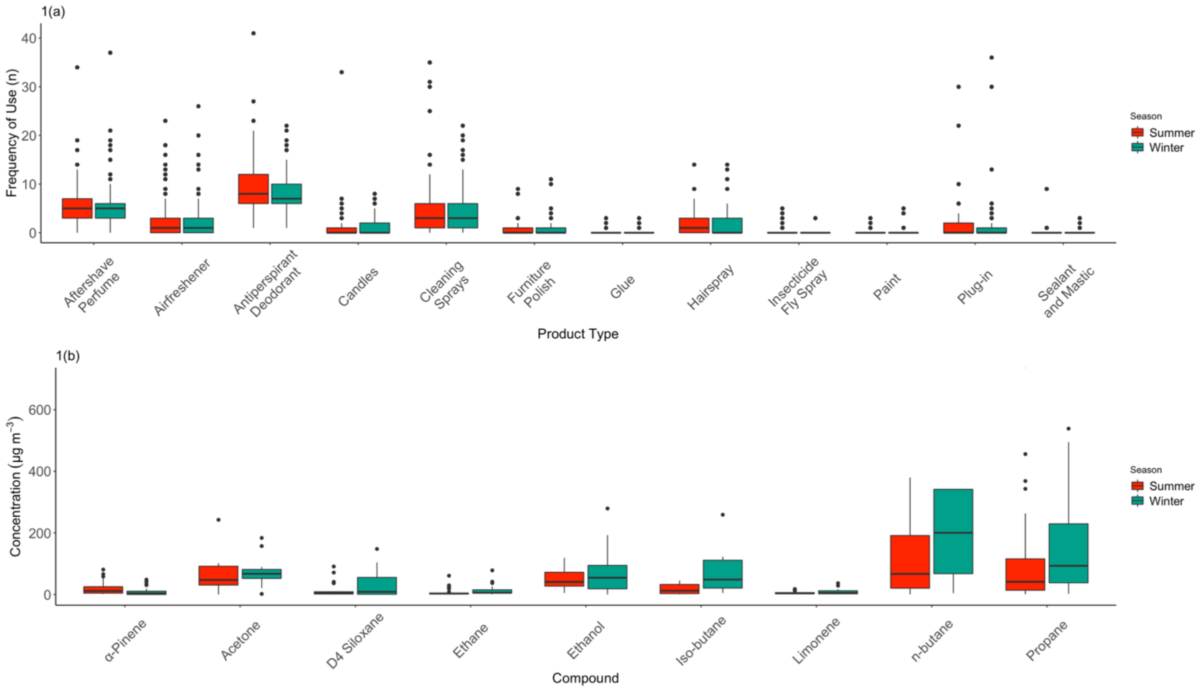

Figure 1.1: (a) Frequency of use of consumer products per three-day sampling period in 60 UK homes (b) concentration ranges of selected VOCs from 60 homes by season (red is summer, green winter). Box size is defined by the 25th and 75th percentiles, with the middle line of the boxes the median value. No greater than 1.5 times the interquartile range from both percentiles defines the whiskers. Outliers are plotted as individual data points beyond the whiskers. To aid visualisation in (b), outliers beyond the 10th and 90th percentiles on n-butane are not included in the plot but are included in calculations used to define box plot parameters.

Of note indoors is the release of monoterpene species from consumer products; these are VOCs included in many commonplace items to add fragrance. The relatively high chemical reactivity of species such as limonene and alpha-pinene with ozone, nitrate and hydroxyl radicals (NO_3 and OH), coupled to their ability to form both formaldehyde and secondary aerosols as oxidation by-products, makes them important potential precursor emissions to other indoor pollutants (Abbatt and Wang, 2020; Carslaw, 2007). This is further discussed in Sections 5.1 and 5.2.

Of relevance to managing and improving indoor air quality, it is notable that this particular sector of emissions is not declining in high-income countries; indeed there is evidence that home use (and indoor release) of volatile chemicals is on an upwards trend (McDonald *et al*., 2018). The global emissions of VOCs from consumer aerosol products alone are currently estimated to be ~ 1.3 Tg yr^{-1}, and projected to rise to 2.2 Tg yr-^{1} by the mid-2040s (Yeoman and Lewis, 2021). Estimated UK emissions from a range of aerosolised personal care products are shown in Figure 1.2 with more than 90% of these emissions estimated to be released inside the home. The consistency between the emissions in Figure 1.2 and measured concentrations in Figure 1.1 is discussed in Appendix 1.

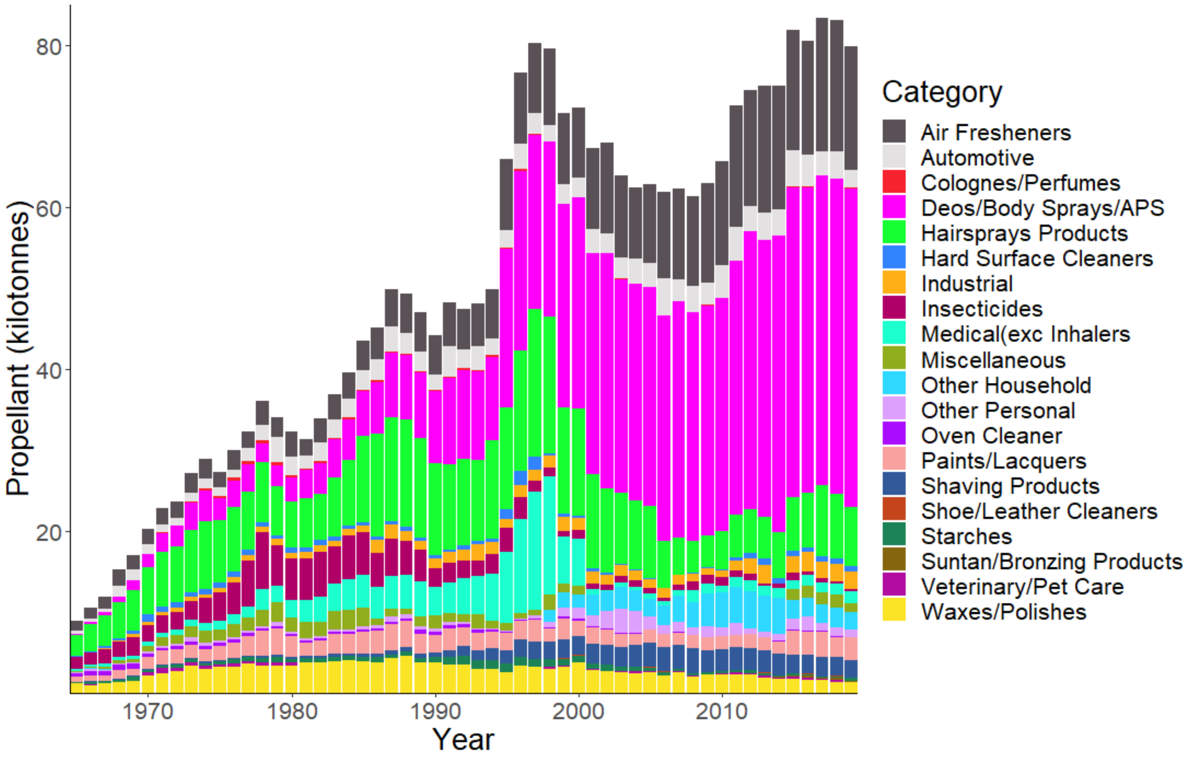

Figure 1.2: Estimated United Kingdom VOC emissions from aerosol-based consumer products in kilotonnes for all propellant types (2018 data). Emissions estimated using industry fill reporting statistics (British Aerosol Manufacturers' Association (BAMA), 2019. Annual Report and Accounts 2018-2019) and corrected for mass emissions specific to individual products based on fill estimates (volume, pressure) (DG Enterprise and Industry, 2014. Impact Assessment Study on the Adaptation to Technical Progress of the Aerosol Dispensers Directive).

1.4 Emissions indoors from cooking in homes

Cooking is widely regarded as one of the more important sources of indoor air pollutants. There have been many global studies that have attributed adverse health outcomes to cooking as an activity, however it should be noted that many of these focused on the developing world, in particular where solid or liquid fuel combustion is used as a heat source in poorly ventilated domestic environments (Lee *et al*., 2020). This is less of an issue in the UK where the main sources of heat for cooking are electricity or natural gas. Gas appliances are significantly cleaner than solid or liquid fuel stoves, however these are still recognised as a source of indoor pollutants such as CO, NO_2 and ultrafine particles (Wallace *et al.*, 2008; Mullen *et al.*, 2016). Irrespective of the source of heat, the cooking itself is a major source of pollutants in its own right (Abdullahi et al., 2013; Farmer et al., 2019).

In general, there are four processes that take place during cooking that can cause air pollutants to be emitted. These are:

1. Products of chemical reactions and thermal breakdown within the food during cooking. Processes responsible can include pyrolysis, hydrolysis and oxidation, possibly involving free radicals and autooxidation. A major example is the breakdown of triglycerides in oils to free fatty acids and other compounds.

2. Volatilisation of compounds, either those already present in the food or the aforementioned chemical breakdown products. These may exist as gases or can condense to form particles as the vapour plume cools.

3. Mechanically ejected material. These can be caused by bubbles bursting on the surface of the food or a body of liquid such as oil or water, or by splashing.

4. Combustion during the burning of the food itself or ejected material. This can be produced by some cooking styles such as flambé, but also unintentionally if food is overcooked or if material (e.g. oil) is ejected onto the heat source.

The material produced is generally carbonaceous in nature but other products can be produced from combustion such as NO_x. The carbonaceous matter can exist in both the gas and particle phases (Abdullahi *et al*., 2013). The organic matter can include polar species produced through thermal breakdown such as glycerol, fatty acids and mono and diglycerides (Nolte *et al.,* 1999). The breakdown processes can also produce some highly toxic compounds such as aromatics (including polycyclic aromatic hydrocarbons (PAHs)), aldehydes, furans and amides (Kubow, 1992; Abdullahi *et al*., 2013). Vapours produced can include molecules associated with aromas, such as terpenes (Klein et al., 2016). Some compounds can be uniquely associated with certain types of cooking, such as cholesterol in mechanically ejected meat cooking emissions.

Particulate matter from cooking consists mainly of organic matter but can include elemental carbon from charring or combustion. Organic particles formed from volatilisation are often semi-volatile in nature and can evaporate when dispersed into the ambient air (Reyes-Villegas *et al.,* 2018). These particles can be very small, in the form of ultrafine particulate (smaller than 100 nm in size), which contribute significantly to particle number concentrations. The organic material produced is chemically reactive and has been shown to be a potential source of secondary organic aerosol, but this is likely of more relevance to outdoor air quality once ventilated out of the building (Zhou *et al*., 2021).

The nature and the amounts of indoor pollutants produced vary strongly according to cooking style. Note that definitions of cooking styles vary, for instance what is known as ‘grilling’ in the British vernacular is normally referred to as ‘broiling’ in the US scientific literature. The volatilisation route is particularly prominent when hot oils are present and when the food is regularly agitated, so shallow frying and, to a greater extent, stir frying are known to be particularly large sources in this regard (Zhao and Zhao, 2018; Amouei Torkmahalleh *et al*., 2017). The temperature of cooking is known to have a major effect, especially if an oil reaches its ‘smoke point’, where a large amount of particulate matter is released, containing toxic breakdown products (Choe and Min, 2007). Mechanical ejection is favoured when food ‘sizzles’, so this occurs during frying or grilling. Conversely, boiling and steaming are regarded as relatively weak sources of air pollution. While thermal sources are present during baking and roasting, the emissions are contained within an oven and only sporadically released. Note that the residue from cooking emissions can also be a source of emissions, particularly if heated, so splashed material on locations such as hobs, internal oven surfaces and grill pans should also be considered as a source.

1.5 Emissions from the use of solid fuels

Domestic solid fuel combustion primarily relates to the burning of coal and wood (logs, chips and pellets) in stoves, boilers, cookers and open fires. Use of solid fuels in the UK is often for aesthetic reasons rather than as a primary heat source. Consequently, there is the opportunity for interventions through increasing public awareness of the impact of burning solid fuels on indoor air quality and on health. It has been proposed that increases in electricity and natural gas costs in late 2021 and early 2022 may possibly lead to increased use of solid fuels for space heating for purely economic reasons, however at time of writing of this report there is no quantitative information to support this proposition.

Emissions indoors from use of solid fuels include particulate matter (PM) and gases including carbon monoxide (CO), carbon dioxide (CO_2), oxides of nitrogen (NO_x), volatile organic compounds (VOCs), sulphur oxides (SO_x) and a range of trace species including polycyclic aromatic hydrocarbons (PAHs). A comprehensive review of combustion processes and the main formation routes for combustion products is given in Williams *et al.* (2012). AQEG has previously provided an assessment of the potential air quality impacts from biomass combustion (AQEG, 2017) and, although that report focusses on outdoor air quality, it provides more details on the specific pollutants generated. Solid fuels and the impacts on child health in developed countries have been reviewed in Guercio *et al*. (2021).

Particulate matter emissions are likely of greatest concern, and for wood combustion 96% are in the PM_{10} size fraction and 93% in $PM_{2.5}$ (Houck and Tiegs, 1998), i.e., the majority of emissions are in the respirable fraction of PM. The size distribution and chemical characteristics of the emissions are likely to vary depending on the type of appliance, burn rates, fuel moisture content, and the type of wood used. Of further concern from a human health perspective are emissions of PAHs which may arise via pyrosynthesis during wood burning. The sulphur content of wood is however low and SO_x is not likely to represent a substantial part of emissions from wood combustion.

Chakraborty *et al*. (2020) provides an overview of existing studies that have monitored indoor pollution from residential heating stoves. In this study PM sensors were placed in 20 houses (at least 3 m from the wood burner, but in the same room), each with a Defra "Smoke Exempt Appliance" certified stove, for a period of over 4 weeks, capturing 260 usages of the stoves. They found that the mean hourly $PM_{2.5}$ concentrations were around 200% higher on days when stoves were used compared to a non-use control group. The peak hourly concentrations were correlated (*r* values of ~0.4) with the number of fuel pieces used and length of the burn period which Chakraborty *et al*. (2020) suggested was due to 'flooding' events associated with opening of the stove door.

The official testing process for stove certification is directed towards Smoke Control Area regulations so is limited to measuring outdoor air pollution *via* flue emissions and rather than indoor pollution. The Government, however, has brought in the Air Quality (Domestic Solid Fuels Standards) (England) Regulations 2020[1] that came into effect on 1st May 2021. Whilst the intention of this legislation is to improve outdoor air quality it will likely help to improve indoor air quality by limiting the moisture content of wood sold for domestic combustion. It also extends existing smoke control areas and smoke emission standards for manufactured solid fuels to cover the whole of England and phases out the use of coal for domestic burning.

Several studies have shown improvements in indoor air quality following the replacement of older sold fuel stove technologies with newer ones that meet more stringent emissions criteria (WHO, 2015). However, some caution was suggested from this WHO report in terms of the likely effectiveness of such intervention strategies due to overall costs and the long service life of appliances once they have been installed.

Candle and incense combustion can also be a significant source of indoor particles (Manoukian *et al*., 2013; Hoek *et al.,* 2008). Detailed characterisation studies show candles to emit very small particles with a modal diameter of 20-30 nm (Sun *et al*., 2006), or smaller, while those from incense tend to be larger, and typically around 100 nm diameter (Vu *et al*., 2017). Both can impact substantially upon particle number in indoor environments ($>10^5$ cm^{-3}), while the larger particles from incense burning are more likely to also influence particle mass concentrations.

1.6 Airborne biological material present within buildings

Microbes are ubiquitous in indoor environments and they are in every breath we take (WHO 2009). Exposure to bioaerosols is associated with both positive and negative health outcomes. On the one hand their presence is necessary for the growing human as they

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[1] **https://www.legislation.gov.uk/uksi/2020/1095/contents/made**

condition the maturation of a healthy immune system (Kirjavainen *et al*. 2019), whilst on the other hand exposures can lead to respiratory symptoms, asthma and infections (WHO 2009). This is especially true for spore-forming moulds which have been perennially linked to health impacts such as an increased risk of asthma (Karvonen *et al*., 2015). Indoor bioaerosols are derived from three main sources: i.) respirable contagious microbes released by humans (principally viruses and bacteria), ii.) bioaerosols produced by microbes that grow on substrates indoors (bacteria and fungi) and aeroallergens that are released by animals in the home (such as dust-mites and pets), iii.) those bioaerosols that are imported from outdoors (e.g. pollens and spores). Therefore, understanding and managing indoor bioaerosols needs to consider all three sources.

i. Respirable contagious bioaerosols.

Individuals who have a microbial infection can create aerosolised particles that can transmit the infection, especially for respiratory pathogens that are adapted for airborne transmission. The likelihood that transmission will occur depends on many factors including the biology and epidemiology of the microbe, its longevity outside the body, the particle size that encapsulates the infectious agent, and the dilution of infectious aerosols by ventilation. While this variation means that a ‘one size fits all’ approach cannot be adopted towards controlling the panoply of respiratory infections, appropriate hygiene approaches such as ‘Catch it, Kill it, Bin it’, the wearing of face masks (Centre for Disease Control, USA, 2021) and effective ventilation (Morawska *et al.* 2020) will lessen the transmission of these aerosols indoors. With the recognition that COVID-19 is largely contracted through the inhalation of SARS-CoV-2 virus-laden aerosols that are >1 um (Wang *at al.* 2021), these methods of controlling infectious aerosols are becoming widely adopted, principally through the widespread use of masks (Cheng *et al.* 2021) alongside operating buildings in a manner that encourages ‘infection resilient environments’[2].

ii. *Bioaerosols from microbes that grow on indoor materials*.

Dampness and humidity in homes are associated with a range of respiratory health effects that include asthma, wheeze, cough, respiratory infections and upper respiratory tract symptoms in many studies (WHO, 2009; Institute of Medicine 2004). While evidence points to these health impacts owing to bioaerosol exposures, the relative contribution of individual species of indoor-dwelling microbes to observed symptoms is generally unclear, as are the causal pathways. The key factor influencing the respirable burden of mould-forming fungi and bacteria is the presence of excess moisture on surfaces and in the air (WHO, 2009). This in turn is influenced by the type of materials used in the home that may support biological growth to different extents. Dampness not only initiates the degradation of materials in the home, it allows the growth of microbes which then pollute the air with their spores, allergens, volatile compounds and toxins (see 3.1.5). Accordingly, controlling sources of dampness can lead to health improvements, such as has been shown by

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[2] https://www.raeng.org.uk/publications/reports/infection-resilient-environments

intervention studies whereby removing sources of dampness has reduced asthma exacerbations in highly symptomatic children (Kercsmar *et al.,* 2006).

The presence of actively growing bacteria, moulds and dust mites, and their attendant bioaerosols, is associated with the relative humidity of indoor air. Relative humidity in its turn is related to air temperature and the ventilation characteristics of the building. Moisture levels that allow the growth of dust mites and microbes vary according to the species in question. For instance, dust mites can survive in very dry environments, requiring a relative humidity of >45% (Institute of Medicine, 2004), while surfaces can generally be kept free of moulds if the relative humidity is kept below 75%. Growth of moulds is influenced by the characteristics of specific materials; critical relative humidity for growth on construction materials varies from 75-80% for wood and wood-based materials, 80-85% for paper on plasterboard and 90-95% for polystyrene and concrete (Johansson *et al*., 2005). Managing microbial growth through controlling relative humidity can be best achieved by addressing moisture-related problems in buildings including leaking, flooding, water wicking through porous building materials, unvented sources (bathrooms and laundry rooms) and condensation on cooler surfaces such as windows and thermal bridges in insulation. Many of these issues are addressed by better building codes for new buildings, and can also be mitigated through behavioral and mechanical interventions in existing buildings.

iii. Importation of outdoor bioaerosols.

While ventilation is key to managing levels of moisture in homes, it also allows the ingress of aerosols from outdoors (see section 3.1.2), including tree and grass pollens, fungal spores and airborne bacteria. For this reason, those who have hayfever or asthma should reduce ventilation during high-risk periods, for instance when high pollen counts are forecast or during thunderstorms, which can release substantial bioaerosols leading to ‘thunderstorm asthma’.

1.7 Emissions from people

1.7.1 Carbon dioxide

Human emissions of CO_2 through breathing are considered to be ‘carbon neutral’, i.e. not adding to atmospheric CO_2 concentrations and contributing to climate change. This is because the CO_2 exhaled is part of the natural closed-loop carbon cycle and is balanced by carbon captured by the plants that are grown and consumed by humans. In the context of this report, CO_2 emissions from people contribute to a deterioration of indoor air quality, particularly in rooms that are inadequately ventilated and/or have higher concentrations of people, such as classrooms, theatres, places of worship and entertainment venues. Elevated concentrations of CO_2 have also been linked to reduced cognitive performance and attention, as detailed in section 3.2.1 Carbon dioxide.

Rates of CO_2 emissions from exhalation vary widely from person to person and depend on their activity – sedentary behaviour producing much lower emissions of CO_2 than active sport and exercise. It is therefore difficult to define a single 'typical' rate of CO_2 emission per person. A study in South Korea calculated CO_2 generation rate from breathing was 19.9 ± 5.3 $l\ h^{-1}$ during sleep and 15.1 ± 5.7 $l\ h^{-1}$ in a sedentary state (Park *et al*. 2019). This is equivalent to 35.8 g $hour^{-1}$ of CO_2 during sleep and 27.2 g $hour^{-1}$ CO_2 in a sedentary state (using a conversion of 1 litre of CO_2 = 1.8 g at 1 atm pressure and 298 K). An article in the New York Times from 1990 quoted a figure of 456 litres/person/day, equivalent to 821 g/person per day (NY Times, 1990). A study by Koerner and Klopatek (2002) calculated that human respiration emits 31.5 mol CO_2/person/day, equivalent to 1.386 kg CO_2/person/day (CO_2 molecular weight of 44.01 g mol^{-1}). This figure was calculated for an average person weighing 70 kg with a metabolic rate per day around twice that of the resting metabolic rate. A post on Globe Scientists' Blog estimated between 0.7 and 0.9 kg/person/day depending on the method used and assumptions made (Globe.gov, 2008). This is the more widely accepted scale of daily CO_2 emissions from human exhalation, around 1 kg.

Together with CO_2, an inactive adult exhales typically about 400 mL day^{-1} of water and a similar amount is lost from the skin; both can contribute to the build-up of moisture discussed before. The accumulation of CO_2 in occupied rooms has been proposed as a proxy for the ventilation rate in general and human emissions in particular (Section 3.2.1), including within the context of controlling aerial COVID transmission.

1.7.2 Volatile Organic Compounds

Numerous studies (e.g. Tamas *et al.*, 2006, Weschlet *et al.* 2007, Wisthaler *et al.* 2021) have shown that the presence of people indoors decreases ozone concentrations, while VOC (e.g. mono- and dicarbonyls) concentrations increase (Lakey *et al*., 2016, Weschler 2016). In particular, Wisthaler and Weschler's 2010 study found that squalene, the most abundant unsaturated constituent of skin lipids, is the major reactor with ozone at the skin to indoor air interface (Wisthaler and Weschler, 2010). The increases in VOC concentrations are caused by two processes: (i) ozone reacts with components of skin oil to produce secondary VOCs (ii) VOCs emitted directly from breath.

Skin oils contain wax esters, glycerols, fatty acids, squalene, esters and sterols and contain unsaturated carbon bonds (C=C) which readily react with ozone, producing a wide range of secondary products, including aldehydes, ketones, acids and secondary organic aerosols (SOA) (eg Wells *et al.,* 2008; Mochalski *et al.*, 2014) The main products of the ozone-squalene reaction are 4-oxopentanal (4-OPA), 6-methyl-5-hepten-2-one (6-MHO), acetone and geranyl acetone (Fruekilde *et al.*, 1998), and following ozonolysis of unsaturated fatty acids, higher aldehydes can be formed, namely hexanal, heptanal, octanal, nonanal, decanal, dodecanal and undecanal. Wolkoff *et al*. showed that 4-OPA and 6-MHO formed through the squalene-ozone reaction are potential sensory and pulmonary irritants and may cause airflow limitation. Mixing ratios of these individual compounds are likely to be in the low ppb range based on limited measurements (Fischer *et al.*, 2013).

Breath is also a source of VOCs indoors, including alcohols, hydrocarbons, aldehydes and ketones (Fenske and Paulson, 1999), with mixing ratios in the exhaled breath of healthy individuals ranging from ppb to ppm (e.g., Conkle *et al.*, 1975; Phillips and Greenberg, 1991;

Hansel *et al.*, 1995; Taucher *et al.*, 1995). Concentrations indoors will depend on the volume of the indoor space, the air exchange rate, the number of individuals present and also individual variations such as dietary habits (Filipiak *et al*., 2012). VOCs are present in food and drinks, which may contribute to the VOCs detected in exhaled breath. For instance, aldehydes can be used as flavoring agents and alcohols (i.e. ethanol) are typically found in coffee, tea, beverages and food (i.e. vegetables, fruits, cheese or meat) (Burdock 2016). For a model study in a simulated school classroom, acetone mixing ratios increased from 3 ppb in the empty classroom to 22 ppb when children were present (Kruza and Carslaw 2019). Methanol, ethanol, isoprene and iso-propanol mixing ratios also increased by a few ppb.

Ventilation has different effects on these two sources of VOCs; the impact of breath emissions is reduced with ventilation, as higher ventilation rates dilute breath emissions. However, the skin oil oxidation VOCs are ozone-derived species. Higher ventilation rates lead to higher indoor ozone concentrations, as indoor ozone derives largely from outdoors. Consequently, skin-oil derived VOCs will increase with ventilation up to the point where loss to dilution exceeds production at higher air exchange rates.

Additional people-related VOC emissions can arise following the use of personal care products, such as fragrances and body sprays. These emissions can include synthetic materials such as siloxanes. For instance, Tang *et al*. measured mixing ratios of selected VOCs in a university classroom and also the emission rates/person based on these concentrations. There was a clear decline in emissions over the course of the day, as the personal care products worn by the students gradually degassed from their bodies. The ‘natural’ human emissions, have been estimated to be only~ 5% of the total human emissions, presented by Tang *et al.*, which included VOCs derived from personal care products. A 2021 study by Liu *et al*. investigated skin oil VOC oxidation products in a real life setting of a two-person dwelling in California, US. A key finding in the study was that VOCs (in particular, 6-MHO and 4-OPA) continued to be produced for up to five days after the occupants had been away. This suggests that the ozone reaction with skin lipid (squalene) does not only occur at the skin to air interface, but with squalene residue deposited on indoor surfaces (Liu *et al*., 2021).

Chapter 2 - Current regulations and guidelines on indoor air.

2.1 Indoor Air Quality legislative and policy framework

The current UK air quality regulatory framework is focused on ambient outdoor air quality and is currently derived from a mixture of domestic, EU and international legislation. There are three main strands:

1. Legislation regulating total national emissions of air pollutants – the UK has implemented EU law (the National Emission Ceilings Directive)[1] and international law (the Gothenburg Protocol to the UNECE Convention on Long-range Transboundary Air Pollution) through the National Emission Ceilings Regulations 2018[2], setting emission reduction commitments for overall national emissions of five key air pollutants (NO_x, SO_2, ammonia (NH_3), $PM_{2.5}$, non-methane VOCs (NMVOCs)).

2. Legislation regulating concentrations of pollutants in ambient air – the UK has implemented EU law (the Ambient Air Quality Directive[3]) through the Air Quality Standards Regulations 2010[4] in England (and equivalent legislation in the Devolved Administrations) which set legally binding limits for concentrations in outdoor air of major air pollutants that impact public health.

3. Legislation regulating emissions from specific sources, such as legislation implementing the Industrial Emissions Directive[5], Medium Combustion Plant Directive[6], and the Clean Air Act[7].

There are currently no regulations dedicated specifically to the issue of determining limits for indoor air pollution. However, there are a number of regulations that restrict emissions of specified compounds and pollutants in some indoor environments.

2.1.1 Workplace exposure limits

HSE has a well-established regulatory framework in place to protect workers from health risks associated with exposure to hazardous substances. The Control of Substances Hazardous to Health Regulations 2002[8] (as amended) (COSHH) requires employers to prevent or control exposure to hazardous substances. COSHH is supported by specific Workplace Exposure Limits (WELs), which set thresholds for substances in workplace environments (including indoor environments). Where a WEL is set for a substance, the employer must ensure that the WEL is not exceeded and for carcinogens, mutagens or asthmagens, is controlled to be as low as is reasonably practicable (ALARP). This means improving controls and mitigation until the cost of further reduction in exposure becomes grossly disproportionate when weighed against the benefit gained.

Not all hazardous substances have been assigned a WEL. In the absence of a WEL it is the responsibility of the duty holder to ensure that the principles of good practice are applied. This means exposure to hazardous substances must be adequately controlled in relation to the health risk, which in effect is the same as ALARP. WELs are approved only for application to people at work, however, employers should also take into account their duties under other relevant legislation such as the Environmental Protection Act.[9]

2.1.2 REACH

The Registration, Evaluation, Authorisation and Restriction of Chemicals[10] (REACH) is a key piece of legislation for regulating the use of chemicals to protect human health and the environment. In general, REACH makes those who place chemicals in any form on the market responsible for providing information on them so that any hazards are understood and the risks associated with their use are properly managed. It therefore applies to manufacturers, importers, professional users and distributors of chemicals. REACH also has authorisation and restriction as mechanisms for introducing additional controls on chemicals. When a substance is made subject to authorisation, a company cannot use the substance beyond the sunset date unless they have been granted a use-specific authorisation. Restriction can introduce a range of controls if an unacceptable risk to human health or the environment is identified. Restriction has been used to introduce protections on air quality (both indoor and outdoor) in specific circumstances; for example, REACH restricts the use of dichloromethane in paint stripper to protect human health.

2.1.3 Volatile Organic Compounds in Paints, Varnishes and Vehicle Refinishing Products Regulations 2012

The Volatile Organic Compounds in Paints, Varnishes and Vehicle Refinishing Products Regulations 2012[11] are intended to complement measures taken to ensure compliance with national emission ceilings and reduction commitments for VOCs. These regulations limit the total content of VOCs in certain paints, varnishes and vehicle refinishing products. As many of these products are used indoors, these regulations have a positive impact on indoor air quality by reducing emissions when the products are used.

2.1.4 Domestic Combustion legislation

Legislation regulating emissions from domestic burning, primarily aimed at reducing ambient emissions of particulate matter, will also reduce indoor air pollution.

The Clean Air Act 1993 enables local authorities to create smoke control areas in which it is an offence to emit smoke from the chimney of a building in a smoke control area unless using an authorised fuel or exempt appliance. The Environment Act 2021 amends the Clean Air Act 1993 to enable local authorities in England to issue financial penalties to those emitting smoke from their chimney in Smoke Control Areas. This will be achieved by replacing the criminal offence with a civil penalty regime, which will also allow for the removal of the statutory defences that currently hinder enforcement. This change will ensure local authorities can avoid lengthy and costly court cases in enforcing smoke emissions.

The Air Quality (Domestic Solid Fuels Standards) (England) Regulations 2020[12], which came into force on 1 May 2021, phases out the sale of bituminous coal (traditional house coal) for domestic burning, introduces restrictions on the sale of wet wood and limits on the emission of sulphur and smoke from manufactured solid fuels. The aim is to encourage householders to burn better quality, cleaner fuels. As burning occurs indoors in open fires and solid fuel appliances, this legislation is expected to reduce harmful indoor emissions.

The Ecodesign Regulations[13] set more demanding emission standards and efficiency requirements for all new solid fuel appliances from 1 January 2022. The test standards for these appliances are still under discussion and as such it is too early to assess the impact these will have on indoor air quality, though it is expected that there will be a positive impact.

2.1.5 Construction Products Regulation

The Construction Products Regulations 2013[14] cover approximately 400 product types where there is an existing designated standard. Manufacturers whose products fall under the regulation are required to declare the product’s performance against the designated standard and put in place measures to ensure their products continue to meet this standard.

The Construction Products Regulations 2013[15] requires designated standards to be prepared on the basis of basic requirements for hygiene, health and the environment set out in the regulation. This includes the requirement that works be designed and built in a way that they will not be a threat to hygiene or health and safety, which includes, for example, ‘emissions of dangerous substances, volatile organic compounds (VOC), greenhouse gases or dangerous particles into indoor or outdoor air’[16]

2.1.6 Building Standards and Regulations

Building Regulations approval is required for most building work in the UK. The Regulations are devolved and there are thus differences between their formulation in the different countries of the UK. Approved Documents (in England and Wales) and Technical booklets/handbooks (in Northern Ireland and Scotland) provide guidance on complying with each country’s Regulations. Differences between these documents are complex and so AQEG has focused on the Building Regulations 2010 for England and Wales[18] (MHCLG, 2010), and on the Approved Documents for England. While details vary across the four UK nations, the overall intentions which underpin the documents are aligned.

Approved Document Part F sets minimum standards for adequate ventilation in new buildings and when work is done to existing buildings. It provides guidance on minimum ventilation rates and performance-based ventilation. Using the performance-based ventilation route a designer can tailor their ventilation strategy by controlling moisture, indoor air pollutants and bio-effluents to the levels described in guidance. Approved Document F specifies whole dwelling ventilation rates to supply air to habitable rooms according to the number of bedrooms, the number of occupants and a minimum ventilation rate by internal floor area. CIBSE AM10 (2005) provides guidance on how to select a ventilation strategy, and guidance on mixed-mode strategies is provided in CIBSE AM13 (2000).

In residential environments, ventilation guidelines are largely driven by the control of humidity levels, as well as pollutants from combustion (e.g. cooking hobs) and odours. Guidance can be found in CIBSE Guide A (2015) Table 1.5 and CIBSE TM60: Good Practice in the design of homes (2018) F. These include minimum whole dwelling rates and minimum rates in kitchens and wet rooms. The recommended range in Guide A is 0.4 to 1 air changes per hour (ach) in living rooms and bedrooms, with higher extract rates in kitchens and wet rooms.

Ventilation rates in non-residential buildings are often driven by factors other than humidity and indoor pollutants which sometimes leads to high ventilation rates. This which may result in excessively low humidity levels in winter, leading to occupant discomfort and/or a need for humidification and associated energy consumption. Ventilation guidelines are also provided by more stringent energy standards such as Passivhaus (new buildings) / EnerPHit (retrofits). Minimum whole building ventilation rates are shown in Table 2.1 below.

Table 2.1. Minimum whole building ventilation rates, from Approved Document F for Dwellings.

Number of Bedrooms	Minimum ventilation rate by number of bedrooms, litres per second
1	19
2	25
3	31
4	37
5	43

Notes:

1. If the dwelling only has one habitable room a minimum ventilation rate of 13 litres per second should be used
2. For each additional bedroom add 6 litres per second to the values in the Table

For offices, Approved Document F specifies that outdoor air should be supplied for occupied rooms at 10 litres per second per person or 1 litre per second per m^2 floor area, depending whichever will provide a higher total rate. Common spaces such as corridors and lift lobbies, should be provided with either: a. natural ventilation by appropriately located ventilation opening(s) with a total opening area of at least 1/50th of the floor area of the common space; b. mechanical ventilation installed to provide a supply of outdoor air of 0.5 litre per second per m^2 of floor area.

In the Passivhaus standard, the ventilation strategy is mechanical ventilation with heat recovery (MVHR). The overall ventilation rate based on the maximum design occupancy to sustain good indoor air quality level is 30 m^3/h or 8 litres per second (l/s) of fresh air supply per person. Bedrooms could have 30 $m^3 h^{-1}$ or 6 $l s^{-1}$ of fresh air supply per person. The extract requirements for wet rooms, kitchens and rooms with pollution sources are:

- Kitchen: 46 m^3/h (13 l/s) | 60 $m^3 h^{-1}$ (17 $l s^{-1}$) in boost mode
- Bathroom or shower room: 31 $m^3 h^{-1}$ (7 $l s^{-1}$) | 40 $m^3 h^{-1}$ (11 $l s^{-1}$) in boost mode
- Utility, WC or storage: 15 $m^3 h^{-1}$ (4 $l s^{-1}$) | 20 m^3/h (6 $l s^{-1}$) in boost mode

It is vital that fresh air entering the building does not exceed 30 m^3 per hour per person, to avoid overly dry air.

Ventilation requirements are also often defined through Part L of the Building Regulations and Approved Document L, which addresses the conservation of fuel and power and, in particular, thermal comfort. Part J of the Building Regulations deals with combustion appliances and fuel storage. It includes provisions to ensure adequate air supply to combustion appliances and discharge of combustion products, as well as warnings of carbon monoxide release, protection from fire, and preventing the release of oil.

Approved Document J focuses on small appliances typical in residential dwellings. It provides guidance on venting emissions from, and air supply to, indoor combustion, including small combustion appliances and open fireplaces. Air for combustion, and operation of the flue, may come from inside the room only if sufficient ventilation is provided. The design and siting of flues for gas and oil appliances is controlled with reference to minimum separation distances from windows, doors or other openings, and from the property boundary. For small solid fuel appliances, minimum rooftop flue clearances are defined in relation to the roof ridge and proximity to adjacent buildings. By way of example, a solid fuel appliance flue must be at least 60 cm taller than any adjacent building within 2.3m. Larger appliances, such as those commonly used in commercial environments or apartment blocks, are covered by the Regulations but not the Approved Document.

List of regulations referenced in this section

[3] Directive 2008/50/EC of the European Parliament and of the Council on ambient air quality and cleaner air for Europe (OJ L 152 11.6.2008, p. 1)

[4] S.I 2010/1001

[5] Directive 2010/75/EU of the European Parliament and of the Council on industrial emissions (integrated pollution prevention and control)

[6] Directive (EU) 2015/2193 of the European Parliament ad of the Council on the limitation of emissions of certain pollutants into the air from medium combustion plants

[7] Clean Air Act 1993

[8] S.I 2002/2677

[9] Environmental Protection Act 1990

[10] S.I 2019/758

[11] S.I 2012/1715

[12] S.I 2020/1095

[13] S.I 2019/539

[14] S.I 2013/1387

[15] S.I 2013/1387

[16] Annex 1 3(b) Construction Products Regulation 2013

[18] S.I 2010/2214

2.2 Indoor concentration standards or recommendations

Approved Document F cites the WHO (2010) and PHE (2019) guidelines for selected pollutants / VOCs. The same guidelines are cited by the NICE (2020) guidelines on indoor air quality at home that focus on interventions. Table 2.2 contains a summary of current guidelines on UK indoor air quality and summary of the guideline values by WHO (2005) – ambient air, WHO (2010) – IAQ, PHE (2019) – IAQ, VOCs.

Lowther *et al*. (2021) carried out a literature review to (a) evaluate the reported health effects and physiological responses associated with exposure to less than 5000 parts per million (ppm) of CO_2 and (b) to assess the CO_2 guideline and limit concentrations in the context of (a). It is not possible to say with confidence whether CO_2 alone is responsible for health effects at exposure levels below 5000 ppm, measured in indoor environments, due to the design of various studies. However, the existing guideline CO_2 concentrations applied in various standards and regulations, across a number of organisations and countries, can be indicative of ventilation, human bio-effluent (chemical contaminant emitted during the metabolism process of a person) and indoor air pollution concentrations; therefore, the current consensus that ≤1000 ppm, 1000 - 1500 ppm, >1500 ppm represent good, moderate and poor indoor air quality respectively seems appropriate.

Table 2.2: Summary of UK and other international guidelines for limits on indoor concentrations of various air pollutants.

Pollutants	**WHO Air Quality Guidelines for ambient air (WHO, 2005)**	**WHO Indoor Air Quality Guidelines (WHO, 2010)**	**PHE Indoor Air Quality Guidelines for selected VOCs (PHE, 2019)**
CO (mg/m^3)	-	100 (15 min)	-
	-	60 (30 min)	-
	-	30 (1 hr)	-
	-	10 (8 hr)	-
	-	7 (24 hr)	-
NO_2 ($\mu g/m^3$)	200 (1hr)	200 (1hr)	-
	40 (1yr)	40 (1yr)	-
SO_2 ($\mu g/m^3$)	500 (10min)		-

Pollutants	**WHO Air Quality Guidelines for ambient air (WHO, 2005)**	**WHO Indoor Air Quality Guidelines (WHO, 2010)**	**PHE Indoor Air Quality Guidelines for selected VOCs (PHE, 2019)**
	20 (24 h)		-
PM$_{10}$ (µg/m^3)	50 (24 h)	[50 (24 h)]*	-
	20 (1 yr)	[20 (1 yr)]*	-
PM$_{2.5}$ (µg/m^3)	25 (24 h)	[25 (24 h)]*	-
	10 (1 yr)	[10 (1 yr)]*	-
Ozone (µg/m^3)	100 (8 hr)		-
Radon (Bq/m^3)	-	No safe level	-
	-	Reference level: 100	Target Level: 100 Action level: 200 (see HPA, 2010)
	-	No more than: 300	300 (1 yr)
Benzene (µg/m^3)	-	No safe level: The concentrations of airborne benzene associated with an excess lifetime cancer risk of 1/10 000, 1/100 000 and 1/1 000 000 are 17, 1.7 and 0.17µg/m^3	As WHO (2010)
Trichloroethylene (µg/m^3)	-	No safe level of exposure can be recommended. The concentrations of trichloroethylene associated with an excess lifetime cancer risk of 1/10 000, 1/100 000 and 1/1 000 000 are 21, 2.1 and 0.21 µg/m^3, respectively.	As WHO (2010)
Tetrachloroethylene (µg/m^3)	-	250 (1yr)	40 (1 day)
Formaldehyde (µg/m^3)	-	100 (30 min)	100 (30 min)
	-		10 (1yr)
Naphthalene (µg/m^3)	-	10 (1yr)	3 (1yr)
PAHs (ng.m^{-3} B[a]P)	-	No safe level	-
Acetaldehyde (µg/m^3)	-	-	1,420 (1h)
	-	-	280 (1 day)
α-Pinene (mg/m^3)	-	-	45 (30 min)
	-	-	4.5 (1 day)
D-Limonene (mg/m^3)	-	-	90 (30 min)
	-	-	9 (1 day)

Pollutants	WHO Air Quality Guidelines for ambient air (WHO, 2005)	WHO Indoor Air Quality Guidelines (WHO, 2010)	PHE Indoor Air Quality Guidelines for selected VOCs (PHE, 2019)
Styrene (µg/m^3)	-	-	850 (1y)
Toluene (mg/m^3)	-	-	15 (8h)
	-	-	2.3 (1 day)
Xylenes (µg/m^3)	-	-	100 (1y)

* According to WHO (2010), the air quality guidelines for particulate matter recommended by the 2005 global update are also applicable to indoor spaces, since there was no convincing evidence of a difference in the hazardous nature of particulate matter from indoor sources compared with those from outdoors

2.3 Labelling schemes of relevance to indoor emissions and air quality

Public awareness of the risk from chemicals is generally poor. Labelling is one mechanism that consumers can use when making purchases, in order to reduce their exposure to chemicals. In the Clean Air Strategy (Defra, 2019), the government was committed to "*work with industry on how best to enable consumers to make informed choices and to switch to low-VOC content products wherever possible*". An option is the development of voluntary labelling schemes for NMVOC-containing products. Labelling schemes have adopted in many European countries. Existing initiatives which could be explored alongside a UK voluntary labelling scheme can be found in the following Table 2.3.

For the development of voluntary labelling schemes for construction products, the harmonised EU-LCI (EU Lowest Concentration of Interest) values might be considered (see https://ec.europa.eu/growth/sectors/construction/eu-lci/values_en) These are health-based reference concentrations of chemical substances for inhalation exposure used to assess emissions after 28 days from a single construction product during a laboratory test chamber procedure as defined in the EN 16516. EU-LCI values are applied in product safety assessment with the goal of avoiding health risks from long-term exposure of the general population. They are expressed in µg m^{-3}.

The derivation of EU-LCI values involved three main steps: compilation of toxicological data, data evaluation, and derivation of the EU-LCI value on the basis of a total (combined) assessment factor, based on established risk assessment principles and expert judgment, laid out in a standardised factsheet.

Table 2.3: Building materials, product labels on chemical emissions in EU (from DFE, 2018. BB101 Guidance on ventilation, thermal comfort and indoor air quality in schools)

Building materials and products labels and guidance on chemical emissions in EU

- European Ecolabel (e.g. textile-covered flooring, wooden flooring, mattresses, indoor and outdoor paints and varnishes: Europe), **http://ec.europa.eu/environment/ecolabel/**
- EMICODE® (adhesives, sealants, parquet varnishes and other construction products: Germany/Europe), **http://www.emicode.com/index.php?id=1&L=1**
- GUT (carpets: Germany/Europe), **http://pro-dis.info/86.html?&L=0**
- Blue Angel (Germany), **http://www.blauer-engel.de/en/index.php**
- Nordic Swan (Scandinavia), **http://www.svanen.se/en/Nordic-Ecolabel/**
- Umweltzeichen (Austria), **http://www.umweltzeichen.at/cms/home233/content.html**
- AgBB (Specifications for construction products: Germany), **http://www.umweltbundesamt.de/themen/gesundheit/kommissionenarbeitsgruppen/ausschuss-zur-gesundheitlichen-bewertung-von**
- M1 (construction products: Finland), **www.rakennustieto.fi/index/english/emissionclassificationofbuildingmaterials.html**
- ANSES (formerly AFSSET) (construction products: France), **http://www.anses.fr/fr/upload/bibliotheque/892980998778406505212938602998/COV_Avis_signe_2009_10.pdf**
- CertiPUR (PU foam for furniture industry: Europe), **http://www.europur.com/index.php?page=certipur**
- Ü mark (specifications in relation to CE marking: Germany), **https://www.dibt.de/index_eng.html**
- Danish Indoor Climate Label,

  http://www.teknologisk.dk/ydelser/dansk-indeklimamaerkning/dim-omfatter/253,2
- Swedish ‘byggvarudeklaration’ (construction products: Sweden, **http://www.byggvarubedomningen.se/sa/node.asp?node=455**
- Natureplus (construction products: Germany/Europe, **http://www.natureplus.org/**

Table 2.4 below reproduced from the ARUP2016 report provides the applicability of relevant labelling schemes to key product groups.

Table 2.4 Summary of EU legislation and eco label applicability to different product groups.

Eco label	Timber	Wet applied finishes	Insulation	Flooring	Panels	Adhesives and sealants	Screed	Metal
EU Ecolabel	×	✓	×	✓	×	×	×	×

Blue Angel	✓	✓	✓	✓	✓	✓	×	×
Nordic Ecolabel	✓	✓	✓	✓	✓	✓	×	×
Nature plus	✓	✓	✓	✓	✓	✓	✓	×
GUT	×	×	×	✓	×	×	×	×
EMICODE	×	×	×	×	×	✓	×	×
CARB	✓	✓	×	×	×		×	×
French Decret	✓	✓	✓	✓	✓	✓	✓	✓
AgBB	✓	✓	✓	✓	✓	✓	✓	✓
Belgian Decret	✓	✓	✓	✓	✓	✓	✓	✓

Eco-labels ✓ Product group is covered by Scheme
Legislation × Product groups is not covered by Scheme

2.4 Indoor air quality within the Government evidence system

Defra has responsibilities that sit across a variety of environmental systems with important interactions both between them and across additional systems in which other government departments have a greater stake. Air pollution is a prime example of this, where the key drivers lie in, for example, energy systems (Department for Business, Energy and Industrial Strategy, BEIS); transport systems (Department for Transport); industry (BEIS & Defra); and food systems, particularly agriculture (Defra and Department for International Trade).

Policy and regulation of air pollution is generally framed around the emissions created from individual activities within these sectors and systems, many of which are controlled by legislation set by other government departments. As a result, the evidence system for air quality policy sits across government departments both in terms of the evidence associated with controlling emissions of air pollution and the wider impacts of air pollution, which manifest in the natural environment (Defra), on human health (Department for Health and Social Care) and in the built environment (Department for Levelling Up, Housing and Communities (DLUHC, formerly MHCLG), and Department for Culture, Media and Sport), all of which ultimately lead to impacts on the economy (HM Treasury).

For indoor air quality the spread of responsibilities across government departments is perhaps even greater, with the overlaps with environmental systems limited primarily to the multiple interactions between indoor and outdoor air quality. Indoor air quality does not obviously fit within environmental systems analysis, which typically focuses on describing how human actions impact on and interact with the natural environment. For the human activities that influence indoor air, any interaction with the outdoor natural environment is arguably mediated solely through the relationships with outdoor air quality.

Domestic emissions and those from commercial and public buildings, are quantified and included in the Defra reporting of total annual national emissions through the National

Atmospheric Emissions Inventory, since they are contributors to emissions to outdoor air, via air exchange. In this sense indoor emissions are treated similarly to many other classes of pollution that contribute to outdoor air pollution.

There are only limited considerations of the reverse, that is the impact of outdoor air pollution on indoor air quality. Defra guidance to local authorities includes consideration of the potential for poor outdoor air quality to impact on indoor environments in the declaration of Air Quality Management Areas (AQMAs). These are areas that modelling, or measurement evidence suggests will have outdoor air pollution above the legal limit values. It states "*In some cases, it may be appropriate to list the individual properties affected, although there is no legal requirement to do this. Alternatively, it might be useful to include in the AQMA Order an approximate summary of the number and type of properties affected*." (Defra-LAQM, n.d.). There is no explicit requirement to consider air indoors in terms of domestic ambient concentrations of pollutants in any of the existing air quality targets or legislation.

The 2019 Clean Air Strategy (Defra, 2019) has an entire chapter dedicated to 'Action to reduce emissions in the home'. This lays out the rationale for government considering indoor air, stating that alongside the commitment to meeting legally binding targets outdoors, the wider national priority is to minimise human exposure to air pollution, and that many activities that create emissions take place in and around the home. Given the evidence that indoor air is in practice a cross-government issue the Clean Air Strategy could be seen as a vehicle for driving action on emissions, and with the Department for Health and Social Care as a key stakeholder for coordinating action on health. The Clean Air Strategy also references the further motivations for Defra engagement with issues of indoor air pollution, since emissions from homes contribute to national emissions that have legally binding limits and annual reporting requirements. Again though, this does not require knowledge of indoor air quality (expressed as concentrations or exposure) only the total tonnage of emissions from activities that take place indoors.

Many of the challenges around indoor air quality stem from deficiencies in the evidence system, particularly associated with observations. The current lack of evidence to inform effective action has been in part a consequence of a small UK research community and limited availability of funding. The historically limited funding for research is a result of many factors, but research has clearly been hampered because as a science topic indoor air quality does not sit solely within the core interests of one research council but across many due to the interactions with outdoor air quality, importance of human behaviour, the influence of building design and engineering, and the impacts on health. The way in which responsibility for indoor air quality issues is dispersed across government has also been a barrier to research with no single department setting the agenda or defining the evidence needs. All of this contributes to a lack of effective public information leading to low levels of public awareness and understanding of indoor air quality and a limited ability to make appropriately informed decisions at all levels to limit emissions of and exposure to air pollution indoors.

As the government department responsible for outdoor air quality Defra have expertise and experience that is of direct relevance to indoor air quality. The understanding of measurement technologies, monitoring strategy, emissions sources and modelling of air pollutant concentrations and their impacts are all applicable to the indoor environment. The inclusion of some considerations of indoor air pollution in Defra's Clean Air Strategy acknowledges the inextricable link between indoor and outdoor air pollution. However, in many cases the techniques used, and types of data collected by Defra are not readily applicable to indoor air quality. For example, inclusion in national emissions reporting requires nationally and annually averaged emissions which are derived from data such as sales of cleaning or personal care products rather than data that is more directly linked to any assessment of air quality in indoor settings. Measurement techniques used in outdoor monitoring networks do not quantify the all most important volatile organic compounds used as solvents in household products (Air Quality Expert Group, 2020) or speciate particulate matter in a way to enable the contribution from domestic activities, such as cooking, solid fuel burning, house dust and biological particulate matter to be characterised or quantified. More fundamentally the equipment used outdoors is rarely compatible for use within homes do to constraints on size, noise power and so on.

To assess the potential use of existing data, expertise and techniques to inform and improve indoor air quality there would have to be clearer articulation of the purpose of government intervention and therefore what evidence is needed to inform action. For example, it may not be necessary to know which pollutants are causing the greatest health impacts if the goal were to reduce all emissions of pollutants into the indoor environment and to increase ability to remove or limit exposure to emissions that cannot be eliminated. This would ideally include consideration of what relevant evidence exists across government and in the research and industrial communities already, and how best to meet the expressed goals without requiring unsustainable research investment or excessively long-time scales for evidence development. In 2010 there was a POSTNOTE on UK Indoor Air Quality published (Parliamentary Office of Science and Technology, 2010) that stated that better coordination of policy in this area was necessary and suggested the creation of an 'overall liaison group' across government. Setting up this type of cross-government group could enable improved coordination of evidence and strategy.

In February 2020, the Chief Medical Officer and Government Chief Scientist held a roundtable on indoor air quality with DHSC, PHE, BEIS, DfT, Defra and HSE. This acknowledged the evidence gaps and emphasised the need for a cross-government approach to address the issues. Given the links with health, DHSC/PHE was proposed to have an oversight role for indoor air quality (IAQ), with the support from other government departments.

In September 2021, the Cross Government Working Level Group on IAQ was established having as an objective to ensure a joined–up approach across government departments and other governmental bodies on IAQ. The following organisations will sit as members: Department of Health and Social Care (DHSC, Chair and Secretariat), UK Health Security Agency (UKHSA, formerly PHE), Cabinet Office, Defra, Department for Levelling Up, Housing and Communities (DLUHC, formerly MHCLG), BEIS Office for Product Safety and

Standards (OPSS), Department for Education (DfE), Ministry of Justice, NHS Test and Trace, Health and Safety Executive (HSE). The Group members will share knowledge and activities with the aim to identify any barriers to action and, potentially, ways to move beyond them as well as potential opportunities for cooperation.

Chapter 3 – The composition of indoor pollution

The composition of air indoors is a function of four distinct factors i) The composition of outdoor air that is drawn in to the building, ii) emissions of pollution from within the building itself, with many sources listed in chapter 1, iii) the rate of air exchange with outdoor air, which impacts not only concentrations (via dilution) but also the rates of chemical reactions occurring indoors (via ingress of oxidants such as ozone, and iv) the behaviour of occupants in the building, for example the locations, rates and types of chemicals and appliances used indoors. In this chapter the detailed chemical nature of particles and gases found indoors is expanded on, with this detailed chemical speciation and abundance affected by these four factors.

Indoor-outdoor ratios of different pollutants in different buildings provide an indication of the strength of indoor sources compared to outdoor sources, with ratios greater than one indicating that indoor sources likely contribute more to indoor concentrations. Ratios less than one indicate that outdoor sources contribute more relatively to the indoor environmental levels. Indoor-outdoor ratios depend on the concentrations of a given substance found indoors and outdoors, and therefore can vary depending on location, time of day, occupant behaviour and variation of outdoor sources, such as traffic.

3.1 Airborne Particles

3.1.1 Introduction

Indoor air pollution comprises both gases and particles and in the indoor environment the latter comprise both chemical and biological material. The composition of indoor particulate matter can be explained by considering its sources. The contributors are:

(a) Particles which have entered the indoor environment from outdoors. These are both primary and secondary and may be modified by chemical and physical processes in the indoor atmosphere,

(b) Primary particles emitted indoors, which may be chemical or biological in their nature,

(c) Secondary particles, which are those particles formed indoors by atmospheric chemical reactions often involving precursor gases released indoors.

The three sources are not always easy to distinguish, but measurement of indoor-outdoor (I/O) ratios is usually informative, as are highly time resolved measurements of indoor particles. It is worth noting that gas-particle partitioning can affect particle composition such that there could be significant impacts on health, e.g. nitric acid evaporation from nitrate-containing particles when they come into warm indoor environments (see work by DeCarlo *et al.*), phthalates partitioning to particles when they come indoors (see work by Goldstein *et al.*, and Eichler *et al.* 2019).

This section reviews knowledge of the composition and sources of indoor particles, including characterisation of the biological component. Concentrations of pollutants in the indoor atmosphere reflect a balance between the sources of input, chemical conversions and sinks, and physical sinks such as deposition to indoor surfaces. Many indoor sources, such as cooking are quite localised, and hence mixing throughout the building is also an important determinant of airborne concentrations.

3.1.2 Particles which enter from outdoors

Many studies have reported simultaneous measurements of indoor and outdoor particle concentrations (see, e.g., Bo *et al.*, 2017, Avery *et al.*, 2019). I/O ratios which are consistently less than unity and tend to change slowly with time are generally indicative of a pollutant whose major origin is outside the building. Jones *et al.* (2000), measuring in roadside, urban and rural dwellings in the Birmingham area concluded that sulphate and lead had predominantly outdoor sources on the basis of their size-dependent I/O ratios. For sulphate, which was considered to be a good indicator of the ingress of outdoor air, the I/O ratio in the PM_1 fraction was 0.8, while for PM_{10} it was 0.6. Size-dependent particle losses account for the ratio being less than unity. Zauli Sajani *et al.* (2015) studied this phenomenon by injecting outdoor air into a well-mixed unoccupied room, finding I/O ratios in the $PM_{2.5}$ size range very similar to those reported for occupied dwellings, concluding that deposition of particles to indoor surfaces and evaporation, rather than loss during penetration of the building envelope are responsible for the concentration reduction. However, theoretical studies reviewed by Hanninen and Goodman (2019) suggest losses of particles penetrating the building are high for the large (>2 µm) and very small (<0.05 µm) particles, but very low for the accumulation mode of ~ 0.1 – 1 µm diameter.

Both Zauli Sajani *et al.* (2015) and Harrison and Li (1990) noted a far smaller I/O ratio for nitrate than for sulphate, which they attributed to the loss of nitrate as nitric acid vapour to surfaces following the evaporation of ammonium nitrate. The measurements of Li *et al.* (1990) were made in an occupied building and showed ammonia I/O ratios of 3.5 – 21 due to the presence of people, which would tend to stabilise the ammonium nitrate in the absence of nitric acid loss. Ammonium also showed a low I/O ratio in the work of Zauli Sajani *et al.* (2015), while both organic carbon and elemental carbon showed high ratios, suggestive of small losses.

Zauli Sajani *et al.* (2015) also measured particle size distributions indoors and outdoors in buildings both close to, and remote from heavy traffic. There was a large loss indoors of smaller particles, particularly those < 50 nm diameter, consistent with this fraction being semi-volatile, leading to evaporative loss or shrinkage of these smaller traffic-related particles, after allowing for enhanced deposition. This was most clearly observable at the traffic influenced site where this size range of particles was very prominent in the outdoor air size distribution.

Sampling in unoccupied rooms overlooking a busy highway in central London, Ni Riain *et al.* (2013) found differences between naturally and mechanically ventilated buildings and a marked sensitivity of I/O ratios to wind direction for the naturally ventilated rooms. I/O ratios soon established rather stable levels after mechanical ventilation was commenced, while the ratios in naturally ventilated rooms were far more temporally variable.

The Indoor to Outdoor (I/O) ratios for NO_2 and $PM_{2.5}$ for around 1.6 million spatially-referenced London dwellings have been estimated by Taylor *et al.* (2019) using a building physics metamodel. The critical assumptions in their model were that fractional losses per hour due to deposition were 0.87 for NO_2 and 0.19 for $PM_{2.5}$, whilst 10% of outdoor $PM_{2.5}$ was filtered in the heating season. With these assumptions the I/O ratios for NO_2 (mean: 0.4, range: 0.3 - 0.6) and for $PM_{2.5}$ (mean: 0.6, range: 0.5-0.7) predicted by Taylor *et al*. (2019) compare well with previous estimates of the I/O ratios predicted by the INDAIR/EXPAIR modelling framework (Dimitroulopoulou *et al*., 2006) (for NO_2 mean: 0.5, for $PM_{2.5}$ mean: 0.6, for non-source scenarios). The implications of these I/O ratios for air exchange rates are discussed in quantitive terms in the examples given in Appendix 1.

3.1.3 Primary particles emitted indoors

Many studies have found I/O ratios greater than unity, indicative of indoor sources. Thus, for example, during their Birmingham measurements, Jones *et al.* (2000) reported I/O ratios of zinc, organic carbon (OC) and elemental carbon (EC) which well exceeded unity in a sub-set of the homes which they studied. On the other hand, Ho *et al*. (2004) sampling in roadside homes in Hong Kong found I/O ratios for OC and EC of 1.02 and 0.80 respectively, attributing EC to outdoor traffic sources and OC to both indoor and outdoor sources. They found much greater attenuation of concentrations in mechanically ventilated buildings, presumably due to inclusion of air filters in air conditioning systems, but also possibly linked to greater particle losses in the ducting. Concentrations of polycyclic aromatic compounds were generally lower indoors than outdoors, except in the presence of indoor sources such as wood burning stoves and tobacco smoke (Delgado-Saborit *et al.,* 2011).

Real time particle measurements are very revealing of indoor particle sources. This is exemplified by Figure 2.1 which shows the major influences of cooking and human activity upon indoor PM_{10} measured with high time resolution.

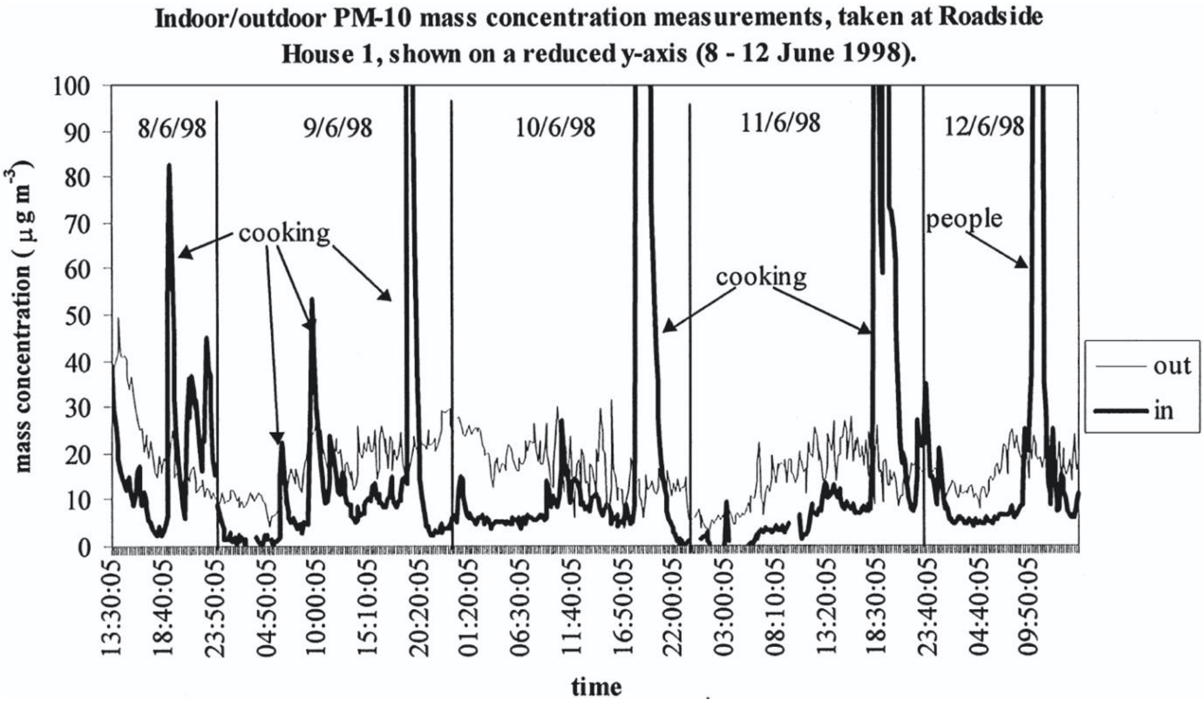

Figure 3.1: Mass concentrations of PM$_{10}$ inside and outside Roadside House 1 (from Jones et al., 2000)

Vu *et al.* (2017) measured nanoparticle emissions arising from five indoor sources. Maximum number concentrations generated in an apartment were 9.38×10^4, $1.46{\times}10^5$, $2.89{\times}10^4$, 2.25×10^5 and 1.57×10^6 particles cm^{-3} for particles released from vacuum cleaning, soap cleaning spray, smoking, incense burning and cooking (frying) activities, respectively. Many studies have reported particle number counts exceeding 10^5 cm^{-3}, with modal particle diameters from 20 – 150 nm diameter (Abdullahi *et al*., 2013). The use of particle counters, which reflect mainly the very abundant nanoparticle fraction is very effective in demonstrating indoor sources. Working in four European cities, including Birmingham, Hoek *et al*. (2008) reported low correlations between indoor particle number concentrations and those outside the home, with major excursions of indoor concentrations associated with activities such as cooking and candle burning. Some hourly indoor particle number counts exceeded 10^5 cm^{-3}, well above the highest outdoor concentrations, although the median indoor concentrations fell below those outdoors. The broad implications for particle source number emission rates of the observed concentrations is discussed further in Appendix 1.

The recent HOMEChem (House Observations of Microbial and Environmental Chemistry) experiment in a test-house in Texas provided some interesting insights into PM behaviour during cooking (Farmer *et al*., 2019; Patel *et al.,* 2020). During this campaign, many experiments focused on cooking and Figure 3.2 reproduced from Farmer *et al*. (2019) shows that the type of meal prepared, affected both PM mass concentration and the resulting size distribution of particles. Breakfast in this case with a typical 'full English' involving frying bacon and sausages and producing a mass concentration of around 300 µg m^{-3}, with the most significant mass fraction of particles in the size range between 0.1 - 0.5 µm. However, the highest concentrations were observed for a stir fry meal, this time with the highest proportion of particle mass in the 2.5 - 10 µm range. In both cases almost all particles by number were in the ultrafine range of <100 nm (Figure 3.2).

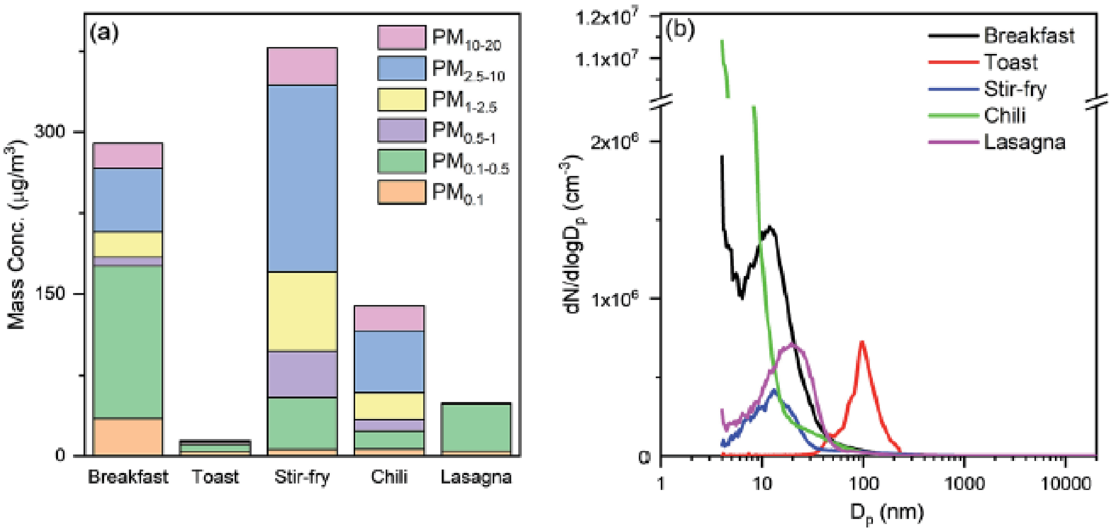

Figure 3.2: PM concentrations and size distributions from cooking experiments during the HOMEChem campaign (taken from Farmer et al., 2019).

Patel *et al*. (2020) used these experiments to calculate the PM respiratory deposition for an adult in the test house kitchen over a 12-hour period in the HOMEChem kitchen. They found that in the absence of cooking, this value was 7 µg, with 68 µg for a day where three meals were cooked and interspersed with cleaning activities and 149 µg where a Thanksgiving meal was prepared.

3.1.4 Secondary particles, formed indoors by atmospheric chemical reactions

Organic vapours are liable to undergo oxidation reactions in the indoor atmosphere forming less volatile products which may partition to the particulate phase (although they can also condense onto other surfaces which act as reservoirs). These are likely to include particles formed from cleaning agents containing reactive fragrances, as reported by Vu *et al.* (2017).

Ozone-monoterpene reactions can produce products which are sufficiently involatile to promote particle formation and growth (Weschler and Shields, 1999; Sarwar *et al*., 2003; Walser *et al*., 2007). Following oxidation, they can produce a wide range of multifunctional species, including alcohol, aldehyde, ketone and carboxylic acid groups: such groups tend to have lower vapour pressures than the parent terpene and can condense to form secondary organic aerosol (SOA) (Walser *et al*., 2007). The chemically active nature of secondary organic compounds may mean that they could potentially be more toxic than other forms of particulate matter (Tong *et al*. 2018).

The effectiveness of different terpenes in forming SOA can be compared through consideration of aerosol yields, defined as the amount of SOA formed from the oxidation of a given hydrocarbon to the amount of the parent hydrocarbon reacted (Griffin *et al*., 1999). Oxidation of monoterpenes produces aerosol yields that are dependent on the experimental conditions, the starting concentration of the parent monoterpene, and on the oxidant used (*e.g.* OH, O_3 or NO_3). Jenkin (2004) found that dark ozonolysis experiments appear to give the highest aerosol yields, which is an interesting observation when considering the indoor environment with lower light levels. SOA yields of up to 54% have been reported for the ozonolysis of terpenes used in fragrances, with highest values for species containing one or more internal double bonds, such as alpha-pinene and limonene (Lee *et al*., 2006, Chen and Hopke 2012). There are many indoor activities such as cleaning (Finewax *et al.* 2020) and applying personal care products that are known to produce large quantities of such compounds. It has also been shown that cooking fumes may also produce SOA (Zhang *et al.* 2020), although many of these cooking experiments have typically been conducted in chemical regimes more representative of outdoor rather than indoor chemistry.

Weschler and Shields (1999) investigated particle formation in two adjacent, identical offices: one office was used as a control whilst ozone and limonene were added to the other. It was found that particle production was significantly enhanced in the office with the ozone and limonene sources, particularly for very fine particles in the 0.1 - 0.2 µm diameter range. A lag (~30 minutes) was observed between peak ozone and particle concentrations, which reflected the time required for primary and secondary reactions in the ozone-limonene system to occur.

Sarwar *et al*. (2003) observed rapid fine particle growth following ozonisation of alpha-pinene, when a burst of particles was noted in the 0.02 - 1 µm range, the number of which decreased and then attained a steady concentration. The same happened in subsequent size ranges and continued to particles with diameters up to 0.5 - 0.7 µm, creating an effective particle growth wave. The secondary particle mass increased substantially as the air exchange rate was lowered and the residence time increased.

The nature of the particles formed from the oxidation of organic precursors has been extensively studied from the perspective of outdoor aerosols (Heald and Kroll, 2020), which has included ‘smog chamber’ and flow tube studies at a number of facilities worldwide, but care must be taken when interpreting the results. Typical experiments are conducted with higher precursor and oxidant concentrations and lower NO_x concentrations than is usually found, and the walls of the chamber present a surface that the gases and particles can interact with. While surface processes are important in the indoor environment (see section 5.2), the materials used in chambers and flow tubes (typically PTFE and glass) are likely not a perfect analogue. When comparing indoor and outdoor secondary organic aerosol systems, it has been suggested through modelling studies that peroxide and nitrated groups could be more important components of the SOA mass indoors (Carslaw *et al*., 2012). However, such predictions need verification through direct measurement under realistic indoor conditions. These processes are discussed in more detail in section 5.1.

Temperature can also have a bearing on the equilibrium between gas- and particle-phases, something which becomes important when considering air moving between the indoor and outdoor environments at different temperatures. Particle formation is more efficient at lower temperatures as volatilities tend to decrease, but also at higher humidity levels when water uptake into aerosols occurs more efficiently (Youssefi and Waring, 2015). Given indoor temperatures and humidity are typically different to the outdoor conditions considered in atmospheric chemistry experiments, there is likely to be an impact on particle formation as outdoor air moves indoors.

It is worth noting the role of ventilation on the nature of indoor particles. At high ventilation rates, particle composition is more likely to represent that found outdoors, albeit potentially modified by indoor processing. At low ventilation rates, the SOA formed indoors through chemistry will become a more dominant component of the indoor particle mixture (Carslaw *et al*., 2012; Sarwar *et al*., 2003). This is because the longer residence times at lower ventilation rates allow particles more time to form through increased chemical reaction time, as well as more time to accrete organic material and grow in size (Weschler and Shields, 2003). This raises an interesting intervention dilemma given we have very little idea about the relative toxicity of indoor and outdoor particle mixtures.

3.1.5 Biological aerosols found indoors

Our indoor environment is an ecologically complex environment that is, on the one hand, distinct to that found outdoors whilst, on the other, also contains the elements - oxygen, heat, moisture and organic matter – that support the growth of various biota leading to the formation of specific ecological niches. Consequently, indoor environments can directly produce a mixture of bioaerosols, the composition of which owes to the ecological characteristics of the building construction and materials, its inhabitants and purpose of use. Indoor bioaerosols may also be imported from outdoors including pollens, spores and other aerosolised microbes.

Bioaerosols of consequence that have the potential to be generated indoors include pathogenic and nonpathogenic live and dead microorganisms. These include viruses (e.g. SARS-CoV-19), prokaryotes (e.g. bacteria), and eukaryotes (e.g. fungal moulds). Many microbial bioaerosols are of inhalable size, ranging from as small as 50 nm (viruses), 0.25 to 20 µm for bacteria, and from 1 to 30 µm for fungi. Accordingly, a substantial fraction of microbial bioaerosols are less than 5 µm, and are respirable (Fennelly 2020). In turn, a substantial fraction of human-exhaled aerosols are >1 µm and can undergo substantial airborne transmission along with the viruses that they may contain (Wang *et al.* 2021). Moulds and spore-forming bacteria are of concern due to their high rates of sporulation when conditions are correct, leading to high densities of spores that may have health consequences (Thorne and Heederik 1999; Meheust *et al*. 2014). In addition, submicron particles of fungi, such as fragments of hyphae, may be aerosolized in higher concentrations than spores (Green *et al*. 2006).

Dampness leading to the growth of mould and their bioaerosols is of particular concern owing to their association with allergies, asthma and respiratory disease. Indoor bioaerosols also include contagions that are generated by infected humans and their pets whom may harbour viral (e.g. SARS-CoV-19, influenza) and bacterial (e.g. *Mycobacterium*) infections. Despite the focus on antimicrobial resistance (AMR), there is a gap in our understanding of the extent that indoor conditions may amplify microbes with AMR and that may become aerosolised (Environment Agency 2020). Under conducive conditions, microbial growth indoors results in the production of diverse microbial volatile organic compounds (MVOCs) through their metabolism. These aerosolised compounds include alcohols, aldehydes, ketones, esters, terpenes, and organic sulphur and nitrogen compounds, and may also originate from nonmicrobial sources as detailed in section 1.3 (Table 1.1).

MVOCs have an unattractive odour and also have the capacity to cause health consequences following exposure (Korpi, Jarnberg, and Pasanen 2009). Other microbial-derived compounds of importance to health include bacterial cell-wall endotoxins, mycotoxins produced by fungal moulds and fungal glucan cell-wall compounds (Thorne and Heederik 1999). Exposures to these compounds is generally via ingestion and physical contact, and respiratory risks associated with their inhalation are less well established. Notable exceptions include industries where nut or livestock feed processing occur and where inhalation of potentially carcinogenic fungal mycotoxins (aflatoxin and ochratoxin) in grain-dust exposures may occur (Autrup *et al.* 1993).

Allergenic bioaerosols (aeroallergens) are a significant problem in the indoor environment, especially when occupational exposures are high such as may be found in farms, and where chronic exposure may lead to immune sensitisation. Allergic reactions are an immunological hypersensitivity reaction that can occur following exposure of sensitised individuals. Aeroallergens that are produced by microorganisms include the spores produced by moulds (eg. *Aspergillus, Penicillium* and *Alternaria*) and some bacteria (actinomycetes such as *Thermoactinomyces, Saccharomonospora*). Animal proteins constitute an important class of indoor aeroallergens and are produced by arthropods and pets. House dust mites (*Dermatophagoides*) flourish when conditions are correct and produce allergenic proteins that are aerosolised when the mites die and decay. Common pets that generate aeroallergens include cats, dogs, guineapigs, rabbits and birds. Indoor occupational exposure to various animal aeroallergens needs to be considered, for instance when working with animals indoors in various settings including farms and veterinary practices (Samadi *et al.*, 2013) and animal laboratories (Pacheco *et al.* 2006).

3.1.6 Persistent organic compounds

Persistent organic pollutants (POPs) are compounds which do not break down quickly in the environment or in the human body. They were used for various purposes, with their relatively stable characteristics useful for applications such as stain and water repellence, non-stick coatings, and flame retarding. Some of the products where they have been used include carpets, furniture, clothing and other textiles, electronics, packaging, etc. These compounds can then migrate into dust or air. Concentrations of halogenated semi-volatile organic compounds in the indoor environment typically exceed those outdoors and act as a source of outdoor pollution (Harrad, 2019). Because of their semi-volatile nature, such compounds partition between particle and vapour phases, with the higher molecular weight compounds typically present mainly in particulate form.

Many classes of POPs are mixtures of congeners and can be expressed as concentrations of single congeners or the sum of congeners in a mixture. POPs can include legacy compounds which are no longer commonly used, such as polychlorinated biphenyls (PCBs), along with perfluoroalkyl compounds (PFAS), brominated flame retardants (Polybrominated diphenyl ethers, PBDEs and hexabromocyclododecanes, HBCDs). PBDEs were more widely used in the US than in Europe, however in recent years some of the congeners have been discontinued in production (Harrad *et al*., 2006). Levels of PBDEs in the US and therefore generally higher, however, BDE – 209 have been found to be higher in both the UK and US than in Europe (Harrad *et al*., 2010). Typical total concentrations in indoor and outdoor air appear in Table 3.1 (to be updated).

Table 3.1: Arithmetic mean concentrations of selected persistent organic compounds in UK indoor and outdoor air.

Contaminant	Indoor air concentration /pg m^{-3}	Outdoor air concentration /pg m^{-3}	Reference
PCBs	9000 (n = 20, homes and offices)	310 (n = 25)	Currado and Harrad, 1998
PBDEs	110 (n = 67, homes and offices)	21 (n = 6)	Harrad *et al.,* 2004 Harrad *et al.,* 2006
Hexabromocyclododecane (HBCDD)	250 (n = 33, homes)	37 (n = 5)	Abdallah *et al*., 2008
Perfluorooctane sulfonate (PFOS)	38 (n = 20, homes)	2.3 (n = 10)	Goosey and Harrad, 2012
Perfluorooctanesulfonic acid (PFOA)	52 (n = 20, homes)	3.5 (n = 10)	Goosey and Harrad, 2012

3.2 Gases found indoors

3.2.1 Carbon dioxide

The main indoor CO_2 sources are human and animal respiration, as well as combustion products. In densely occupied spaces with no combustion, CO_2 concentrations are driven primarily by respiration, but CO_2 in indoor air is also affected by concentrations outdoors which are increasing over time, and are also higher in urban areas.

While elevated CO_2 concentrations have been linked with health impacts (Simoni *et al*., 2010, Jacobson *et al.,* 2019) attention usually focuses on CO_2 as a measure of the overall adequacy of ventilation, indicating oxygen availability as well as concentrations of total human bio-effluents with respect to odours. CO_2 is only a good indicator in this respect in densely occupied spaces with no combustion sources (ISO 16814, 2008). Elevated CO_2 concentrations have been linked with diminished attention levels, particularly of office workers (Allen *et al*.,2016) and school children, as well as reduced school attendance (Gaihre *et al*., 2014). Worker performance has also been linked to high concentrations of CO_2 (Gupta *et al*. 2020). In a naturally ventilated office, the median scores were found to be up to 12% higher for performance tasks (numerical and proof reading) conducted at CO_2 concentrations below 1400 ppm, compared to those conducted above 1400 ppm, whereas in the mechanically ventilated office, this threshold was measured to be 1000 ppm (Gupta *et al*., 2020).

Although several investigations associated CO_2 concentrations (below 5000 ppm) with effects on health, others did not and given that CO_2 is commonly linked with other human bio-effluents, which may have effects on health (Zhang *et al*., 2017), it is difficult to say whether CO_2 itself is directly responsible for the health effects observed. As Lowther *et al*. (2021) recommended, investigations should aim to measure and report the variation of confounding factors to allow health scientists to understand whether these are likely to impact the measured health outcomes or not.

CO_2 concentrations are often measured to help control building ventilation. For example, in schools where mechanical ventilation is used the Department for Education requires provision of sufficient outdoor air to achieve CO_2 concentrations < 1000 ppm and < 1500 ppm (1829 and 2744 mg m^{-3}) as a daily-mean (during the occupied period only) and 20-minute mean respectively [Higher values are set when natural ventilation is used.] (EFSA, 2018). Figure 3.3 illustrates how CO_2 concentrations vary within a naturally-ventilated classroom at different times of year, showing the relationship with both occupancy and ventilation. Further discussion of these data and their implications for indoor outdoor air exchange rates are given in Annex A.

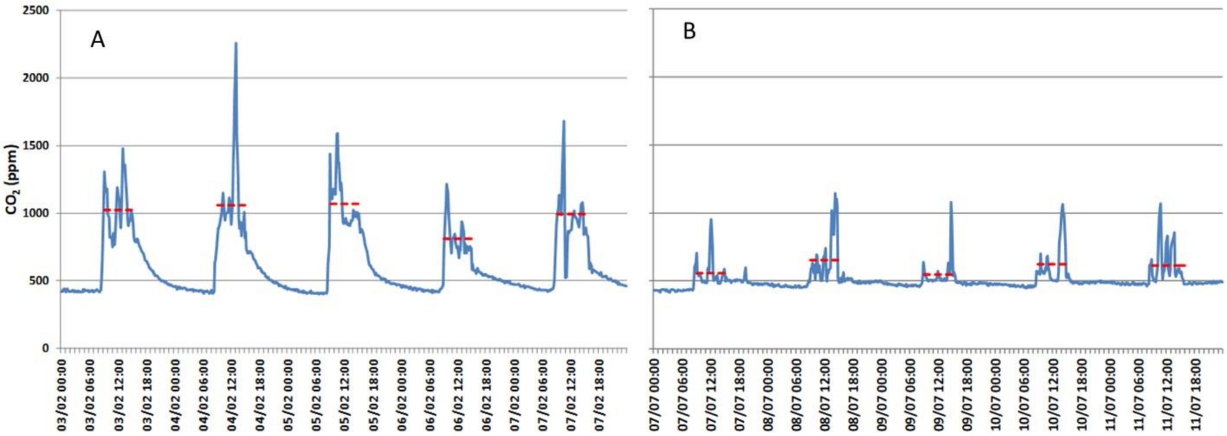

Figure 3.3: CO_2 Concentration in a Naturally-ventilated Secondary School Classroom Over 1 Week in February (A) and 1 Week in July (B). Red dashed lines represent the average occupied concentration. EFSA, 2018

3.2.2 Carbon monoxide

Most indoor CO is produced by the incomplete combustion of fuels in cooking and heating appliances, for example when inadequate ventilation depletes oxygen at the point of combustion. Other sources can include smoking, the burning of candles and the ingress of outdoor air. CO is fatal at high concentrations (PHE, 2016) and concerns have been raised regarding prolonged exposure to lower concentrations (Townsend and Maynard, 2002).

The Building Regulations require that ventilation rates are designed to achieve specified CO concentrations (e.g., Table 2.1) (see also Sections 4.5 and 4.6). A carbon monoxide alarm is required whenever a new or replacement solid fuel appliance is installed, but not for other appliances. Alarms for residential use are fitted with electrochemical cells designed to detect CO concentrations >300 ppm, >100 ppm, and >50 ppm (349, 116, and 58 mg/m^3) over 3 minutes, 10 - 40 minutes, and 60 - 90 minutes respectively (BS EN 50291). A study of CO alarm incidents in local authority homes in Hackney identified 5 incidents per 1,000 homes over 6 months. In nearly 40% of these cases, the alarm was found to be faulty or incorrectly positioned, but a faulty gas appliance was identified in more than 30% of cases (McCann *et al.*, 2013).

A survey of 830 homes in England showed significantly higher 2-week mean CO concentrations in homes with gas cooking than in those without. It also showed significantly higher concentrations during winter than summer. All 2-week mean measurements were <10 mg/m^3 and all monthly-means were <1 mg/m^3 (Raw *et al*., 2004).

3.2.3 Nitrogen oxides

NO_x is the collective term for NO and NO_2, both of which can contribute to elevated NO_2 concentrations. NO_x is formed indoors by combustion, principally for cooking and heating.

Ingress of outdoor air is also an important source of NOx, particularly when natural or mechanical ventilation intakes are close to combustion sources such as road traffic or plant/boiler flues.

Most residential buildings rely on natural ventilation, which tends to be used more in warm weather. Kornartit *et al.* (2010) measured weekly-average indoor and outdoor NO_2 concentrations using diffusion tubes in 60 UK homes. During winter, NO_2 concentrations in the bedrooms, living rooms, and kitchens of homes with gas cookers were all significantly higher than in homes with electric cookers, but significantly lower than concurrent outdoor concentrations. During summer there were no significant differences between either type of home, or between indoor and outdoor concentrations. In residential dwellings with indoor combustion, NO_2 concentrations have frequently been shown to be higher indoors than outdoors (e.g. Garcıa Algar *et al.,* 2004); although in 14 out of 15 published studies collated by Kornartit *et al*. (2010) in which NO_2 concentrations had been measured both inside and outside homes, the overall average reported outdoor concentrations were higher than those indoors.

Concurrent 3-month mean NO_2 measurements made inside and outside 17 nursery schools in London are summarised in Figure 3.4. The outdoor monitors were sited to characterise nearby roadsides, as well as the nursery playgrounds and the area immediately outside the nursery entrance. The indoor monitors were immediately inside the entrance and in a classroom (GLA, 2020a). The authors noted a clear correlation between I/O NO_2 ratios and the perceived airtightness of the building; with "*older, draughty buildings*" being associated with the highest infiltration rates (Walsh, 2020). This was underlined by 2 weeks of real-time measurements shown in Figure 3.5 (GLA, 2020b). In the high infiltration environment, indoor NO_2 concentrations closely track those at the external monitor, while in the low infiltration environment they appear to be driven by the opening of doors for pupil access.

Mean I/O ratios in Figure 3.4 range from 0.3 to 0.8, which is similar to the range of 0.3 to 0.6 (at 99% CI) in London dwellings predicted by Taylor *et al.* (2019) (see Section 3.1.2). In the absence of indoor combustion, I/O ratios for NO_2 are typically smaller than for $PM_{2.5}$ owing to the more rapid indoor losses of NO_2 (Taylor *et al*., 2019; Dimitroulopoulou *et al*., 2006).

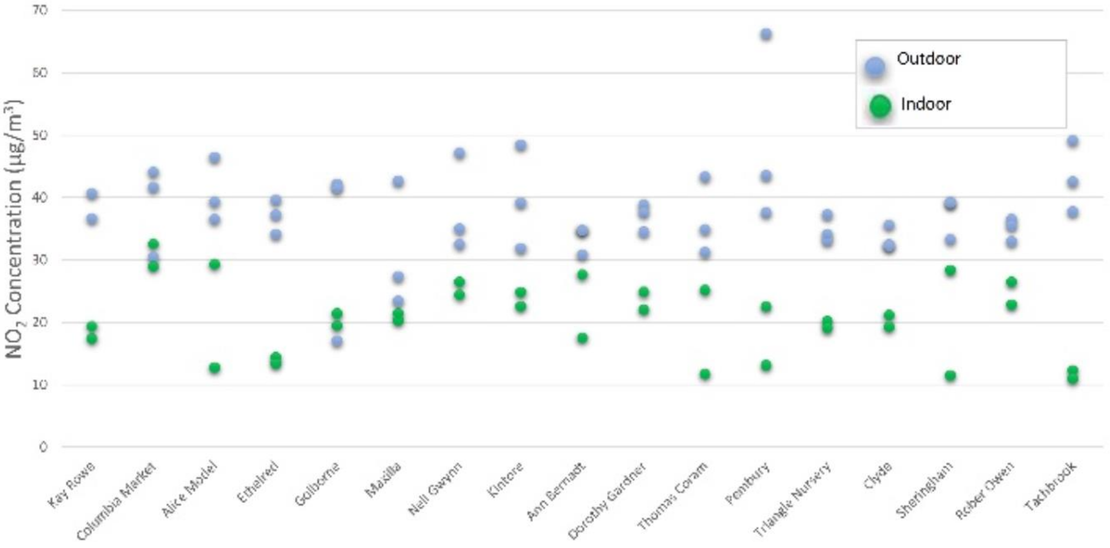

Figure 3.4: 3-Month Mean NO₂ Concentrations Measured Inside and Outside 17 Nursery Schools in London using Passive Diffusion Tubes (Walsh, 2020).

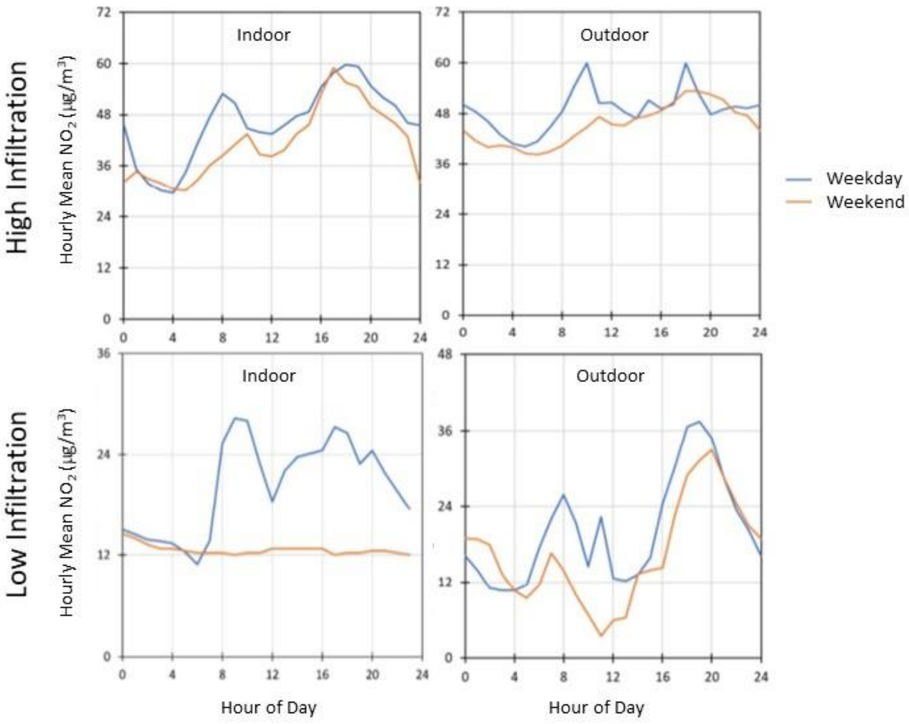

Figure 3.5: Hourly-mean NO₂ Measured over 2-weeks Inside and Outside of Two Nursery Classrooms in London using electrochemical ZEPHYR instruments – Note Variable Y Scale (GLA, 2020b, Walsh, 2020).

3.2.4 Formaldehyde

Formaldehyde is released indoors from various sources such as resins, phenol-formaldehyde and urea formaldehyde (UF) from wood-based products such as particleboard in furniture, UF-based lacquers and UF foam cavity wall insulation. It is also a constituent of tobacco smoke and of combustion gases from fossil fuel burning appliances. Formaldehyde emissions from building materials reduce over time, due to off-gassing, but reaction of other VOCs released from everyday products may also make a notable contribution to indoor concentrations (Wang *et al*., 2017).

A large number of odour detection thresholds have been reported, varying from 50 to 500 μg m^{-3} (WHO, 2010). Three-day mean formaldehyde measurements made in 833 bedrooms in England showed concentrations ranging from 1 to 171 μg m^{-3} (mean 22 μg m^{-3}), with season and building age both having significant effects. The presence of particleboard flooring was also shown to be a key determinant of concentrations. Three-day mean concentrations exceeded 100 μg m^{-3} in 0.7% of homes, most of which were built less than 3 years before the study (Raw et al., 2004). A similar study of Japanese homes observed a linear relationship between formaldehyde concentration and apartment age (Maruo *et al*. 2010). 72-hour mean formaldehyde concentrations measured in three homes in York by Wang *et al.* (2017) ranged from 33 μg m^{-3} to 66 μg/m^3. Three-month mean formaldehyde concentrations measured at 19 nursery schools in London ranged from 4 μg m^{-3} to 25 μg m^{-3} (mean 10 μg m^{-3}) (GLA, 2020a).

Chapter 4 – Exposure indoors

Definition: Exposure to air pollutants is a function of the amount of time a person spends in contact with a particular pollutant and the amount of pollutant present that they may potentially breathe in. A person's exposure to an air pollutant would therefore be the sum of the time spent in each different location they go to (termed a 'microenvironment') and the concentration of air pollutant in each microenvironment. As an example, a person may spend eight hours at their office or other workplace, 12 hours at home, two hours commuting between home and work, and another two hours doing miscellaneous errands. Their exposure for those 24 hours is usually expressed as the average of the concentrations in each of those microenvironments weighted by the time that person spends in them over those 24 hours. The above example is fairly typical of a working adult in the UK, although this will vary between individuals. Thus, indoor environments (assuming work is done indoors) would be a main determinant of a person's exposure – particularly the home and workplace. The term 'exposure' is often used in different ways in the air pollution literature and may refer to an approximate representation of people's exposure to outdoor air pollution on a population level, rather than individual level. Here we will refer to 'personal exposure' to indicate a person's exposure as measured by a monitor that they carry around throughout their daily activities.

4.1 Typical balance of air pollution exposure indoor / outdoor

Overall people in developed countries spend 80 to 90% of the time indoors (Kleipeis *et al*., 2001; Schweizer *et al.,* 2007; Dimitroulopoulou *et al*., 2017), therefore much of their exposure to air pollution will occur indoors. Of the time spent indoors much of it is spent in the home with work or school generally being other important microenvironments from the time activity standpoint. The balance of the contribution of indoor versus outdoor sources to exposure will depend on the ventilation of the building, often represented by the air exchange rates. For substances with primarily outdoor sources, the contribution of outdoor sources will account for the majority of the person's exposure, even if this exposure occurs indoors (Loh *et al*., 2007). The opposite will be true of substances where indoor sources are much larger. This section will discuss the relationship between personal and/or indoor measured concentrations of two pollutants of concern from an outdoor air pollution perspective – NO_2 and $PM_{2.5}$. While indoor exposures are likely to drive other pollutants, especially VOCs, most recent studies published (within the last 12 years) in the UK or Europe, which include personal, indoor, and outdoor measurements have been regarding NO_2 and $PM_{2.5}$.

Studies to understand how concentrations of pollutants in different microenvironments contribute to a person's total personal exposure generally involve a person carrying an air pollution monitor with them throughout their daily activities. These studies may also involve measuring pollutants inside a person's home and outdoors, to evaluate how much their exposure is influenced by and can be approximated by these measurements. However, there are relatively few recent studies that measure personal exposure, indoor and outdoor levels together, particularly in the UK.

A recent review examined the contribution of outdoor sources to personal exposure for $PM_{2.5}$ and NO_2 to better evaluate how representative outdoor measurements are of personal exposures (Evangelopoulos *et al*., 2020). Studies included in this review were primarily from North America and Europe. Personal exposure could be deconstructed into the indoor and outdoor contributed portions using personal time activity patterns and outdoor-to-indoor infiltration factors for specific pollutants. The review estimated that about 44% of personal exposure to $PM_{2.5}$ comes from outdoor sources while 74% of NO_2 exposures comes from outdoor sources.

Individual exposure to nitrogen dioxide is more likely to be influenced by outdoor traffic, as there are few major sources of NO_2 indoors. The exception is for those exposed to gas cookers. Between 2000-2001, a study in Hertfordshire and North London found that weekly average personal exposures to NO_2 were higher in summer than winter (around 13 ppb vs. 8 ppb) and especially in winter, personal and indoor concentrations were higher in homes with gas cookers. NO_2 measured just outside the participants' front doors (13 ppb) were also higher than personal exposures in winter (8 ppb). Within the home, kitchens with gas cookers had the highest concentrations of all microenvironments measured especially in winter (mean concentration in winter 21 ppb vs. 7 ppb in homes with electric cookers). This difference was less pronounced in summer (14 ppb gas cooker kitchens vs. 11 ppb for electric cookers) (Kornartit *et al.,* 2010). These data indicate that, except for homes with gas stoves, NO_2 indoors likely comes primarily from outdoors.

A study from Oxford found that over a 48-hour period, mean personal exposures to NO_2 were 29 $\mu g\ m^{-3}$ compared to $\mu g\ m^{-3}$ inside the home and 27 $\mu g\ m^{-3}$just outside of the home. Concentrations at the participants' workplaces had a mean value of 32 $\mu g\ m^{-3}$ (Lai *et al*. 2004). Using real-time sensors, Delgado-Saborit found mean NO_2 personal exposures of 23 ppb for 16 participants, and a mean outdoor fixed site concentration of 47 ppb for the same period of time. Compared to the previous two studies, the latter study was able to capture more temporal variability of exposure over the day. In this study, participants with gas cookers had higher personal exposure concentrations compared with outdoor concentrations, confirming that gas cookers can elevate personal and likely indoor concentrations of NO_2.

As indicated by the review by Evangelopoulos, particulate matter exposures are more evenly driven by outdoor or indoor sources. A more recent study (2013-2015) in Norwich, UK, Basel, Switzerland, Utrecht, Netherlands, and Turin Italy found that median $PM_{2.5}$ personal exposures were higher in Norwich and Turin (mean 12 and 18 $\mu g\ m^{-3}$ respectively) than Basel and Utrecht (mean 9 and 10 $\mu g\ m^{-3}$ respectively) while residential outdoor $PM_{2.5}$ were of a similar for all cities, except for Turin, where it was higher (van Nunen *et al*. 2020). Mean ultrafine particle personal measurements were around 5000 particles/cm^3 in Norwich, the lowest of all cities. The mean outdoor $PM_{2.5}$ concentration measured just outside the participants' homes for Norwich was 11 µg m^{-3} , only slightly lower than the measured personal exposure concentrations $PM_{2.5}$ personal absorbance (soot) was, however, lowest in Norwich, despite $PM_{2.5}$ personal exposures being slightly higher than in Basel and the Netherlands. For Basel, Utrecht, and Turin, correlations between mean ultrafine personal exposures for adults, residential, and central site UFP number concentration measurements were low, while for $PM_{2.5}$ these relationships varied by city, with higher correlations in Utrecht than the other cities (van Nunen *et al*., 2021). Soot was well correlated across personal exposure, residential, and city centre concentration measurements in all cities but Basel. Mean measurements of $PM_{2.5}$ were below the WHO daily guideline of 25 µg m^{-3} in all three city centres. These data indicate that personal exposure to soot, which was likely from traffic, was driven by outdoor sources, and with $PM_{2.5}$ variability potentially driven by both indoor and outdoor sources. Ultrafine particle personal exposures are likely also driven by both indoor and outdoor sources.

While this report primarily addresses inhalation exposure to pollutants indoors, it is important to recognize that exposures to chemicals indoors can also occur via ingestion (dietary and non-dietary) and, to a lesser extent, dermal exposure. The predominant route of exposure for chemicals will depend on their volatility likelihood of partitioning into different environmental media (e.g. air, particles, other surfaces, water). Ingestion is a notable route for air pollutant exposure and children who often spend long periods on indoor floors and carpet, and these contain higher concentrations of deposited particles. For a recent overview of this topic, see for example Li *et al.* 2019.

4.2 Exposure via commuting and transport hubs

Air pollution concentrations during commuting are often considered in two different ways: by comparing concentrations between different travel modes over comparable journeys or by focusing on understanding the sources in specific micro-environments; train stations and underground systems for example.

4.2.1 Comparing commuting modes

Comparing commuting modes in London - A frequently cited study comparing concentrations in different transport modes is that of Adams *et al*. (2001) who made measurements of $PM_{2.5}$ exposure during 465 multi-modal journeys in London. Lowest concentrations were found during cycling, but these were similar to those in buses and in cars. Far greater concentrations were found in the London Underground, greatly exceeding that of other transport modes. Summer geometric means were 34.5 µg m^{-3} for cycling, 39 µg m^{-3} for buses, 37.7 µg m^{-3} for cars and 247.2 µg m^{-3} for the London Underground. In a similar winter measurement campaign, the geometric mean concentrations were 20.2 µg m^{-3} for cycling 30.9 µg m^{-3} for buses, 23.7 µg m^{-3} for cars and 103.4 µg m^{-3} for the Underground.

Looking within modes, Adams *et al.* (2001) found differences in concentrations between open back "Routemaster" buses and those with closed doors. However, these mainly reflected the routes that these types of buses ran. No significant difference in concentrations was found looking at the two types of buses on the same route. It should be noted however that these measurements were made at a time with greater background $PM_{2.5}$ and greater exhaust emission from traffic than are experienced today. Taking the average of the summer and winter measurement periods, central London $PM_{2.5}$ was 14 µg m^{-3} at background and 31 µg m^{-3} at the roadside during the Adams *et al.* (2001) measurements, compared with 11 and 14 µg m^{-3} respectively at the same locations in 2019 (the most recent year pre-covid). The large decrease in the difference between background and roadside $PM_{2.5}$ concentrations, from 16 µg m^{-3} to 3 µg m^{-3}, is indicative of a decrease in emissions from road traffic and would result in a large change in exposure if the experiment was repeated today, an important issue when comparing these types of measurement campaigns done at different times. These changes are less likely to affect the relative ranking of concentration by mode.

In 2019, as part of a public engagement exercise the environmental campaign group Hubbub worked with King's College London (McAll *et al*. 2019) to determine air pollution exposure profiles of a diverse range of Londoners, revealing when and where individuals were most exposed, then used their personal data and stories as part of a wider communications campaign. Stories from the measurement campaign were published in *The Times*. Ten Londoners, each with different occupations, backgrounds and travel routines, were given micro-aethalometers (AE51) to carry with them for one week. Black carbon (BC) was used as a surrogate for exposure to air pollution from road transport sources. Participants included a school pupil, a college student, a construction worker, an inner-city HGV driver, a Gas Safe engineer, a city cyclist, a doctor, an office worker, a runner and a retired librarian.

Importantly the results from McAll *et al*. (2019) were presented as a cumulative exposure expressed as concentration x time in µg m^{-3} minutes, rather than a simply mean concentrations per mode. Substantial differences were found in exposures. The participants with greatest exposures, 13,000 µg m^{-3} minutes for the long-distance HGV diver and 8,450 µg m^{-3} minutes for a construction worker, had occupational exposures as part of their job. The lowest exposures were 725 µg m^{-3} minutes for the school child. Modern mechanically ventilated buildings had lower concentrations than naturally ventilated buildings. In each case exposure was dominated by the time spent in transport modes (in some cases as part of their work) however this reflects vehicle exhaust as the dominant source of BC in London. The study did not consider other components of the $PM_{2.5}$ mixture.

Commuting modes in European cities - In their review De Nazelle *et al.* (2017) added further cautions when comparing transport exposure studies, especially between places with differing background pollutant concentration, different urban street layout (separate vs on street cycle lanes for instance) and with different transport system and vehicle designs. Their review found ten studies from Europe between 2000 and 2016 that measured $PM_{2.5}$, BC, ultrafine particles (UFP), and/or CO in the walk, bicycle, car and/or bus modes. Eight of these studies were in the first decade of the century, which highlights the need for more contemporaneous studies.

The review found that pedestrians were consistently the least exposed, with the bus, bicycle and car modes on average 1.3 to 1.5 times higher for $PM_{2.5}$; 1.1 to 1.7 times higher for UFP (though this was not statistically significant); and 1.3 to 2.9 times higher for CO. It is only for BC that pedestrians were more exposed than bus users on average (bus to walk ratio 0.8), but they were less exposed than those on bicycles or in cars. Car users tended to be the most exposed (from 2.9 times higher than pedestrians for BC down to similar exposures to cyclists for UFP on average). Bus exposures tended to be similar to that of cyclists (95% CI including 1 for $PM_{2.5}$, CO and BC), except for UFP where they were lower (ratio 0.7). However, these studies do not consider the journey time and the time spent in each mode.

4.2.2. Air pollution in railway stations and in underground railway systems

Few studies have considered air pollution in enclosed railway stations. A larger number of studies have looked at concentrations of airborne PM in underground railways.

Ground-level railway stations - Air pollution concentrations in ground-level railway stations are determined by air pollution from the outside and also sources within the station including trains and catering outlets (Chong *et al*. 2015). Air pollution measurements at UK mainline rail stations have shown that concentrations of NO_2 and $PM_{2.5}$ depend strongly on the frequency of diesel trains and on the station design, with far higher concentrations being measured at the enclosed Birmingham New Street compared with London King's Cross and Edinburgh Waverley.

Font *et al*. (2020) conducted an air pollution measurement campaign at London's King's Cross (KGX) and Edinburgh's Waverley railway stations to characterize the impact of diesel-powered train emissions on concentrations of NO_2, $PM_{2.5}$ and PM_{10} inside the stations. Measurements took place over several months during 2018. The maximum NO_2 concentrations in Edinburgh Waverley were lower than the short-term WHO guideline value threshold of 200 µg m^{-3} defined for outdoor air. This threshold was breached at King's Cross but here the measurement site was closer to stationary trains than in Edinburgh. Concentrations in the outdoor environment had a large influence on the in-station concentrations, as did wind speed and direction.

The average NO_2 concentrations at King's Cross were lower than those measured at Edinburgh Waverley (71.4 and 86.5 µg m^{-3}, respectively) despite urban background NO_2 concentrations being greater in London than in Edinburgh. The mean station increment in NO_2 Edinburgh was 1.7 times higher than that at King's Cross. This was consistent with the greater numbers of diesel trains at the Edinburgh station (~490 trains day^{-1}) compared with KGX (~80 trains day^{-1}). For PM_{10}, average concentrations ranged from 17 to 25 µg m^{-3} across the four inside locations at Edinburgh Waverley and from 18 to 30 µg m^{-3} at King's Cross. For $PM_{2.5}$, concentrations ranged from ~10 to 15 µg m^{-3} at both stations. However, background concentrations of both PM_{10} and $PM_{2.5}$ were slightly greater in London (15 and

12 µg m^{-3}, respectively) than they were in Edinburgh (10 and 7 µg m^{-3}). The study provided clear evidence that diesel-powered trains increase concentrations of NO_2 and $PM_{2.5}$ in enclosed stations to levels that exceed WHO guidelines for their concentrations in ambient air, but no assessment of exposure time was undertaken.

Hickman *et al*. (2018) made air pollution measurements in Birmingham's New Street Station in 2016 and 2017. In contrast to King's Cross and Waverley's largely Victorian station coverings, New Street is far more enclosed. A large, new concourse and shopping centre has been constructed over the twelve platforms to create a tunnel like environment, approximately 5 m high, 160 m wide and 240 m in length making the volume considerably smaller than other enclosed railway stations. Approximately 600 diesel trains use the station daily. The greater numbers of diesel trains and more enclosed space contributed to far greater air pollution concentrations at New Street compared with the other stations. Diffusion tube measurements across two deployment periods showed two-week mean NO_2 concentrations of between 178 to 508 µg m^{-3} at platform level and between 145 and 353 µg m^{-3} in the public concourse / shopping area. Outside concentrations ranged 45 and 89 µg m^{-3}. In the platform area hourly mean concentrations of NO_2 exceeded 200 µg m^{-3} for 49% of the 3888 hours measured. All three locations in New Street with measurements recorded hourly mean concentrations greater than 1,000 µg m^{-3}. Mean $PM_{2.5}$ concentrations were between 29 and 42 µg m^{-3}.

Although of short duration, a measurement campaign at London Paddington by Chong *et al.* (2015) adds to the evidence that stations served by diesel trains have greater concentrations of NO_2, PM and also SO_2 compared with ambient background and roadside locations.

Underground railways - Mean concentrations of NO_2 between 100 and 260 µg m^{-3} have been found in city underground railway systems around the globe (Smith *et al*., 2020). Concentrations of $PM_{2.5}$ on the London Underground were also much greater than ambient concentrations but there was a considerable range of concentrations depending on station and tunnel design.

The first measurements of air quality on the London Underground were made in the late 1800s by Robert Angus Smith who measured concentrations of oxygen in train carriages (mainly first class) with windows opened (Angus Smith, 1872) in what is now part of the Circle Line. A slight decrease, around 1%, was measured compared with concentrations at the surface at a time when carriages were hauled by steam engines. This century, Seaton *et al*. (2005) reported $PM_{2.5}$ concentrations in the London underground as 270 – 480 µg m^{-3}. More recently Smith *et al.* (2020) found mean $PM_{2.5}$ concentrations of 302 µg m^{-3} across 22 journeys compared with a mean concentration of 18 µg m^{-3} measured at a background site in central London and 25 µg m^{-3} alongside a busy road. Concentrations of $PM_{2.5}$ in the underground were therefore 17 times greater than central London background and 12 times greater than at the roadside. Highest concentrations were found in the deeper tunnels of the system and especially furthest from the tunnel portals. Lowest concentrations (in some cases lower than outdoor background) were found in open parts of the system and in the shallower cut and cover tunnels. Accounting for trip duration and frequency, travellers on the underground, are spending an average of between 44 and 57 minutes of their day in an environment where $PM_{2.5}$ concentrations are many times higher than ambient concentrations. It is not possible to compare concentrations in the underground directly between Adams *et al.* (2001) and a later study by Smith *et al*. (2020) since different routes were used, however, the relatively elevated concentrations as compared with other modes are a common finding.

Other underground rail systems in the UK, in Liverpool, Tyne and Wear, Glasgow are of different age and design to the London Underground, but it is likely based on measurements made in many types of similar systems, that $PM_{2.5}$ concentrations would be greater than those at the surface.

From a methodological perspective both Smith *at al*. (2020) and also Nicolosi (2019) found that instrumentation designed to measure ambient air pollution were confounded by the different particle types found in the underground, often with high metal content. Specifically, light absorbing Fe particles and substantial elemental carbon in the coarse size fraction interfered with BC measurement by aethalometer and prevented detection of pyrolyzed carbon in thermo-chemical measurement of organic and elemental carbon. The factory calibration light scattering measurement of $PM_{2.5}$ against a standard Arizona road dust was also found to be inappropriate.

4.3 Air pollution found in retail spaces and sports venues

Other examples of enclosed and indoor space used by the public include retail and sports venues. Measurements from high streets in Dublin (Challoner and Gill, 2014) showed that NO_2 and $PM_{2.5}$ measured in shops depends on the ventilation system. Concentrations were influenced by traffic emissions from the nearby road and varied between shops, even those in the same streets. The merchandise can also have an impact on air quality inside shops. A study from France (Robert *et al.* 2020) measured VOCs in ten shops. Greatest concentrations were found in a car equipment shop, followed by clothing, shoes, and leather, and DIY outlets. Types of VOC differed between the shops too; the maximum α-pinene concentration was in furniture and DIY outlets, and the greatest heptane concentration was in the car equipment shop. Two VOCs classified as toxic to reproduction were measured: hexane in the car equipment shop and the bazaar, and dimethylformamide in the sports goods shop.

Few studies are available for indoor air quality in retail establishments. Most have been completed in the US, China or other countries. A review by the Zaatari *et al*. (2014) found only two studies in the UK – one of them testing radon and the other VOCs in stores. These were conducted over 20 years ago. Looking across all studies, particulate matter concentrations were higher in studies done in Hong Kong, Taiwan, and Japan compared to the USA, likely due to higher levels of particles outdoors in these countries. Grocery stores appear to have higher particulate concentrations compared to other types of stores with cooking suggested to be the main contributor to these levels. Overall, the main sources of particulate matter in retail stores were identified as outdoor contributions, cleaning, cooking, smoking, and photocopying (for submicron particles).

Concentrations of VOCs across the studies were also variable as might be expected given the different types of sources in different stores, ventilation, and locations relative to traffic sources. The Kim *et al*. study in the UK found mean toluene concentrations of 56.7 µg m^{-3} (SD 29.2) in department stores, which was higher than found in the same study for homes and offices (Kim *et al*., 2001). This was similar to levels found in other studies. Formaldehyde was found to exceed recommended exposure limits for the state of California (7 ppb) in the studies reviewed, with an average concentration of 20 ± 8 ppb across the studies. Home goods, office, and furniture stores were found to be particularly high in formaldehyde, with levels reaching as high as 67 ppb in one store (Loh *et al*., 2006; Nirlo *et al*., 2014). Acetaldehyde was found to be higher in grocery stores in the studies as well with levels ranging from 3.2 to 92 ppb in the Nirlo study likely due to baking.

Tetrachloroethylene can be high in stores with dry-cleaning or dry-cleaning clothing - a study by Eklund *at al.* (2008) found mean concentrations of 2540 µg m^{-3} in a clothes rental store and 1650 µg m^{-3} in a dry cleaner’s (median levels 568 and 692 µg m-3 respectively). Overall, exposure to VOCs in retail environments will vary according to the type of store and related sources. Sources will be similar to those identified in the previous section, but one might expect higher concentrations in retail stores, especially where many emitting items are stored such as furniture or hardware stores. It is difficult to say with certainty what might be encountered in UK stores as most data comes from other countries, however one might expect broad similarities with other studies done in European countries or the USA.

Watching sport is an important pastime for many with total attendance (BBC – 19th May 2019) at Championship, League One and League Two football clubs was over 18 million the 2018-2019 season. Very little is known about air quality in these environments, the only recorded studying being measurements of PM composition (Faber *et al.* 2013) made using aerosol mass spectrometry at a German Bundesliga match in the Coface Arena (Mainz, Germany). This highlighted elevated PM exposure from smoke from cigarettes, flares and fireworks, along with the cooking of match-time food.

4.4 Indoor exposure and locations with vulnerable populations

This section considers three kinds of vulnerable populations and indoor air pollution relevant to the locations where they spend significant time: children in pre-schools/nurseries and schools; patients in hospitals and residents in residential/nursing care homes. These vulnerable groups are more acutely affected by poor air quality than the wider population and therefore warrant special consideration here.

4.4.1 Nurseries and schools

Studies investigating indoor air quality in nurseries and schools have tended to focus on temperature and CO_2 concentration as input parameters, and pupil (and occasionally teacher) perceptions of thermal comfort, air quality and fatigue as outputs. Nursery rooms and classrooms in the UK are in the vast majority likely to be naturally ventilated, sometimes with opportunities for cross ventilation, but often with only single-sided ventilation. A recent study found pupils' perception of air sensation was related to CO_2 levels during the non-heating season and to operative temperatures during the heating season (Korsavi *et al*., 2020). Although many UK classrooms may have the potential to provide sufficient window-ventilation, windows often remain closed during the heating season. Indeed, many windows – particularly in classrooms above ground-floor level – have restricted openings to protect children from falls. Furthermore, pupils and teachers often remain in their classrooms for extended periods of time, particularly for younger age groups in nurseries and primary schools where a class and their teacher will often spend the whole day in one room. Consequently, they may not be aware of rising CO_2 levels until a visitor to their classroom points it out.

Children spend about 30% of their time in schools, which makes this the second most important microenvironment for their exposures after the home, on a time-activity basis. Pre-school children may spend even longer than this at nurseries which often provide care from early morning breakfast time until 6pm or later. Particulate matter and traffic-source pollutants such as NO_2 and BTEX (benzene, toluene ethyl benzene and xylenes) in schools are strongly influenced by outdoor sources (Morawska *et al*., 2017; Salthammer *et al*., 2016), although sources of NO_2 such as gas heaters and laboratory burners (e.g. Bunsen burners) can lead to higher levels indoors (Salonen *et al.,* 2019). Branco *et al.*'s 2019 study of 101 rooms in 25 nurseries and primary schools found a majority had environments that were thermally uncomfortable and had high levels of VOCs and $PM_{2.5}$ (Branco *et al.* 2019).

The location of nurseries and schools and their ventilation systems therefore play important roles in determining indoor concentrations of these pollutants. Those located in rural areas or with lower traffic density tend to have lower concentrations than those located in urban areas or areas of high traffic (Salonen *et al*., 2019). Many VOCS, as with homes, tend to have I/O ratios >1, indicating indoor sources (Salthammer *et al*., 2016). Sources of VOCs and SVOCs, such as phthalates and flame retardants include building materials, furniture, cleaning products, and craft materials (see section 1.1).

Appropriate ventilation in nurseries and schools now needs to take into consideration minimising the risk of aerosol transmission of viruses such as SARS-CoV-2 (Health and Safety Executive, 2020). Ensuring that these settings meet adequate ventilation standards to reduce the risk of viral transmission is important owing to the potential impact that nurseries and schools can have in accelerating or perpetuating outbreaks of respiratory infections and onward vectoring of infection to older and vulnerable members of the community (European Centre for Disease Prevention and Control, 2021).

4.4.2 Hospitals

There are around 160,000 NHS hospital beds in the UK (Statista, 2021), with quarterly hospital admissions of almost 1.49 million and an average stay of six days (ibid). This is an inherently vulnerable population, not only dealing with their primary medical needs, but also risking secondary infections and other negative health consequences from being in hospital for days and weeks at a time. A study of 2016-17 data estimated that there were 834,000 cases of healthcare-associated infections annually, resulting in 28,500 deaths and costing the NHS £2.7 billion (Guest *et al*., 2019). (It should be noted that these statistics and study are based on pre-Covid data and may have changed to a greater or lesser degree at the time of writing, although the long-term effect of Covid-19 on these figures remains unclear). A study based around a 1970's designed hospital in Cambridge, typical of many hospital designs from that period, found overheating to be a current concern within the wards which would only increase in future climate scenarios if preventative measures (such as additional shading, insulation, advanced natural ventilation promoting downdraught cooling) were not taken (Short *et al*., 2015). Ventilative measures (e.g. fans, mechanical ventilation, air conditioning) to mitigate against high temperatures in hospitals are hampered by the potential that they may spread airborne infections from one bed to another, and from one ward to another, and therefore need to be designed, installed and operated carefully to minimise this risk.

4.4.3 Care homes

A similar and often overlapping/interchangeable vulnerable sector of the UK population are those living in residential/nursing care accommodation. The UK has a current care home population of over 490,000, with more than 83% aged 65 and over (Carehome.co.uk, no date). This sector of society has been shown to be particularly vulnerable to adverse indoor environmental conditions, particularly high temperatures (Hajat *et al*., 2007). Residential care homes have a reputation for being warm all year round. Indeed, this is a key consideration in how they are designed and operated (Gupta *et al*., 2021b). However, with the UK experiencing more prolonged and intense periods of high temperatures, particularly during the summer, overheating has been found to be prevalent within care homes, particularly in bedrooms. The vast majority of the UK's 17,500 residential and nursing care homes are naturally ventilated, but ventilation is severely hampered by the regulated 10 cm limit on window opening issued to protect residents, many of whom have some form of dementia. Consequently, care homes have been found to also have high concentrations of CO_2, particularly overnight in bedrooms. They also often have low RH – partly due to the prevalence of "always on" heating systems throughout much of the year (ibid.).

In common with schools, hospitals can act as foci for the transmission of a wide range of contagious bioaerosols. Specifically, those who suffer from respiratory disease such as cystic fibrosis, or have active bacterial and viral infections, can expectorate high burdens of microbes that may include those that have acquired resistance to antimicrobial drugs. As vulnerable populations mix in hospitals, then it is important that infection prevention and control recommendations are closely followed in order to minimise nosocomial exposures. These may include clinical triage and separation of patients with different risk-factors, isolating patients with contagious infections, appropriate ventilation in wards, respiratory hygiene and cough etiquette (Catch it, Bin it, Kill it), and wearing appropriate personal protective equipment (PHE, 2016)

In summary, vulnerable populations in schools, hospitals and care homes face many common threats to their health in relation to indoor air quality, and a constellation of related effects including high indoor temperatures, high concentrations of CO_2 (often associated with other indoor air pollutants) and low RH.

4.5 The role of indoor / outdoor air exchange

Sources determining the total air pollution indoors comprise both indoor sources and the ingress of outdoor pollutant into the indoor environment. Sinks of indoor air pollution include loss through windows or ventilation systems and other leakage and deposition. Chemical reactions can act as both sources and sinks. Indoor air pollutant concentrations are determined by the magnitude and location of these sources and sinks and the airflow and turbulent mixing in the indoor environment, which determines the distribution of the pollution (see Appendix 1). Indoor sources have been discussed in Chapter 2; this section focusses on air exchange at the indoor/outdoor interface and the flow, turbulence and mixing within buildings.

4.5.1 Indoor - outdoor air exchange

The rate of ventilation or air exchange rates for a building or room is often expressed in ‘air changes per hour’ (ACH); this is the ratio of the volume of air exchanged per hour to the volume of the building or room. Alternative descriptors use the volume of air exchanged per second, or sometimes the volume of exchanged per second per person when CO_2 is the principal pollutant being considered.

There have been numerous experimental studies of air exchange rates in houses and in other larger buildings such as hospitals. The following provides some examples of measured and recommended air exchange rates; these are summarised in Table 4.1. In early studies relevant to older houses without any mechanical ventilation, the focus was on reducing pollutant concentrations from indoor sources; these houses typically showed high exchange rates of greater than one air change per hour with the exchange rate increasing with higher external wind speeds, when outdoor air pollution is typically lower. Dick and Thomas (1951) provided empirical expressions for dependence of ventilation rate based on the number of open vents, and wind speed for a more exposed site, and temperature difference for a more sheltered site.

Recent studies (*e.g*. MHCLG, 2019) have looked at the impacts of both mechanical and natural ventilation including use of windows and trickle ventilation. These studies are mostly relevant to newer construction where buildings are increasingly airtight. Average exchange rates were 0.37 air changes per hour, the rates being strongly influenced by extent to which windows were opened. This had more influence than trickle ventilation or mechanical ventilation. In these studies, a conflict was noted between increased ventilation and increased external noise and, for mechanical ventilation, the noise of fans. In 37 newer builds Crump *et al.* (2005) measured mean exchange rates of 0.44 (winter) and 0.62 (summer) air changes per hour. Raw *et al.* (2004) measured indoor air pollution in 876 houses and concluded that most indoor air pollution came from internal sources implying low exchange rates. Johnston and Stafford (2017) found exchange rates of less than 0.4 per hour in 4 newly built houses which is somewhat lower than exchange rates implied by ventilation rates required in the Standard Assessment Procedure (SAP) for new dwellings of about 0.6 air changes per hour (MHCLG, 2010). SAP includes an assumption that new dwellings have an air change rate of 0.15 per hour with no ventilation (i.e. only infiltration). For industrial/commercial premises CIBSE (2015) recommends between 5 and 15 air changes per hour for ‘typical applications’.

Data from other countries is broadly consistent with the above. The German PassivHaus standard, consistent with ultra-low energy buildings (ATTMA, 2010), limits air exchange rates to less than 0.6 per hour for new-builds. The PassivHaus standard for refurbishment of existing buildings (EnerPHIT) increases the air exchange rates slightly to 1.0 per hour. In the USA, Wallace *et al.* (2002) measured exchange rates in one house over a year and found a mean rate of 0.65 per hour with much higher rates when windows were open, but relatively little dependence on indoor outdoor temperature difference or windspeed. In China, Yan You *et al.* (2012) observed a range in exchange rates from 0.12 to 3.46 per hour depending on season and use of windows. In Peru, Escombe *et al.* (2019) found high exchange rates (>2.7) in hospital waiting and consulting rooms, more especially if mechanical ventilation was optimised or if numerous windows were open (>12).

Table 4.1. Examples of measured and recommended air exchange rates.

Buildings	**Air Changes Per Hour**	**Source Literature**
Older houses measurements	<2 to >3 depending on windows, ventilation and weather conditions	Dick and Thomas (1951)
10 new houses measurements	0.19 - 0.69, mean 0.37; strongly influenced by use of windows	MHCLG (2019)
37 newer houses measurements	0.19-0.68, mean 0.44 (winter) 0.19-1.06, mean 0.62 (summer)	Crump *et al.* (2005)
4 new houses measurements	0.2 - 0.34	Johnston and Stafford (2017)
New houses – building regulations for ventilation	minimum ~0.6 implied[1] for determining ventilation rate	MHCLG (2010)
PassivHaus Standard	<0.6 (new build) <1 (refurbished)	ATTMA, 2010

Office	minimum ~1.3 implied[2] by required ventilation rate	MHCLG (2010)
Commercial/Industrial recommendations	5-15 recommended for ‘typical applications’	CIBSE (2015)
One House (USA) measurements	~0.65 (higher if windows open)	Wallace *et al.* 2002
Homes, schools (China) measurements	Homes 0.12 - 3.46 (median 1.15) Schools 0.73 - 1.91	Yan You *et al.* (2012)
Hospital waiting and consulting rooms (Peru) measurements	Increased from 2.7-11 to 12-66 with improved ventilation	Escombe *et al.* (2019

[1] Approved Document F (MHCLG 2010) states whole dwelling ventilation rates for a three-bedroom house should be at least 31 l/s (see Table 2.1 of this report). For typical house volume ~180m3, this implies an air exchange rate of 0.6 exchanges per hour.

[2] Approved Document F (MHCLG 2010) for an office requires at least 1 litre per second ventilation per metres square floor area. This implies 1,3 air exchanges per hour for an office of height 2.7m.

4.5.2 Factors affecting transport and mixing of indoor pollutants

Exchange of air can be through ventilation (controlled exchange of air through windows, doors and ventilation systems) or air leakage (uncontrolled exchange of air through cracks, porosity or other unintended openings). Prior to 2006, buildings tended to be designed and built with little thought towards airtightness, primarily because there was no requirement for testing. Consequently, the legacy building stock in the UK is considered quite ‘leaky’ (Coxon, 2013). Making buildings more airtight may be an effective measure to reduce heat loss, but can lead to unintended consequences. A study of six dwellings – all designed for Code for Sustainable Homes Levels 4 and 5 – found issues of overheating and poor indoor air quality caused by interrelated socio-technical factors (Gupta and Kapsali, 2016). Poor indoor air quality was linked with insufficient MVHR air supply due to inadequate commissioning and lack of user comprehension in operating the systems.

A 2019 report from the Building Research Establishment (BRE) illustrated examples of typical external and internal pollutants found in buildings and also examples of common sources of these pollutants (Figure 4.1). For buildings that have poor airtightness, externally sourced pollutants can enter easily, and pollutants that originate indoors can escape. But as buildings become more airtight, through retrofits or new constructions, pollutants originating outdoors may be more effectively kept out whereas pollutants originating indoors may be trapped inside. This has the potential to significantly change the concentrations of different pollutants found inside buildings. Reducing ventilation may also lead to the build-up of condensation and humidity, leading to levels of humidity that encourage microbial growth including moulds and mildews. Such microbial overgrowth is destructive to the fabric of the building as well as generating potentially harmful bioaerosols. In such cases additional measures to keep relative humidity at levels below 75% may be needed.

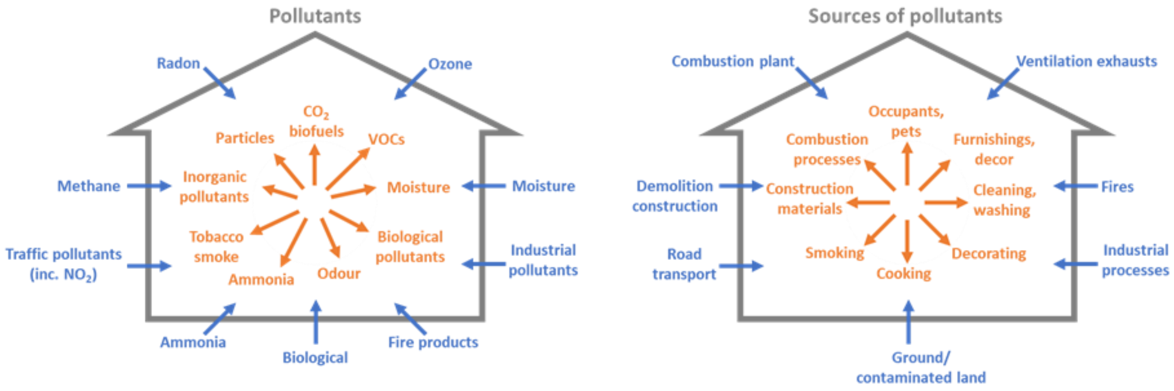

Figure 4.1: Examples of typical external (blue) and internal (orange) pollutants found indoors (left) and sources of pollutants affecting IAQ (right) (adapted from Kukadia and Upton, 2019).

Mechanical ventilation will therefore be increasingly important to effectively manage indoor air in new buildings. It not only needs to be designed well, and installed well, but also commissioned adequately and operated well by the building's users. Failure in any of these aspects could lead to reduced air exchange and increases in indoor pollutants.

The relationship between indoor air quality, ventilation and human factors are not well understood (Taylor and Morgan, 2011) (Zero Carbon Hub/ NHBC Foundation, 2013). A 2011 report by the Good Homes Alliance found that in new-build homes, even ones designed to be low energy, commissioning of ventilation systems often takes place before the building is finished, and with monitoring rare, many buildings could suffer from undiagnosed ventilation and air quality problems (Taylor and Morgan, 2011).

Poorly commissioned or faulty ventilation systems likely result in poor air quality since stale indoor air is not replaced at a sufficient rate by fresh outdoor air. Emissions from building materials and combustion products will adversely affect the indoor air quality in air-tight homes (Yu and Kim, 2012). In addition to a build-up of pollutants, humidity and condensation are likely to increase, leading to mould growth, damage to the building fabric proliferation of house dust mites (Crump *et al*., 2009). Unfortunately, the limited studies of commissioned MVHR systems have found failures in typical design, installation and commissioning practice to be “all too common” (Zero Carbon Hub/ NHBC Foundation, 2013). A two-year research project conducted by the Building Research Establishment (BRE) assessed and monitored ten zero carbon Code for Sustainable Homes Code Level 6 at a development in Berkshire (Dengel and Swainson, 2013.). The homes were studied during construction and for almost two years post-occupancy, but after one year of occupation, nine of the MVHR systems needed to be recommissioned, highlighting a lack of expert installation and initial commissioning and the importance of occupant feedback during early occupation.

A 2016 study commissioned by Innovate UK as part of the Building Performance Evaluation (BPE) Programme investigated MVHR systems in 85 dwellings from 29 projects (Sharpe *et al*., 2016). Developments ranged from single homes to 700+ dwellings, so the study potentially represented in excess of 3300 dwellings. The study found only 16% of systems to have been commissioned correctly, and only 56% of installations meeting the design air flow value. The study concluded that when MVHR systems are well designed, installed, maintained and used, they can be effective in reducing energy consumption and providing good ventilation. However, in practice this is often not achieved due to a combination of poor design, poor installation, poor (if any) occupant handover, and inadequate maintenance. As an illustration of how poor occupant understanding of the MVHR systems can be a problem, half of the projects sampled in this study had occupants who had disabled the system, most commonly due to concern that the running cost would be high.

A more recent report from the Ministry of Housing, Communities and Local Government (MHCLG, 2019) looked at 80 homes – 55 naturally ventilated, 25 mechanically ventilated – across seven developments in England. The study found that only two of the naturally ventilated homes met Approved Document F’s guidance for trickle-vent provision and intermittent extract fan air flow rates – a quarter of fans tested providing less than half of the recommended flow rate. In homes with continuous mechanical extract, only one met Approved Document F’s guidance with respect to flow rates and trickle-vent provision. Many of the homes in the study had problems with damp and mould, and 60% had levels of TVOCs in excess of the performance standard in Approved Document F. Analysis suggested that if the extract fans and mechanical ventilation systems able to deliver the recommended air flow rates, IAQ levels would have been significantly improved.

4.5.3 Parameterisations of exchange rates used in models

Models investigating indoor air quality in buildings need to consider many different parameters in order to provide as accurate a representation of reality as possible. These parameters may be grouped into three categories: building characteristics, climate parameters and human factors such as occupancy pattern, activities etc.

The UK Government’s preferred method to assess and compare the energy and environmental performance of dwellings is SAP – Standard Assessment Procedure (BEIS, 2014). This considers energy used for space heating, water heating, heat pumps, fans and lighting, but ignores energy used in cooking and small appliances, and assumes a standard occupancy profile regardless of a building’s real-world occupancy. It uses standardised procedures for estimating air exchange rates for windows, trickle ventilation, mechanical ventilation etc.

EnergyPlus is an energy analysis and thermal load simulation programme used mainly by building engineers (Energy Plus, 2021). Using an all-building model it takes inputs on geometry, construction materials, HVAC (heating, ventilation and cooling), usage and systems, and calculates heating and cooling loads necessary to maintain thermal comfort and the energy consumption of primary plant equipment.

The Passivhaus Planning Package is a software programme developed by the Passive House Institute which prides itself on being able to predict comfort and energy consumption very accurately for Passivhaus and low-energy buildings (Passivhaus Institute, 2021). It requires detailed, accurate inputs in four categories: Verification (general project information); Heating (inputs include detailed building description, climate, U-values for heat transfer through walls, windows, shading, ventilation) – this is where much of the architectural design features are entered; Cooling (inputs include details about MVHR operation, window opening patterns); and Primary Energy (inputs relating to domestic hot water, PV electricity, electrical devices within the home, internal heat gains and other systems). It is increasingly recognised that in these modelling tools there are often significant discrepancies between the predicted and actual performance due to a number of reasons including modelling assumptions about occupant behaviour related to the use of ventilation. The consideration of indoor environmental conditions in these models tends to be focussed on operative temperatures, CO_2 concentrations and relative humidity with limited input on indoor pollutants such as $PM_{2.5}$ and VOCs.

In a study of a fabric-first deep retrofit of a low-rise block of flats in the UK, a Passivhaus Planning Package (PHPP) energy model was used to predict pre- and post-retrofit energy and comfort (indoor temperature) (Gupta and Howard, 2021). Pre-retrofit environmental monitoring (temperature, RH and CO_2), energy monitoring, fabric tests (U-value and air tightness) and occupant surveys (detailing number of occupants, occupancy patterns and energy-related behaviours) were used to refine and calibrate the PHPP model and also inform the design of certain retrofit features. These included the re-sizing of MVHR, and having openable windows to appease occupant behaviours and expectations. The project was conducted before and during the Covid-19 period, resulting in significant changes in the number of occupants in the flats and occupancy patterns. The results highlighted the importance of using empirical data to help calibrate models, and the need to consider occupant-related factors within any models.

4.6 Ventilation, planning and indoor air quality

New developments close to roads (and less commonly other emission sources) are frequently granted planning permission on the basis that a system of mechanical ventilation will be in place, drawing air from the rooftop or rear of the building and away from the road emission source. The requirement for mechanical ventilation usually stems from other design considerations (for example heating, cooling and controlling humidity), but can be driven by air quality concerns; most commonly for residential dwellings and schools, where the annual mean NO_2 objective applies (Defra, 2018).

A need for mechanical ventilation will typically be identified as part of the Standard Assessment Procedure (SAP) calculations for a new development, carried out under Part L of the Building Regulations (MHCLG, 2010). This considers energy consumption, principally in relation to thermal comfort. Ventilation for new residential dwellings will often be specified using the Chartered Institution of Building Services Engineers (CIBSE)'s TM59/TM52 design methodologies for thermal comfort. Even where a specific need for air quality mitigation has been identified, the engineering solution will usually be based on thermal requirements.

There is a distinction between centralised and individual mechanical ventilation systems. Commercial and educational developments typically include centralised systems, while individual systems are preferred for residential use. Either system might include an element of both natural and mechanical ventilation. Space for the required ducting for mechanical systems presents a major constraint, particularly for residential units, where delivering roof-level air to multiple privately-owned apartments is a significant challenge. Ventilation from the nearest façade is thus most common for residential apartments.

For schools, detailed guidelines on ventilation are available which include advice on minimising indoor air pollution (ESFA, 2018). More broadly, Part F of the Building Regulations specifies the ventilation requirements for new and existing buildings, including residential dwellings. This includes a performance criterion for long-term average NO_2 concentrations which mirrors the annual mean objective. Despite this, it is not uncommon for planning permission for a new development to be granted on the basis that façade-level air will not be drawn into the building while the final building fails to meet this requirement; either because ventilation specifications agreed at the planning stage are misunderstood, or because they are later disregarded. Changes to planning rules in 2015 have also allowed the creation of new residential dwellings from non-residential use without local planning oversight.

Mechanical ventilation systems may be specified either with or without the sealing of windows. Where windows are sealed, alternative façade ventilation often remains a requirement. Amenity concerns have been raised historically (Planning Appeal Decision APP/E5330/A/12/2178469) regarding sealing of residential windows, but such a design has become common in urban areas.

Centralised mechanical ventilation systems are often driven by rooftop Air Handling Units (AHUs). An important issue can be proximity to exhaust flues from combustion plant. Part J of the Building Regulations defines acceptable release locations for small appliances, but emissions of NO_x and PM from centralised plant, or the mandated testing of back-up generators, can be appreciable and are not addressed within the Building Regulations Approved Documents (MHCLG, 2010). Large AHUs typically project at least 1 metre above rooftops, which is comparable with the height of many combustion flues. Available roof space is often limited, and it is not uncommon for buildings to be designed with both the exhaust from large combustion plant and the AHU for the ventilation system on the same roof or on the same façade. Similarly, back-up generators can be designed to emit from ground level façades close to windows or other air intakes.

Ventilation systems are increasingly sophisticated and may include, for example, automatically actuated windows linked to an array of sensors, most commonly for temperature and indoor CO_2 concentrations. Some ventilation systems have also been coordinated by electrochemical NO_x sensors, the intention being to use ambient NO_x measurements to better time the scheduled purging of internal air. This would provide significant potential benefits to indoor NO_x concentrations (Mills, 2018), but the use of such systems is not widespread, and electrochemical NO_x sensors are currently not sufficiently accurate or reliable. Using ambient NO_x measurements to time the cessation of external air provision to rooms has also been proposed, relying on the building as a reservoir of ‘fresh air’ for short periods. There is currently a lack of evidence to support a reliance on NO_x sensors for building management applications where measurement accuracy is critical, however the technology is developing, and this approach may become more robust in the future.

Increasingly, mechanical systems are specified to include air filtration specifically to mitigate elevated external NO_2 concentrations. Filtration of PM remains more common but is not generally a specific planning requirement. A variety of proprietary systems have been used to remove NO_2, typically based on activated carbon media. Available independent evidence suggests high efficiency (e.g. 90 - 98% NO_2 removal (Upton, 2016)) but continued ongoing maintenance is likely to be critical.

Overall building design, including location, layout and shape, also all have roles to play in controlling the effects of external pollutant emissions on indoor air.

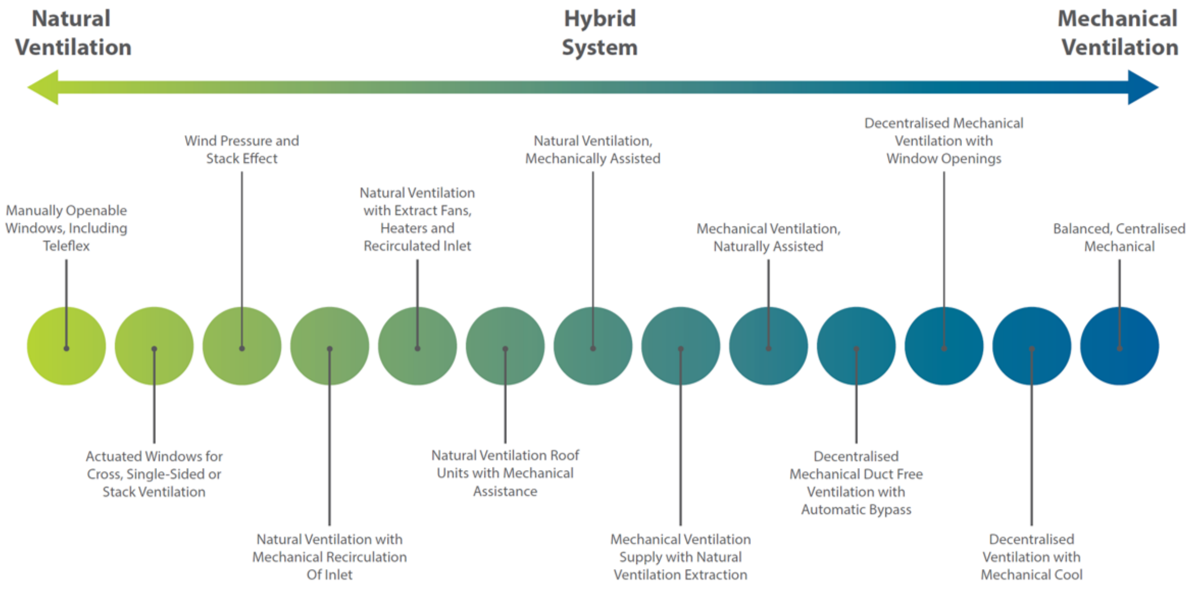

Figure 4.2: Types of Ventilation System (from EFSA, 2018).

Chapter 5 – Indoor air processes

5.1 Overview of indoor air chemistry processes

Despite the low light levels indoors compared to outdoors, there is a vast array of gas-phase chemistry that can happen under typical indoor conditions, with many of these reactions similar to those in the ambient atmosphere. Differences between conditions indoors and outdoors, mean that indoor reactions can be more or less important than their outdoor counterparts, but just like outdoors, much of the indoor gas-phase chemistry is driven by oxidation reactions, particularly with ozone (O_3) and hydroxyl radicals (OH) and to a lesser extent by nitrate radicals (NO_3) and chlorine (Cl) radicals.

Unlike outdoors where the hydroxyl radical (OH) tends to dominate oxidation of VOCs, the lower light levels indoors mean that the photochemical reactions that lead to OH production are less favoured and VOC oxidation by ozone becomes relatively more important. Ozone concentrations indoors are much lower than outdoors, as it rapidly deposits on indoor surfaces and also reacts with nitric oxide (NO). Just like outdoors, there are seasonal and latitudinal implications for the importance of ozone concentrations and hence ozone chemistry indoors (Weschler, 2000). High NO_X concentrations reduce those of ozone and tend to peak in the winter months outdoors when NO_X dispersion is reduced. Outdoor ozone concentrations tend to peak in spring/summer (Carslaw, 2005), coinciding with a time of year when windows are typically open more often. Given the major contribution of outdoor ozone to indoor ozone concentrations, the location, time of year and air exchange rate with outdoors are all vitally important in determining the indoor ozone concentration, which is typically 0.2 - 0.7 of that outdoors (Weschler, 2000).

One group of indoor reactions that has received significant attention to date has been ozone-monoterpene reactions indoors (Weschler and Carslaw, 2018). These reactions have been studied in detail for several reasons: (i) monoterpenes are ubiquitous indoors given they are used in large quantities in personal care and cleaning products (Nazaroff and Weschler, 2004); (ii) ozone is also ubiquitous indoors, gaining access from outdoors via windows, doors, and cracks in the building envelope as well as from indoor sources such as photocopiers and laser printers where they exist (Weschler, 2000); and (iii) ozone-monoterpene reactions are fast enough to compete with typical air exchange rates indoors (Weschler and Carslaw, 2018), and can produce a range of short-lived (e.g. radicals) and longer-lived complex multi-functional species in both the gas- and condensed-phases (Walser *et al*., 2007).

Once the hydroxyl radical is formed indoors for example through ozonolysis reactions or indoor photolysis, it can initiate oxidation reactions just as for the outdoor environment, leading to more oxidation chemistry indoors than might otherwise have been expected (Figure 5.1). Reaction rates for the hydroxyl radical with VOCs indoors are much faster than both its deposition onto indoor surfaces and exchange with outdoor air, which are both negligible (Weschler and Shields, 1996). Apart from the photolysis rates, many of the rates of reaction indoors are comparable to outdoors and some can even exceed those typically observed outdoors (Figure 5.1), such as the reactions of ozone and the hydroxyl radical with monoterpenes.

The presence of OH indoors has been confirmed through measurement studies. Gomez-Alvarez *et al*. (2013) found that OH concentrations were elevated (1.8 x 10^6 molecule cm^{-3}) near windows owing to the photolysis of HONO (nitrous acid). Carslaw *et al*. (2017) measured elevated concentrations of OH radicals (4 x 10^6 molecule cm^{-3}) during surface cleaning with a limonene-based cleaner in a University PC classroom. In the same study, they operated an air cleaning device which elevated concentrations even more, peaking at 2 x 10^7 molecule cm^{-3}. Indoor OH concentrations in this study were determined to be 6.5 x 10^5 molecule cm^{-3}, about a factor of 5 -10 less than outdoors in summer and comparable to outdoor concentrations at night-time (Faloona *et al*., 2001) and during the daytime in winter (Heard *et al*., 2004). The enhanced indoor OH concentrations during cleaning are similar to, or even greater than ambient concentrations. Clearly then, OH formation and subsequent OH-driven oxidation is possible for a range of indoor conditions and could be particularly important depending on the activities that are taking place within a building.

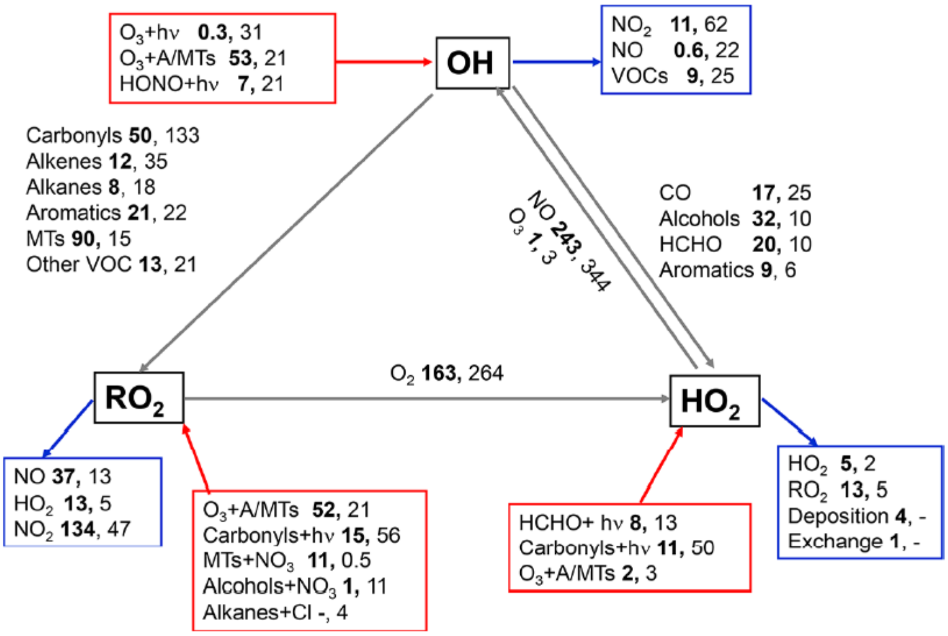

Figure 5.1: Rates of reactions for indoors (bold font) and outdoors (normal font) in units of 10^5 molecule cm^{-3} s^{-1} (after Carslaw and Weschler, 2017).

Note that there have been very few studies of PAN-type species indoors. One such study showed that PAN was formed indoors through chemistry rather than transported from outdoors and that concentrations were enhanced when the ozone concentration was higher (Fischer *et al*., 2014). PAN species are thermally labile and so the indoor temperature will also determine how quickly PAN is degraded. The major fate of HNO_3 indoors depends on the conditions. Deposition onto indoor surfaces and exchange with outdoors are likely to be important (Weschler *et al*., 1992), although partitioning to the aerosol phase to form ammonium nitrate could be important if ammonia concentrations are high indoors (Ampollini *et al*., 2019).

The nitrate (NO_3) radical is another oxidant outdoors, participating in various oxidation reactions. Outdoors, it only becomes important at night-time since during the day it is photolysed rapidly (lifetime of ~5 s), and reacts rapidly with NO. Given the lower light levels indoors, it might be assumed that the nitrate radical could reach higher concentrations within buildings in the absence of combustion sources and hence high NO. The NO_3 radical had not been directly measured indoors until a recent study confirmed its presence for the first time with an experiment that manipulated indoor concentrations (Arata *et al*., 2018). A portable butane stove was used to provide NO_X emissions in a residential kitchen which removed indoor ozone and consequently, no NO_3 was observed. However, when an ozone generator was used to enhance the indoor ozone concentration to ~40 ppb, there was sufficient NO_2 to produce NO_3 at mixing ratios of ~3-4 ppt. Although these concentrations are low, they can still have significant impacts on indoor chemical processing (Carslaw, 2007).

Reactions of the chlorine (Cl) radical are analogous with those of the OH radical. They are generally faster than with OH, and there is also the potential to make chlorinated products through oxidation reactions (such as HCl instead of H_2O). Chlorine radical formation indoors is possible following cleaning activities that liberate various chlorine compounds. For instance, significant concentrations of both chlorine molecules and hypochlorous acid (HOCl) were shown to be formed in the 10s - 100s of ppb range following floor mopping with a bleach cleaner (Wong *et al.,* 2017). Both of these species can be photolysed to liberate chlorine radicals, the latter also producing OH radicals. The level of light indoors is therefore critical in determining the importance of these sources of chlorine radicals. For instance, predicted indoor concentrations of chlorine radicals varied from near zero in the dark to approximately 3×10^5 molecule cm^{-3} if it was assumed that 3% of outdoor UV light and 10% of outdoor visible light were able to penetrate indoors (Wong *et al*., 2017).

It is important to note that ozone-monoterpene reactions can also produce products which are sufficiently involatile that they promote particle formation and growth. Following oxidation, they can produce a wide range of multifunctional species, including alcohol, aldehyde, ketone and carboxylic acid groups: such groups tend to have lower vapour pressures than their parent terpene and can condense to form secondary organic aerosol (Walser *et al*., 2007).

There can be a wealth of gas-phase chemistry indoors, with the dominant reactions determined by the indoor conditions, such as light levels, oxidant and NO_X concentrations, building location, occupant activities and ventilation rates. The numerous controlling factors mean that two identical buildings could have very different indoor pollutant concentrations. A building next to a busy road where vehicle-emitted pollutants were able to infiltrate could have very different indoor air chemistry to one in a rural setting with higher outdoor ozone concentrations. Even the same building could have different indoor air pollutant reactions depending on the time of year, as ozone concentration and window opening frequency varies with season.

It is likely that indoor chemical processing indoors has become more important in recent years, through a combination of increased outdoor ozone concentrations, greater use indoors of terpenes in cleaning products and fragrances, and decreased ventilation rates (Weschler *et al.,* 2006). The international lockdowns accompanying the 2020 COVID-19 pandemic have also allowed a glimpse of how indoor air chemistry might change in the future. The decreased NO_X emissions in urban areas that accompanied various lockdowns could be viewed as a proxy for future improvements in vehicle technologies that aim to decrease vehicle NO_X tailpipe emissions outdoors.

For instance, in the UK, the combination of prevailing meteorology and the lockdown in spring 2020 led to decreases in ambient NO_X concentrations of about 30-40% and ozone increases at some locations. (AQEG, 2020). Model simulations showed that this increase in outdoor ozone led to an increase in indoor ozone concentrations of around 50% for typical residential conditions, with an accompanying 30% increase in HCHO caused by additional chemical reactions indoors. Further, the future expected improvement in vehicle emissions technologies also means that VOCs from use of personal care products indoors are likely to be responsible for an increasing proportion of fossil fuel VOC emissions in industrial areas outdoors, with a recent study estimating this proportion approached half of the total (McDonald *et al*., 2019).

5.2 Role of heterogeneous reactions

Heterogeneous reactions play a much greater role in indoor air composition because of the far greater surface-to-volume ratio (typically 2-4 m^2 m^{-3}) than in outdoor air. One consequence is that indoor surfaces are dominated by those of macroscale objects such as walls, carpets and furniture, rather than by the surfaces of micron-sized – and transient – airborne particles. For example, for a room with 10 μg m^{-3} aerosol loading and S/V = 3 m^{-1}, surface films 10 nm thick have more than 3 orders of magnitude higher volumes than provided by the aerosol. The effective partitioning volumes of building materials and furnishings can be much larger still due to their effective surface areas being greater than the basic footprint area of the object (Weschler, 2003). Despite the fact that organic films are a small fraction of the potential partitioning volume represented by indoor surfaces, several recent studies have tried to estimate how they may impact on indoor air quality (Weschler and Nazaroff, 2017;Wang *et al.,* 2019; Algrim *et al*., 2020).

The large surface areas indoors drive equilibrium phase partitioning and/or reactive loss of chemicals towards the surface rather than the gas phase. Many chemical compounds that are almost exclusively in the gas phase outdoors exhibit semi-volatile behaviour indoors, with comparable proportions partitioning between the gas phase and the surface films, as for example for glyoxal, the monoterpenes α-pinene and limonene, and nonanoic acid (Abbatt and Wang (2020). As well as acting as a sink and reaction site, the permanence of most indoor surfaces means they can build up substantial reservoirs of stable chemical constituents, particularly condensable organics, which in some cases may exist for very extended time periods, e.g., months and years (Weschler and Carslaw, 2018; Wang *et al*., 2020). This includes surfaces acting as reservoirs for known toxic species such as PCBs, PAHs and tobacco smoke.

Under typical indoor conditions, layers of semivolatile organic compounds (SVOCs) accumulate on new impermeable surfaces at the rate of a few nm (= 10^{-9} m) per month (Weschler and Nazaroff, 2017). After an initial aging process, the reactivity of many indoor surfaces changes little over time, reflecting their ongoing acquisition of reactive compounds derived from typical indoor sources such as skin oils, cooking and cleaning (Wang and Morrison, 2010). The chemical complexity in the deposited organic film substrates is probably analogous to that of secondary organic aerosol (SOA). Attention is also now being given to partitioning to indoor materials themselves, such as paintwork and upholstery, which dominate the available partitioning volume (Algrim *et al.*, 2020). However, despite the dominance of partitioning of semivolatiles to permanent indoor surfaces, their parallel partitioning to aerosol particles indoor can still be an important human exposure pathway given the ease with which the airborne particles are inhaled.

The generally high relative humidity of indoor air means that indoor surfaces also possess substantial molecular layers of adsorbed water; and given the generally high concentrations of ammonia in indoor environments (10s of ppb) and basic nature of some building materials such as concrete, the pH of these surface layers may be less acidic than outdoor aerosol particles (Abbatt and Wang, 2020).

In respect of the heterogeneous loss of gaseous species to indoor surfaces, by far the most studied is for ozone. Uptake probabilities (also quantified as deposition velocities) for ozone are high because of the high reactivity of ozone with surface constituents, particularly with organic layers on the surfaces (Shen and Gao, 2018). Reaction of ozone with the double bonds of unsaturated organics can result in bond breakage and the formation of oxidised organic products (oxygenated VOCs) of lower molecular mass and greater volatility, which may subsequently partially partition back into the indoor air. The production of aldehydes, through surface reactions on various materials, also leads to the enhanced formation of nitrated organic material such as peroxyacetylnitrates (Kruza *et al.*, 2017). The ozone-reactivity of certain terpenoids (e.g., Δ3-carene) is significantly enhanced on surfaces compared to the gas phase (Weschler and Carslaw, 2018). Benzo[a]pyrene (BaP) is one of many PAHs produced during cooking, smoking and other combustion activities. In the gas phase there is negligible reaction between BaP and ozone, but when BaP is adsorbed to glass it reacts with ozone to produce both mono- and diol-epoxides (Zhou *et al.*, 2017).

Another well-studied indoor heterogeneous loss process is that of NO_2, particularly because its reaction with surface water has long been known to yield nitrous acid (HONO), which occupies an important role in atmospheric chemistry, and also nitric acid (HNO_3). It has been observed that light can enhance indoor HONO production from interfacial reactions between NO_2 and household chemicals (Gómez Alvarez *et al.*, 2014). As well as being a source of OH radical, HONO can react with amine-containing molecules to yield toxic nitrosamines. Of particular relevance here is the reaction of HONO with nicotine from tobacco smoke adsorbed on indoor surfaces to produce carcinogenic tobacco-specific nitrosamines (Sleiman *et al.*, 2010).

A further important category of indoor heterogeneous reactions concern molecules containing chlorine found in cleaning products. Wong *et al*. (2017) reported that washing a floor (in a laboratory setting) with chlorine bleach solution produced elevated concentrations of both HOCl and Cl_2, with the HOCl concentration decaying faster than air exchange, irrespective or light in the room or not. They attributed this to heterogeneous reaction of HOCl with molecules present on surfaces in the room, supported by studies that showed efficient heterogeneous reactivity of HOCl with squalene and oleic acid (components of skin oil) to produce high molecular weight chlorine-containing condensed-phase products. $ClNO_2$, NCl_3 and $NHCl_2$ were also identified in the air; $NHCl_2$ may result from HOCl reacting with amines on indoor surfaces (Weschler and Carslaw, 2018).

5.3 Cigarette smoke and vaping in the indoor environment

There are three distinct types of tobacco smoke, two of which are important in the indoor atmosphere. The first type of smoke is referred to as mainstream smoke and is that inhaled directly by the smoker from the cigarette, or other smoking medium. The second is that exhaled by the smoker, together with sidestream smoke generated by the burning tobacco, but not directly inhaled. The combination of exhaled and sidestream smoke is referred to as Environmental Tobacco Smoke (ETS), or as Second-hand Smoke. The third type, referred to as Third-hand Smoke (THS), comprises components of the smoke which deposit to surfaces of people and inanimate objects within the room, and then slowly desorb back into the room as vapour. It is the latter which provides the odour of stale smoke which persists in a room long after smoking has ceased. ETS can be a major contributor to particle mass in the indoor atmosphere. Increasingly, as smoking is restricted to outdoor locations, infiltration of ETS generated outside of the building is becoming a problem (Hanninen and Goodman, 2019).

Mainstream smoke typically has a unimodal size distribution with a mode around 100-200 nm, while ETS can be significantly smaller than this, but depends upon many factors such as the smoking rate and background aerosol concentration in the room. Absolute differences in mass concentrations range from 10 - 45 $\mu g\ m^{-3}$ in the indoor environment (Rivas *et al.*, 2019) with particle number concentrations in excess of $10^4\ cm^{-3}$ (Vu *et al.*, 2017). Major chemical components of ETS include alkanoic acids, alkanes and N-heterocyclics. Minor constituents include PAH and nitrosamines, which contribute to carcinogenicity (Vu and Harrison, 2019). ETS is also a contributor to indoor VOCs, including benzene.

There has been far less work on Third-hand Smoke. Many of the constituents of ETS are semi-volatile. Both particle and vapour phases will deposit to indoor surfaces, with subsequent vapour release responsible for THS. The compounds can undergo chemical reactions, and there has been interest in the reaction of nicotine with nitrous acid, which leads to formation of tobacco-specific nitrosamines. These have been measured at concentrations in European homes at concentrations that would elevate cancer risk, particular for children, and via ingestion routes. (Ramirez *et al.* 2015, Yeh *et al.* 2022).

Reactions of volatile constituents with ozone is also reported to lead to formation of ultrafine particles (Burton, 2011).

The use of electronic (e-) cigarettes and vaping devices continues to increase. E-cigarettes contain an ‘e-liquid’ which contains various levels of nicotine and often a flavouring typically dissolved in water and propylene glycol (PG) and/or glycerol. This is vaporised by an internal heating device before inhalation. In terms of release of pollutants to the indoor air, this consists mainly of a fine aerosol, made up of water and PG/glycerol. In addition, there are now many ‘heat-not-burn’ products available on the market, in which synthetic or processed tobacco containing nicotine is heated to temperatures below those required for conventional combustion. At the time of a 2018 Public Health England review (PHE, 2018) there was not yet any evidence of significant risks to bystanders from passive vaping.

Chapter 6 – Future trends

6.1 Future trends in emissions of relevance to indoor air

There have been no previous estimates made of future trends in the emission of indoor air pollutants in the UK, but some insights can be gained by examining trends given in certain key sectors by the UK's National Atmospheric Emissions Inventory (NAEI, 2021). The NAEI is the main source of atmospheric emissions inventory data for the UK, including historical and future trends taking into account activity levels for different source sectors and changes in emission factors. Like inventories compiled in other countries, it is mainly for international reporting purposes where the main driver is monitoring progress against emission reduction targets for the purpose of improving outdoor air quality. The NAEI estimates emissions for all anthropogenic sources, without distinguishing between emissions occurring indoors and outdoors. The NAEI uses methods from the EMEP/EEA emission inventory guidebooks (EMEP/EEA 2021) which are mainly based on the principle of defining an emission rate according to a sector-specific emission factor and activity rate in the expectation that activity data are usually available from national statistical sources or from industry data and surveys.

Although it does not report explicitly on indoor air emissions, a close examination of the sectors covered in the NAEI can identify those sources which are likely to occur at least partially in an indoor environment. However, for most of these sources, separating emissions between different types of indoor environment is far more challenging, e.g. residential, commercial, public buildings. The NAEI covers >900 main source/activity combinations where activity here refers to a specific activity from a source such as a type of fuel used. Each source/activity combination will be responsible for different pollutant emissions. Nitrogen oxides (NO_x) will be dominated by combustion sources which in the majority of cases will be vented to air outdoors, whereas pollutants like volatile organic compounds (VOCs) come from many sources emitting indoors.

The NAEI indicates 12 sources, or groups of sources, that occur indoors, at least to some extent with a fairly high degree of confidence. These are listed in Table 6.1. This table shows the total UK emissions of NMVOCs, NO_x, $PM_{2.5}$ and NH_3 from each source in 2019 according to the NAEI (NAEI, 2021). As no statistics are available to show the extent of emissions from sources occurring indoors, an estimate has been made of the fraction of total UK emissions occurring indoors in a residential domestic environment and indoors in other, unspecified indoor places such as offices, schools, hospitals etc. Table 6.1 shows the percentage range considered to occur indoors in domestic and other public and office buildings. There are high levels of uncertainty in these fractions, particularly in the split between residential and other building environments and the figures are based purely on best judgement. The balance in total emissions is considered to occur outdoors. Further research and activity surveys are required to confirm these fractions and reduce the levels of uncertainty.

The majority of emissions from domestic fuel combustion will be vented outdoors but a small proportion may occur indoors, most likely from domestic gas cookers. The importance of good ventilation from gas fires or solid fuel stoves and fires used for room heating should ensure the vast majority of emissions from these sources occur outdoors, but it is not known how much residual emission may occur indoors. As stated earlier, the NAEI uses figures suggesting around 95% of emissions from natural gas used for domestic combustion are from domestic boilers, and it is assumed that the remaining 5% occur mainly from gas cookers rather than from gas fires used for room heating. Based on these assumptions, it is assumed that only gas cookers contribute to emissions from domestic combustion occurring indoors (at 5% of all domestic gas consumption emissions), with no indoor emissions occurring from other combustion sources, whether using gas or solid fuels. This is reflected in the figures in Table 6.1.

Most of the VOC emissions occurring indoors are from use of personal care and household products. For many of these aerosol and non-aerosol product sources, the inventory is based on product sales or related data from industry or from market research on consumption data; emission factors are taken from industry recommendations or from the emissions inventory guidebook. Further details are given in the NAEI's annual Informative Inventory Report (Defra, 2021) which includes future projections. The NAEI estimates refer to total UK emissions. No information is directly available on the spatial distribution of these indoor activities but the NAEI maps emissions onto a 1x1 km UK grid using various proxy datasets such as population, number of households, employment statistics etc. to spatially distribute the national emission estimates (Ricardo plc, 2020).

Table 6.1: Total UK emissions of NMVOCs, NO_x, $PM_{2.5}$ and NH_3 in 2019 from sources covered in the NAEI which are expected to occur at least partially in indoor environments. The percentage figures refer to estimates on the fractions of these UK emissions that may occur indoors in domestic residences and those which may occur inside other buildings. The remainder are assumed to occur outdoors.

Emission source	**NMVOCs**	**NO_x**	**$PM_{2.5}$**	**NH_3**	**% occurring indoors: Domestic**	**% occurring indoors: Other**
	ktonnes	**ktonnes**	**ktonnes**	**ktonnes**		
Domestic combustion of natural gas	2.74	19.48	1.21	0	5±2%	
Use of non-aerosol products - cosmetics and toiletries	11.97	0	0	0	95±5%	
Use of non-aerosol products - general household	13.76	0	0	1.21	80±10%	20±10%
Use of non-aerosol products - domestic adhesives	4.48	0	0	0	95±5%	
Use of non-aerosol products - paint thinner	13.69	0	0	0	40±10%	40±10%
Use of aerosol products - cosmetics and toiletries	44.59	0	0	0	95±5%	

Use of aerosol products - general household	7.07	0	0	0	80±10%	20±10%
Use of decorative paint - retail decorative	15.29	0	0	0	80±10%	
Use of decorative paint - trade decorative	12.81	0	0	0		80±10%
Cigarette smoking	0.15	0.06	0.85	0.13	80±10%	
Infant emissions from nappies	0	0	0	0.04	50±20%	
Adult breath and sweat	0	0	0	0.97	60±20%	20±20%

Tables A1-A4 in the Appendix 2 show the total UK emissions of NMVOCs, NO_x, $PM_{2.5}$ and NH_3 in 2019 for each of these sources (where relevant) occurring indoors in residential and other indoor environments derived from combining the UK totals and indoor percentage figures shown in Table 6.1. These tables also show the total emissions occurring indoors as a fraction of all UK emissions in 2019. Table 6.2 summarises the total UK indoor emissions for each pollutant and as a percentage of all UK emissions.

Table 6.2: Total UK indoor emissions of NMVOCs, NO_x, $PM_{2.5}$ and NH_3 (in kilotonnes) derived from the UK totals (indoor plus outdoors) according to the NAEI in 2019 and the outdoor emission percentages shown in Table 6.1. Further details for each sector and pollutant are given in Tables A1-A4 in Appendix 2.

	NMVOCs	**NO_x**	**$PM_{2.5}$**	**NH_3**
Total UK indoor emissions (ktonnes)	112.5	1.02	0.74	2.10
Percent of total UK emissions occurring indoors	13.8%	0.10%	0.66%	0.75%

It is important to recognise that the uncertainty estimates shown in the Appendix 2 tables only consider the uncertainties in the distribution of emissions between indoor and outdoor environments from best judgement and not in the uncertainties in the overall UK estimates themselves for each source as derived by the NAEI. For many of these indoor sources, the uncertainties in the UK totals can be high due to the lack of available activity data and emission factors, especially for sources such as cigarette smoking. The true range in UK emissions occurring indoors will therefore be higher than indicated in these tables, an important factor when considering human exposure from these sources indoors vs outdoors.

The purpose of these data is to show the overall contribution of each source to national total and the extent to which these occur indoors. It is also important to note that the NAEI does not make estimates at all for certain processes occurring indoors such as cooking and use of food and drink products

These tables highlight the importance of indoor sources of VOCs from consumption of various product types. These are estimated to be approximately 14% (±1%) of national total emissions from all sources in the UK. The contributions of indoor sources to total emissions of other pollutants are estimated to be much lower (<1%), although the uncertainty in sources such as cigarette smoking and the fractions of these occurring indoors are relatively high. What is also evident from the NAEI data is that the indoor sources of VOC emissions have remained fairly static over the time-series, ranging from 112-127 ktonnes per annum between 2000 and 2019.

Due to decreases in emissions from outdoor sources, the proportion of VOCs emitted indoors has increased from around 7% in 2000 to 14% in 2019. The same assumptions imply that indoor emissions of NO_x and $PM_{2.5}$ have roughly halved over this time period leading to little overall change in the proportions emitted indoors.

This analysis is only based on data held by the NAEI from the time-series trends in activity and emission factors of each source and assumptions based on our best judgement of the proportions emitted indoors. It relates only to the pollutants and sources covered by the NAEI which is mainly developed for understanding emissions and their contributions to outdoor air quality. Nevertheless, the NAEI provides a useful starting point for understanding the contributions of different sources to VOC emissions indoors and how the trends may continue into the future.

Figure 6.1 shows the NAEI estimates of emissions of VOCs from different sources indoors in 2019 and 2030, both for residential buildings (peoples' homes) and from other indoor buildings. These are based on the assumptions described above and NAEI predictions in future trends in activity levels and emission factors for each source. The projections indicate little change in activity data and emission factors are expected for each source and also assume no change in the share of UK emissions between the indoor and outdoor environment. With modest decreases expected from other sources of VOCs occurring outdoors, then the share of overall emissions occurring indoors increases very slightly from 14% in 2019 to 15% in 2030.

Based on this simple analysis of NAEI data, the relatively small amount of NO_x and NH_3 emitted from indoor sources will remain almost unchanged, both at ~1-2 kt per annum between 2019 and 2030, while $PM_{2.5}$ emissions which are also emitted in relatively small proportions may decrease from 0.74 kt in 2019 to 0.56 kt in 2030 due to predicted reductions in emissions from cigarette smoking.

Figure 6.1 does indicate how there may be a difference in the contribution of different sources to indoor VOC emissions in homes and in other buildings. In homes, 58% of emissions are estimated to occur from cosmetics and toiletries (aerosol and non-aerosol products) whereas in other buildings the majority of VOC emissions (79%) may come from paint products. However, this assertion does need testing by obtaining further data on usage patterns of these products and should be further validated by indoor VOC measurements.

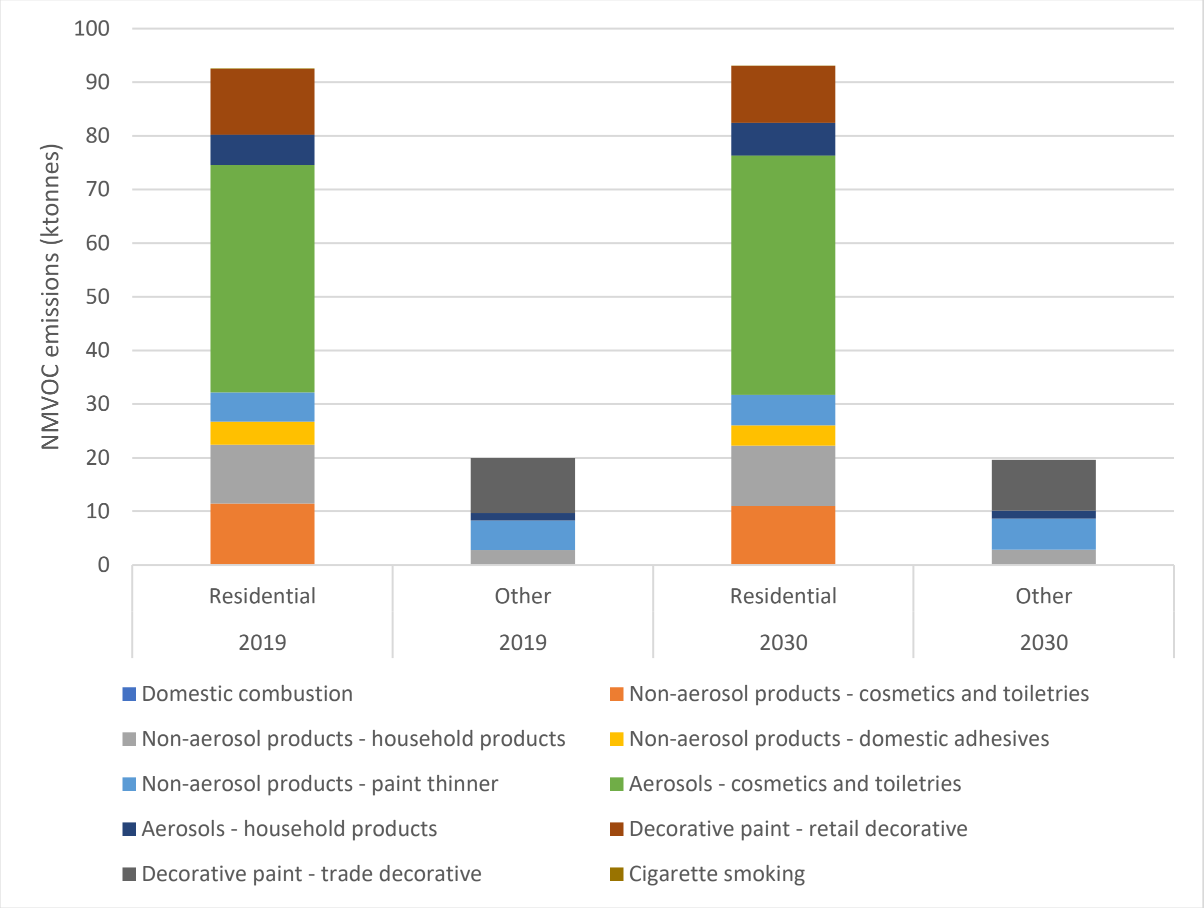

Figure 6.1: VOC emissions from key sources occurring indoors for residential and other buildings estimated from the NAEI's data for 2019 and projected to 2030.

To interpret the link between indoor VOC measurements and the emissions inventory, it is constructive to examine the NAEI's speciated inventory for VOCs. The NAEI's NMVOC inventory is broken down into 664 chemical species or groups of species using profiles for each detailed emission source sector. Further details were given in the AQEG report "Non Methane Volatile Organic Compounds in the UK" (AQEG, 2020b)

Using the speciated profile for each of the sources in Table 6.1 and their relative contributions to indoor emissions it is possible to derive the dominant VOC species emitted in an indoor environment. There are many individual VOCs or types of VOCs emitted from each individual source but Table 6.3 shows the top 50 VOCs estimated to be emitted from sources in residential buildings in 2019 based on the NAEI attributions.

Table 6.3: Top 50 NMVOC species estimated to be emitted indoors in residential buildings in 2019. Figures based on current NAEI estimates of total UK NMVOC emissions and estimates of the fractions emitted indoors according to Table 6.1 combined with current the current NAEI speciation profiles for each source.

Species	ktonnes
ethanol	26.23
butane	20.81

propane	4.01
2-propanol	3.23
decane	2.12
nonane	1.25
1,1,1-trichloroethane	1.12
undecane	1.11
acetone	1.00
1-propanol	0.93
1,4-dichlorobenzene	0.92
2-methylpropane	0.79
dimethyl ether	0.79
dipentene	0.78
4-methyldecane	0.65
1,2,4-trimethylbenzene	0.58
ethyl acetate	0.57
3-methylnonane	0.53
3-ethyl-2-methylheptane	0.52
2-butanone	0.51
2-butoxyethanol	0.47
pine oil	0.46
2-methylnonane	0.46
propylcyclohexane	0.44
2,6-dimethyloctane	0.43
C10 cycloalkanes	0.40
toluene	0.39
3-methyldecane	0.39
C11 alkanes	0.37
(1-methylpropyl)cyclohexane	0.37
trichloroethene	0.37
dichloromethane	0.37
4-methylnonane	0.36
1-methyl-4-isopropylcyclohexane	0.36
2-methyldecane	0.34
butylcyclohexane	0.34
1-methyl-4-isopropylbenzene	0.33
C10 alkanes	0.32
1-ethyl-3-methylcyclohexane	0.31
4-methyl-2-pentanone	0.30
ethylene glycol	0.30
3-ethyltoluene	0.29
unspeciated aromatic hydrocarbons	0.28
1,3,5-trimethylbenzene	0.28
pentane	0.28
1,2,3-trimethylbenzene	0.25
m-xylene	0.23
butyl acetate	0.21
(1-methylethyl)cyclohexane	0.21
(2-methylpropyl)cyclohexane	0.20
Other	14.01

The top 50 NMVOC species represent ~85% of all NMVOCs estimated to be emitted indoors in residential buildings. Approximately 50% of all NMVOCs emitted indoors are as ethanol and butane, but there are many more each emitted in relatively small amounts. Most of the these are emitted from use of cosmetic and toiletry products as well as a range of other household products.

Table 6.4 shows the top 50 VOCs emitted from sources in other, non-residential buildings in 2019 based on NAEI attributions.

Table 6.4: Top 50 NMVOC species estimated to be emitted indoors in other non-residential buildings in 2019. Figures based on current NAEI estimates of total UK NMVOC emissions and estimates of the fractions emitted indoors according to Table 6.1 combined with current the current NAEI speciation profiles for each source

Species	**ktonnes**
decane	1.46
ethanol	1.17
nonane	0.84
undecane	0.76
dipentene	0.65
butane	0.63
4-methyldecane	0.44
1,2,4-trimethylbenzene	0.41
3-methylnonane	0.37
3-ethyl-2-methylheptane	0.36
2-methylnonane	0.32
propylcyclohexane	0.30
2,6-dimethyloctane	0.30
C10 cycloalkanes	0.28
3-methyldecane	0.27
C11 alkanes	0.26
(1-methylpropyl)cyclohexane	0.25
4-methylnonane	0.25
1-methyl-4-isopropylcyclohexane	0.25
2-propanol	0.24
2-methyldecane	0.24
1-methyl-4-isopropylbenzene	0.23
butylcyclohexane	0.23
1,4-dichlorobenzene	0.23
C10 alkanes	0.22
1-ethyl-3-methylcyclohexane	0.21
3-ethyltoluene	0.21
1,3,5-trimethylbenzene	0.20
propane	0.19
1,2,3-trimethylbenzene	0.18
(1-methylethyl)cyclohexane	0.14
1,2-propanediol	0.14
2-(2-butoxyethoxy)ethanol	0.14
1-(2-butoxy-1-methyl-ethoxy)-2-propanol	0.14
2-(2-ethoxyethoxy)ethanol	0.14
2,2,4-trimethyl-1,3-pentanediol monoisobutyrate	0.14
benzyl alcohol	0.14
tri-n-butyl phosphate	0.14
(2-methylpropyl)cyclohexane	0.14

1-methyl-3-propylbenzene	0.14
propylbenzene	0.13
1,2,3-trimethylcyclohexane	0.12
5-methyldecane	0.12
pine oil	0.12
2-methylpropane	0.11
4-methyloctane	0.11
2-butoxyethanol	0.11
2-methyloctane	0.11
2,5-dimethyloctane	0.10
3-methyloctane	0.10
Other	5.50

The top 50 NMVOC species emitted in these environments are slightly different and represent ~72% of all NMVOCs estimated to be emitted indoors in non-residential buildings. Because of the different sources believed to be important compared with residential buildings, there are fewer really dominant types of NMVOCs, but a few of the larger alkanes such as decane and nonane, as well ethanol, are emitted in relatively large amounts due to the contributions from paint thinners and decorative trade paints.

These speciated figures are highly uncertain and should only be seen as indicative of the possible differences in the types of VOCs emitted in indoor environments. There are uncertainties in the total UK estimates of NMVOC emissions themselves, and particularly high uncertainties in the proportions estimated to be emitted indoors and their speciation profiles.

A key point to note about this data is that it mainly refers to the VOCs occurring from the main carrier or solvent species associated with a product and may not capture the very specific chemical species used as the active ingredient, fragrance etc. These may be present in very small quantities in a particular product and be unique to each manufacturer, yet may still be detectable in indoor air and play an important role in determining indoor air quality.

6.2 Future Transport Emissions

Exposure to emissions inside different modes of transport will occur from ingress of ambient air, where concentrations are themselves influenced by emissions to air outside from exhaust and non-exhaust sources, and from sources within the vehicle itself (car, bus, train etc). Outdoor air pollution caused by transport sources have been relatively well-characterised and exhaust emissions are generally predicted to decline as new, lower emitting vehicles enter the fleet, coupled with further electrification of road and rail transport. In the case of road transport, exhaust emissions of NO_x, $PM_{2.5}$ and NMVOCs are predicted to decrease by 70%, 40% and 74%, respectively, between 2019 and 2030. These reductions are predicted to be offset by increases in emissions from non-exhaust sources such as VOCs from evaporative losses from petrol vehicles (increasing by 11%) and PM from tyre and brake wear and road abrasion (increasing by 13%) as traffic and numbers of vehicles on the road continue to increase. [Note: The NAEI currently assumes only a modest uptake of battery electric vehicles using figures from DfT, with an increase in number of petrol vehicles (including hybrids) on the road up to 2030.]

Overall, the NAEI currently predicts emissions of NO_x, $PM_{2.5}$ and NMVOCs from road transport sources to decrease by 70%, 16% and 16%, respectively, between 2019 and 2030. The modest reduction in $PM_{2.5}$ emission reflects the dominance of unregulated non-exhaust sources from tyre and brake wear and road abrasion. Small reductions are also predicted in future emissions from other transport sources (see https://www.ceip.at/webdab-emission-database/reported-emissiondata for further data on emission projections by source).

Emissions from sources inside a vehicle have not been covered by the NAEI, but may include emissions of VOCs from materials and furniture in the vehicle, use of car care and cleaning products and windscreen wash and de-icing products (the NAEI does include emissions from car-care product use, but this covers a range of applications which does not allow separation between outside and in-vehicle emissions). Assuming no measures are put in place to reduce these emissions, e.g. by change of product formulation or application, then these emissions are expected to increase with increases in numbers of vehicles, coupled with increases in number of trips, distances or time travelled leading to increases in in-cabin exposure. The Department for Transport current road traffic forecasts imply a growth in vehicle kilometres travelled of around 14% from 2019 to 2030 associated with a growth in the number of vehicles on the road. Whilst forecasts in number of passenger kilometres travelled may be a better indicator of changes in future in-cabin exposure, the anticipated future increases in traffic and vehicle numbers combined with growth in demand for other modes of transport would indicate that in-cabin emissions are likely to increase in the future.

Chapter 7 - Interventions to improve indoor air quality

There are many practical ways in which indoor air quality may be improved, which include those associated with the building design and its in-built ventilation and/or fume extraction systems, those attributed to source control (external and internal sources, including those due to occupant activities and habits), occupant behaviours, and use of specialised equipment for filtration, purification and cleaning of indoor air.

Drivers for the application of solutions for improvement of indoor air quality may arise from global or national policies, from local and community initiatives, or be implemented at organisational or single building level. Such interventions may be proactive, instigated as a result of incentivisation or the desire to adopt best practice, or reactive in response to concerns for the health and wellbeing of building occupants (often due to potential exposure to specific air pollutants).

Some mitigation measures around source control are in the gift of individual building occupants, such as control or eradication of habits such as smoking and vaping; reduced use of certain consumer products; choice of low-emitting decorative products, furnishings, flooring, etc. However, there is very often the need for enhanced education and awareness when it comes to understanding the effects of certain activities and choices on indoor air, and then in taking actions which will improve air quality in indoor spaces.

The National Institute for health and Care Excellence (NICE) worked with the UK Health Security Agency (UKHSA, formerly Public Health EnglandPHE) on the development of the NICE guidelines on indoor air quality at home (NICE, 2020), which are UKHSA co-badged. The focus is on interventions related to the structure of, ventilation of, and materials used in, new / existing or retrofitted homes as well as on people’s knowledge, attitude and behaviour in relation to indoor air pollution. They are addressed to Local Authorities (LA), medical and health professionals and building industry. Although the guidelines were published before the pandemic, those for the LAs are applicable for every home occupant who wants to improve and maintain a good air quality at home. The COVID-19 pandemic has served to heighten awareness of indoor air quality issues and the need for adequate ventilation in buildings, arguably acting as a catalyst for a paradigm shift in how this is approached.

See Also: “Infection Resilient Environments: Buildings that keep us healthy and safe.” Initial Report. Royal Academy of Engineering, 2021. This report sets out several key recommendations including the need for joined-up collaboration between government, its agencies and professional bodies to act to close gaps in knowledge, as well as the importance of technological interventions, and use of incentivisation to improve poor indoor environments (raeng.org.uk).

7.1 Ventilation and local moisture/fume extraction.

To make the UK's building stock more insulated and airtight and to reduce the amount of energy required for heating, the objective is to reduce uncontrolled, and hence overall, ventilation. The current minimum standard permissible under UK Building Regulations (Part L) is 10 $m^3/hr/m^2$ @50 Pa, with 'good practice' defined as a maximum of 7 $m^3/hr/m^2$ @50 Pa and best practice as 3 $m^3/hr/m^2$ @50 Pa. Any building with an air tightness tighter than 4-5 $m^3/hr/m^2$ @50 Pa is recommended to have some form of mechanical ventilation, since windows, trickle vents and extract fans would not be sufficient to maintain healthy rates of ventilation.

The majority of low-carbon building certifications (e.g. BREEAM or Passivhaus) require air tightness even lower than this, therefore necessitating the incorporation of mechanical ventilation – often with heat recovery (MVHR) – into the design of the build or retrofit. Indeed, a 2012 report by Zero Carbon Hub and NHBC Foundation predicted MVHR to become the dominant form of ventilation in new homes (Dengel *et al.,* 2013) It should be noted that Passivhaus measure airtightness in terms of air changes per hour (ach^{-1}), dependent on the building's internal air volume rather than envelope area and therefore a standardised conversion between these units and air permeability is not possible (BRE/Passivhaus Institute, no date).

As well as retaining heat, moisture in the form of water vapour is also retained in an airtight building. With modern homes becoming more airtight, and with few new homes being built with airing cupboards, continuous background ventilation is essential in allowing that moisture to escape the building. Atmospheric water vapour levels are high in the UK compared to other parts of the world (Madgwick and Wood, 2016), making UK homes particularly vulnerable to high RH, leading to condensation and mould growth. Indeed, a study of Code 3 sustainable homes found that many that were naturally ventilated had RH which exceeded 70%, giving rise to mould (McGill *et al*. 2015). One of the primary negative health consequences of living in homes with high humidity – and consequent damp and mould – is an increased risk of asthma (Sharpe *et al.* 2015b, and Fisk *et al.* 2007).

Mechanical ventilation systems (MEV, MVHR, Demand controlled ventilation) that are able to provide sufficient air changes and limit air pollution to within air quality guidelines need to consider the following:

- Ventilation systems should be fully serviced after all other construction work has been completed: dust etc. produced can take up to 6 months to settle and can block filters and ducts.

- Systems require regular cleaning and replacement of filters to maintain performance as well as preventing microbial overgrowth that may lead to the generation of high burdens of bioaerosols when ventilation systems are turned on. They should therefore be installed where they are accessible, avoiding less accessible spaces such as lofts where possible.

- Actual as-built air permeability rather than designed air permeability should determine the ventilation strategy.
- Occupants are generally happier if they feel they have control over their environment, but this means controls need to be clear and simple or the system performance can be undermined.
- Occupant interference with the systems is more likely if they generate excessive noise.
- Systems need to be designed with varying occupancy rates in mind.
- The location of air intake is important in determining the amount of pollutants brought into a building from outside.

7.2 Source control (see also occupant behaviours)

One of the most effective methods of source control is the use of low-emitting products to reduce the air pollution load in buildings. Instruments such as the Construction Products Regulations 2013 (see Section 2.1.5) and a range of voluntary product labelling schemes (see Section 2.3) can be applied - in order to specify and procure low-emitting materials for use in construction and fit-out of buildings. Environmental evaluation schemes such as BREEAM, LEED and WELL offer credits for use of low-emitting construction products. Indoor air quality monitoring can be carried out to verify the actual level of airborne contaminants such as VOCs.

7.3 Occupant activities and behaviours.

A most obvious way to reduce indoor pollution is to remove the sources. However, given that occupant activities lead to indoor air pollutants and that some of these activities are essential (e.g., cooking and cleaning), complete removal of emissions is not always possible. Consequently, mitigating measures can be considered to reduce the size of emissions and/or reduce indoor concentrations. The following sections consider different sources and mitigation measures in more detail.

7.3.1 Use of chemical-emitting consumer products

If these emission sources cannot be removed or reduced, they should as a minimum be used in well ventilated areas. For example, the burning of candles emits HCHO and PM as for any combustion product, and scented candles can release other VOCs in addition. In Denmark, exposure to PM from candle burning has been estimated as the main indoor source of PM in around 60 typical Danish homes studied (Beko *et al*., 2013). Electric candles are an alternative and they directly reduce air pollution emissions; however, they use batteries so are associated with production and disposal impacts.

Many personal care products release VOCs, the most significant in a UK context being compressed aerosol devices containing typically *n* and *iso* butane, and propane as propellent and ethanol as a co-solvent (Yeoman and Lewis 2021). Many other personal care products also contain some VOC content that is released when the product is used. Some simple parameterisations for emissions from individual products found in UK supermarkets were developed by (Yeoman *et al.* 2020), and the build-up of VOCs during personal care product during showering was evaluated in Yeoman *et al*. (2021). Any individual actions which lower overall consumption of personal care or cleaning products or lead to the use of lower VOC containing products feed through directly into lower indoor emissions and better indoor air quality.

Sources of room fragrance can also release VOCs upon use. Some air fresheners (diffusers) work by using a power supply to heat a mixture which then provides a constant stream of low volatility fragrance into an indoor space. Elevated concentrations can be generated inside a test chamber (Uhde and Schulz, 2015), although less is known about the significant of these products in homes with likely higher rates of air exchange, and many other VOCs sources. Minimising the operating time can reduce the build of pollutants, as can opening windows. Air fresheners and incense are often used to mask odours, although the odour of fragrances themselves generates mixed reactions, with many finding them offensive in their own right. Identifying and removing the source of malodour or ventilating by opening windows is both preferable to attempting to mask an odour with other VOCs. It also needs to be noted that fragrances and poor indoor air quality are sometimes associated with one another because this class of VOC is detectable by the human nose. Most VOCs have little or no smell, so the absence of an odour, good or bad, does not necessarily indicate low VOC concentrations.

7.3.2 Use of combustion appliances

A major source of NO_X and PM indoors is cooking using gas, either natural gas or LPG mixtures. A simple mitigation measure is to replace gas cooking appliances with electric equivalents whenever possible, a change that is likely to be needed as part of wider home decarbonisation and transition to net zero. Other approaches to reducing emissions from cooking include: always use the extractor fan when cooking; use the back rings on a hob if possible (as the extractor fan works more efficiently for these) and, open the window for 10 minutes once cooking has ended. Note that extractor fans need to vent outdoors and not back into the kitchen. Research has also shown that when frying, different oils produce different numbers and size distributions of particles: one simple measure to reduce particle formation is to replace olive oil with sunflower oil when frying (Abdullahi *et al.*, 2013).

Wood stoves are a further source of indoor (and outdoor) air pollutants from combustion although are used in a much smaller subset of homes and cooking appliances. It is important that fires and stoves are well maintained. Whilst a modern, well-maintained wood stove using well-seasoned wood would generate lower indoor emissions than an open coal fire, all solid fuel combustion leads to substantial outdoor emissions with negative effects both locally and more broadly.

7.3.3 Use of windows, trickle vents, extractor fans

The activities and behaviours of occupants can play a significant role in determining the indoor air quality and pollutants that are added to the air they breathe. Historically and traditionally, people have ventilated their homes by opening windows. Throughout the heating season (broadly considered to be October to April in the UK), windows tend to stay closed to prevent heat loss. Throughout the non-heating season (May to September in the UK), windows are much more likely to be opened, particularly on hot days. These practices can have negative consequences for indoor air quality. Keeping windows closed for long periods through the heating season can lead to build ups of moisture in the air, increasing the likelihood of damp, mould and mildew on windows and walls, particularly in high-humidity rooms such as bathrooms and kitchens. Conversely, in the summer it may often seem intuitive to open windows on particularly hot days. But when outdoor temperatures exceed what would be considered a comfortable temperature range indoors, opening windows only serves to allow the hotter outdoor air into the building, exacerbating the problem.

Modern buildings designed to provide more comfortable indoor environments with lower energy demand are often built with the philosophy of being as insulated and airtight as practically possible. But these cultural habits of opening and closing windows to control indoor environments can undermine the design intentions. Furthermore, research has found that in offices where the indoor environment is mechanically controlled and windows cannot be opened manually, occupants can have a significantly lower tolerance towards changes in temperature, RH and CO_2 concentrations than their counterparts in naturally ventilated offices where the occupants can open and close windows as and when they want (Gupta *et al*. 2020).

Trickle vents are a common feature of many modern windows, providing background ventilation at a more gradual but consistent level. Modern windows are airtight, making them more energy efficient, but this can lead to moisture build-up and reduced indoor air quality. Therefore, trickle vents are included in the majority of modern window frames, and they are compulsory if replacing windows which also have trickle vents (MHCLG, 2010). The vents are typically small slots, often located in the window frame above the pane, with a cover on the internal side of the window which allows the user to open and close the slot and let air gradually trickle into the room. This background ventilation is particularly important in buildings where there is no mechanical ventilation. However, a 2015 study by Sharpe *et al.* found that fewer than one in ten survey respondents ever adjusted their trickle vents, and fewer than three in ten respondents kept the trickle vents constantly open (Sharpe *et al*. 2015). Reasons why the majority kept their trickle vents closed included noise from outside, accessibility – particularly for the elderly, and concerns about heat loss. The effectiveness of trickle vents in providing adequate ventilation can be further reduced by the vents being obscured by curtains and blinds, and by internal doors being closed, which greatly reduces the potential for cross-ventilation (Sharpe *et al*. 2015, Sharpe *et al*. 2014)

7.3.4 Education and awareness

The incorporation of mechanical ventilation into UK buildings, particularly domestic dwellings, represents something of a culture change – more so than in warmer climates where air conditioning and other forms of climate control are more widespread. Therefore, it is important that building users understand how these systems operate, perform and should be maintained. This includes understanding how previous behaviours and patterns, such as window opening habits, may need to be modified so as not to offset or negate the benefits of the new ventilation system.

A good example of behavioural impacts is the drying of laundry, something which can also have a significant effect on indoor air quality. David MacKay, in his book *How People Can Use Energy More Efficiently*, weighed laundry before and after washing and found a 4 kg dry load weighed 2.2 kg more coming out of the machine – equivalent to 2 litres of water. In his book *Dampness in Buildings*, Oliver concluded that drying clothes can produce between 3 and 7.5 litres of moisture (Oliver, 1996). In addition to adding moisture to the air, drying laundry indoors may increase levels of other indoor air pollutants, particularly if fragranced laundry products have been used (Goodman *et al*. 2019).

Taking the clothes drying example, different methods of drying clothes indoors (excluding use of a tumble drier) have different indoor air effects. Drying laundry on radiators may be superficially a quicker way of getting the job done, but requires energy from the boiler and therefore more energy is required to heat the home. Drying laundry on a rack is a slower process, with the energy needed to evaporate the water coming from the ambient air. This can take significantly longer time, but ultimately the same amount of water vapour is added to the indoor air. Using a dehumidifier to remove the moisture from the air can help alleviate this problem. However, dehumidifiers cost money to buy and to run – although are significantly cheaper than a tumble drier – and they are not a common feature in UK homes.

The activities and behaviours described above all highlight the need for building occupants to understand the implications and consequences of their actions. To some extent, how we interact with the buildings we occupy is intuitive – if we're feeling cold, we might turn up the heating, if we're feeling warm or the air feels stuffy, we might open a window. However, many of the activities and behaviours described have more subtle effects on the indoor environment. Drying clothes can raise indoor RH, but building occupants may not be able to perceive this at the time, and even when mould starts appearing on windows and walls, not necessarily associate their actions as being part of the root cause.

It is therefore vital that building occupants – the population as a whole – is made aware of the implications of actions possibly using communication through media, better user guides and training by housing officers. As has been found with many other 'public awareness campaigns', this is often most effective when presented positively – offering alternative actions and behaviours that can have a more beneficial effect on the indoor environment, rather than simply telling people what they are doing wrong and why they need to stop.

7.4 Filtration and removal of air pollutants

Air-cleaning technology is being increasingly prominent as a possible means of improving air quality (Siegel, 2016). Air-cleaning devices (ACD) adopt one of a number of different techniques: thermal-or photocatalytic oxidation, adsorption, filtration (of particles), UV germicidal irradiation, ion generation, and electrostatic precipitation (Zhang *et al.*, 2011). However, none of these approaches can remove all of air pollutants present indoors and many can generate undesirable secondary products, see for example the review of available literature by Zhang *et al.*, 2011. Some air cleaning devices operate by generating high concentrations of hydroxyl (OH) radicals, with the aim of removing biological pathogens. However, OH radicals can initiate gas phase chemical oxidation indoors, leading to a wide variety of chemically complex products some of which are likely to be harmful to health (Waring and Wells, 2015).

It is important to note that, broadly speaking, such techniques should not be seen as a substitute for ventilation; (SAGE-EMG, 2020) this consideration has grown in importance in recent times since the Covid-19 pandemic. A potential unintended consequence of the use of some ACDs is disruption of airflows associated with existing mechanical ventilation systems. It must also be noted that the current regulatory landscape for air cleaning devices is complicated and, in some cases, patchy in terms of performance metrics and to which pollutants performance standards apply.

7.4.1 Air Cleaning Technologies

Zhang *et al*., (2011) identified 26,000 research articles that had been published up to June 2009, which were filtered down to 59 relevant articles on ACDs for more detailed consideration. The articles were focused on those where air was drawn into a device and then returned to an indoor space (so-called 'fan-driven') and operated within a room, so excluding those intended only for outdoor air intakes. The air cleaning devices reviewed were therefore: filtration, including high efficiency particulate air (HEPA) filters, adsorption, ultraviolet germicidal irradiation (UVGI), photocatalytic oxidation (PCO), thermal catalytic oxidation (TCO), plasma, botanic air cleaners, ion generators, and electrostatic precipitators. Their efficiency at removing different indoor pollutants was then reviewed and the authors focused on realistic conditions, so excluded industrial or unrealistically high concentrations. The key conclusions from this review were:

> '*(1) None of the reviewed technologies was able to effectively remove all indoor pollutants and many were found to generate undesirable by-products during operation.*
>
> *(2) Particle filtration and sorption of gaseous pollutants were among the most effective air cleaning technologies, but there is insufficient information regarding long-term performance and proper maintenance.*
>
> *(3) The existing data make it difficult to extract information such as Clean Air Delivery Rate (CADR), which represents a common benchmark for comparing the performance of different air cleaning technologies.*

(4) To compare and select suitable indoor air cleaning devices, a labelling system accounting for characteristics such as CADR, energy consumption, volume, harmful by-products, and life span is necessary. For that purpose, a standard test room and condition should be built and studied.

(5) Although there is evidence that some air cleaning technologies improve indoor air quality, further research is needed before any of them can be confidently recommended for use in indoor environments.'

A more recent update was provided by the US EPA in 2018 (US EPA, 2018). The key points from the two reviews are summarised in Table 7.1.

Table 7.1: Summary of different techniques reviewed and advantages and disadvantages of each based on (Zhang et al., 2011) and (US EPA, 2018).

Technique	**How?**	**Removes**	**Benefits**	**Issues**
Catalytic oxidation	Commonly photolytically using TiO_2	Aldehydes, aromatics, alkanes, olefins, halogenated VOCs, odour compounds, NO	Good at removing single compounds; efficiency ranges from 16-90%. Can be combined with adsorbent media to increase efficiency	Competitive adsorption effect by contaminants and water vapour can affect oxidation rate; not as good with mixtures; catalyst has finite lifespan. Can form HCHO and CH_3CHO and other aldehydes, NO_2 and CO_2 as by-products, depending on target compound
Electrostatic precipitation (ESP)	Corona discharge wire charges incoming particles which are collected on oppositely charged plates.	Particles	High collection efficiency (60-95%); no pressure drops, low maintenance	Can generate NO_X and O_3; high energy requirements; efficiency decreases with loading; plates need cleaning; efficiency varies with particle composition
Fibrous filter media	Filter fibres capture particles. Mechanical media filter (MMF) Electrostatic charge via electret media filters (EMF)	Particles	If rated high efficiency, very good at removing particles; MMF have improved efficiency with loading	Poorly maintained filters produce sensory irritation; filters need regular replacement; high pressure drops for some filters affecting HVAC systems EMF have reduced efficiency with loading; high pressure drops for some filters affecting HVAC systems
Ionisers	Similar to ESP: ions are produced which stick to particles, which	Particles	Low power; quiet, low maintenance	Generates ozone, low effectiveness because of low airflow rates and low CADRs, ionised particles can settle on indoor

	can then be collected or stick to other surfaces in room			surfaces rather than actually being removed
Ozone oxidation*	Can use O_3 and various micro- or meso-porous adsorbents to enhance catalytic reaction in the porous structure and reduce residual O_3	VOCs	Can enhance catalytic oxidation for VOC removal	Ozone reacts with many indoor species to produce harmful secondary products, e.g. SOA
Plasma	Corona discharge with alternating current, direct current and dielectric barrier discharge to ionise pollutants	Particles; can be combined with catalytic technology to remove some VOCs	Particles removed at 76-99% efficiency. Can enhance other technologies, e.g., filtration	Not good at removing gas-phase pollutants. Produces NOX and O3
Sorption	Adsorption (gases physically adsorb onto high surface area media, e.g., activated carbon)	Gas pollutant removal	Potential for high removal efficiency for many gases no by-products;	Sorbed VOCs and O_3 may generate reaction products; humidity/other adsorbents make sorption less efficient; quantity required and lifetime of sorbent not really known; regular replacement needed
	Chemisorption	Gases chemically absorb onto media impregnated with reactive media.	Potential for high removal efficiency for many gases; pollutants permanently captured	Regular replacement needed; effectiveness often unknown;
UVGI	Use of UV light from 200-365 nm to kill / inactivate microbes	microbes	Effective at high intensity with sufficient residence time;	Produces secondary pollutants, e.g., ozone; high power, inactivates but doesn’t remove microbes; potential for eye injury.

*electrostatic precipitators, and ion generators typically use ozone oxidation.

Zhang *et al*. (2011) noted that some newer technologies including plasma and photocatalytic oxidation had the potential to remove multiple pollutants but that there was likely to be formation of secondary pollutants. The advantage of such technologies is potentially the lower maintenance compared with having to regularly replace a filter. However, the fact that many such appliances generate possibly harmful by-products means that in reality, they likely need to be combined with a filter (Gunschera *et al.*, 2016). Sorption, UVGI and filtration have been used for longer, but tend to be better at removal of one pollutant at a time. A future development may be the combination of some of these technologies to remove by-products formed (Pierpaoli *et al.*, 2017). For instance, combining a corona discharge with electret filters increased particle collection efficiency from around 34% for just the filter to nearly 100% for the filter and discharge combined (Sambudi *et al.*, 2017). Care does of course need to be taken with the ordering of the different removal technologies to avoid emitting secondary pollutants from these technologies as detailed in Table 7.1 (US EPA, 2018). A systematic literature review by UKHSA (Cheek *et al*., 2021), evaluated the impact of portable air filtration equipment on indoor $PM_{2.5}$ focusing on effects on adults and children in indoor environments (homes, schools and offices). This showed that portable filtration devices lead to variable levels of reduction in $PM_{2.5}$ when compared to controls. Finally, the position of any portable air cleaning device in a room needs careful consideration, as this can affect the efficiency of cleaning and reduce the efficiency below that stated by the manufacturer (Akbari and Salmanzadeh, 2019; Jin *et al*., 2016; Kupper *et al*., 2019). The ventilation rate can also affect performance (Ciuzas *et al*., 2016).

Built in mechanical ventilation systems in larger buildings typically use fibrous filters to capture suspended particles as the air passes through the ducts. These filters are most effective in reducing concentrations of particulate matter. Research suggests that they are less effective/ineffective in filtering gases (Luengas *et al.* 2015), since these filters can become clogged over time and require regular cleaning or changing to reduce the risk of the ducting becoming clogged or the captured pollutants being reintroduced to the environment. Many air cleaners relying on recirculation of air within an indoor space have filters of one or more type (e.g., HEPA, activated charcoal) as part of their mechanism. It should be noted that filtration systems are selective and therefore only effective in dealing with the air pollutants for which they are designed.

Chapter 8 - Monitoring and assessment of indoor air

The importance of good indoor air quality in contributing to the health, wellbeing and performance of building occupants is well established, but it is very rarely actively managed, something that would need observations to initiate actions or mitigation. Currently air quality monitoring is normally only conducted within an indoor space when:

1. there is a problem experienced by building occupants which is suspected to be caused by poor IAQ, and the risks to health and wellbeing require assessment;
2. a building or developer is seeking to gain credits in environmental assessment schemes such as BREEAM, LEED, WELL; or
3. measurements forms part of limited, and often short-term, research study or campaign.

In the years leading up to the pandemic, the use of low-cost air quality monitors was becoming more prevalent, however a UK Government initiative instigated as a result of the pandemic has seen large-scale deployment of CO_2 monitors in schools. When monitoring and assessment of indoor air quality is to be undertaken there are several important factors to be considered including: type of sampling (i.e., passive, active spot sampling or active continuous sampling); choice of equipment (for sampling, laboratory analysis or continuous monitoring); sampling locations and access to them; sampling periods.

Since the data from monitoring may be used for occupational health, medical, operational, financial or HR decisions, it is important to consider the following attributes of the method(s) used: applicability; range; reproducibility; selectivity; siting; reliability, maintenance and calibration; potential influences of outdoor air; visibility and use of output data; economic viability. This is particularly important when low to medium cost sensors are used, and where the output data may be used to signal an alarm or non-compliance with an imposed maximum concentration of an pollution parameter.

There is recently published guidance on assessing indoor air quality by the World Health Organisation (WHO) and the Institute of Air Quality Management (IAQM). WHO (2020) reviewed a number of methods for sampling and analysis that have been used to characterize indoor air pollutants. They explored the advantages and drawbacks of some of these methods which were selected using the following criteria: a. most commonly used for sampling and analysis of indoor air pollutants; b. recommended by the International Organization for Standardization (ISO) (as the first choice); c. enable quantitative analysis of the concentrations of chemicals and other pollutants.

The Institute of Air Quality Management (IAQM, 2021) published guidance to assist its members in the assessment of indoor air quality (IAQ), in terms of monitoring, modelling and mitigation, in residential and non-residential buildings.

8.1 Indoor PM measurements

The measurement of indoor PM employs methods which are largely based on the same principles of measurement as utilised for the measurement of PM in outdoor environments. For example, ISO 16000-37: 2019 is based on the principles of the outdoor air quality measurement standard for PM: EN12341. This is a gravimetric – filter-based measurement approach which utilises an air pump with sampled air being drawn through a filter medium, the increase in mass of which is attributed to the PM mass captured during the sample period. Understanding the sampling rate and the pre-exposure filter weight to calculate the change in mass provides for a mass per unit volume of air sampled to be determined. The extent to which differing PM mass fractions are measured is determined through the use of a size selective cut off on the sample inlet – the absence of which will provide a measure of the total suspended particulates in air.

The gravimetric approach is typically deployed to understand the PM mass concentration in air and gives rise to a fixed period mean for the duration of sampling e.g.1-hour, 8 hour or 24-hours depending on the focus on the measurement strategy in place. ISO 16000-37:2019 provides for details on the type of low-volume sampler to be utilised, the flow rate and the filter media which should be deployed in the monitoring strategy. Such considerations provide for quality assurance and control in the programme of monitoring which adheres to such standards and provides for confidence in the outcome of the monitoring undertaken.

Although regarded as the more accurate of PM measurement methods some of the disadvantages of use include the time taken to analyse the filters as a result of the controlled exposure conditions required, and the delays in achieving the results, which can lead to delays in the findings of the programme. Moreover, the use of gravimetric sampling methods is expensive and provides the result for a single sample duration in time. As a consequence, high temporal resolution methods have been established which provide for a more continuous reading and instantaneous measurement result. Such methods are not without their own challenges as accuracy is generally regarding as being less than that of the gravimetric methods. Typically, continuous methods do not undertake a direct measurement of PM mass but determine PM mass through the use of algorithms based on differing characteristics of the particles including number, size, surface area or charge. Algorithms within the firmware of the instruments convert the measurement of these characteristics into mass based on relationships determined through controlled laboratory trials or field trials.

It should also be noted that continuous-reading monitors will also be subject to the same problems that affect all particle samplers. The efficiency of the instrument inlet which extracts particles from air within the indoor environment varies dependent on the particle size and the air-flow characteristics of the environment in which it is sampling (which is likely to be low for indoor measurements). Significant particle losses may occur to the internal walls and surfaces of the instrument between the inlet and the sensing zone. The detection methods used at the sensing zone will also have an efficiency of their own - for example, most light-scattering devices can detect particles only within the approximate size range of 0.3 to 10 µm. Also, depending on the method, some sensors may be sensitive to the water present in the particles, which may create undesirable artefacts and may need to be removed (e.g. through heating).

Types of direct-reading airborne particle monitors are given below:

Optical (light-scattering). Light-scattering photometers are multiple-particle sensing zone instruments, measuring the size and number of any particles passing through the sensing zone. Light scattered in the sending zone falls on to a receptor which is positioned off the optical axis. Instruments measure the total amount of scattered light over a specific solid angle either in totality, as pulses corresponding to individual particles, or both. The light scattering pattern is related to particle diameter, refractive index, particle shape etc. Particles only capable of detecting the total amount of scattered light will report an indicative mass concentration for a given PM definition set by a size-selective inlet (e.g. $PM_{2.5}$, PM_4, PM_{10}), whereas those detecting individual particles can report size-distributed number and mass concentrations.

These instruments generally operate in the range of approximately 0.3 to 10 µm particle volume-equivalent diameter (although some research-grade instruments are capable of measuring outside this range), with their ability to detect particles within this size range diminishing at both the lower and upper size ranges. Outputs displayed in terms of mass concentration require certain assumptions to be made about particle density, refractive index, shape etc. and that the characteristics of the particles detected remain constant. For this reason, it may be necessary to truth this to a more authentic measurement, e.g. gravimetric. Also, some of the measurements are strongly affected by hygroscopic growth of the particles, which means they may be unsuitable for high-humidity environments, such as bathrooms and kitchens. Despite these limitations related to determining the mass concentration of particles in the air, these types of monitors remain the most practical to use for surveying of air quality in buildings and other areas - due to their portability, detection range and rapid response. Increasingly miniaturisation of optical methods is giving rise to more portal and low-cost sensors available to the market. Appraisal of such sensors varies from study to study with collocation of emerging sensors with existing outdoor PM measurement methods being typical (Badura, *et al*. 2018)

Condensation particle counters: These instruments count particles by growing them in a working fluid (normally butanol, although less dangerous alternatives exist such as water) before counting optically. These are of much use when counting very small particles (less than 100 nm) that are difficult to detect with purely optical instruments. However these are often expensive, or use chemicals unsuitable for use in an indoor environment.

Light attenuating: These devices are suited to more ‘industrial-type’ applications (cross-stack or cross-duct measurements to monitoring discharges from chimney stacks etc.) They are not suitable for routine air quality measurements in buildings.

Filter-based light attenuation: The rate of change of light attenuation through a filter as air is drawn continuously through it can be taken as a measure of the black carbon (BC) mass concentration in the air. This is the principle used by the Aethalometer for outdoor air quality monitoring, and there are now miniaturised versions of this technology that are suitable for measurements of indoor air quality and personal exposure.

ß-Attenuation: These instruments measure the mass of particulate in the form of the attenuation of radiation from a β source by airborne particles collected on a thin film of filter-capturing media, typically in the form of a tape that move forward on a spool on an hourly basis. While performing the analysis in situ, they do not measure particle concentrations in real time, are typically not very portable and require a radioactive source. As such, they are also not suited to undertaking routine air quality measurements in buildings.

Resonant oscillation mass monitors: The most common of this type of instrument is the TEOM-type monitor (Tapered Element Oscillating Microbalance). Airborne particles are deposited on to a small filter fitted to the top of a tapered glass element. As the mass of particles deposited on the filter increases the resonant frequency changes at a rate proportional to the deposited mass. While this type of instrument does measure airborne particle mass directly, they are large instruments and thus not portable, and therefore not suitable for routine air quality measurements in buildings. In recent years the Filter Dynamic Measurement System (FMDS) has improved the TEOM method in order to overcome problems of heated sampling chambers giving rise to the loss of volatile fractions of PM sampled. This provides for a dual measurement of mass and non-volatile fractions but remains a non-portable instrument, still presenting challenges for instrument deployment in buildings.

Electrical mobility: Scanning Mobility Size Spectrometers are based on the movement of airborne particles that carry a known electrical charge towards an electrode of opposite charge before counting with a condensation particle counter. Electrical mobility is primarily used for the measurement of airborne particles in the approximate size range of 0.01 to 0.5 µm. While useful for number concentrations, this range is therefore unsuitable for measuring the larger particles that are present within, for example, both the $PM_{2.5}$ and PM_{10} particle size mass-fractions. Furthermore, these instruments typically take minutes to complete a scan, which means that the highly dynamic nature of indoor aerosols will often confound the measurement.

Charge detection: Particles moving in an airstream develop an electrical charge, the magnitude of which is dependent on the size, shape and composition of the particle. This type of instrument is primarily used for industrial cross-stack or cross-stack monitoring. They are not suitable for routine air quality measurements in buildings.

Charge coupling: This is another detection method based on the measurement of electrical charge carried by particles, but without collecting the particles on a sensor surface. A clean air supply is ionised before it enters the sensor. This ionised air is then used to charge the particles in the sensor, with the charging of the particles being relative to their size. This forms the basis for a number of portable instruments that are particularly suitable for the study of ultrafine particles in particular, delivering measures of particle number concentrations and surface area.

The extent to which continuous monitors are required to achieve data quality objectives in indoor environments is, at present, unclear, with the exception of the gravimetric approach set out in ISO 16000-37: 2019. Moreover, the extent to which continuous PM measurement methods are deployed systematically and utilised to evidence levels of PM in indoor environments is unclear. Indoor clean environments such as data centres and operating theatres utilise ventilation systems with filters in place to capture the PM prior to entry of the “clean air” into the internal rooms being utilised. As such, the need for active continuous monitoring is negated and any checks on PM levels in the internal spaces which arise through concerns are generally checked through “spot measurement” for short duration period sampling.

8.2 Gas sensors for buildings

There are a vast range of different analytical techniques that can be applied to the measurement of gas phase pollutants based on methods such as absorption spectroscopy, mass spectrometry and chromatography. Such techniques are accurate, sensitive and traceable, but are often very expensive to operate, physically bulky, noisy and power intensive. They are rarely applicable to long-term measurements for building management systems. Instead, a range of lower cost and simpler techniques can be applied, but they inevitably come with trade-offs in terms of measurement performance. The key types of sensors for indoor monitoring and their principles of operation are given in this section.

8.2.1 Electrochemical sensors

Electrochemical sensors exist for a wide variety of pollutants including CO, CO_2, NO_2 and O_3 and operate on the basis of oxidative or reductive reactions between the pollutant of concern and the electrolyte. Such reactions generate positive or negative current to an electrical circuit, the magnitude of which is registered according to the concentration of the pollutant in the air. Electrochemical sensors comprise of a “working electrode” and “counter electrode”, and usually a reference electrode. The electrodes sit inside a housing alongside a liquid electrolyte. On the housing a mechanism exists which comprises of a membrane and a diffusion limiting outlet, which allows the outside air to interact with the liquid electrolyte.

Electrochemical sensors work on the basis of gaseous diffusion. The oxidative or reductive reaction generated by the diffusion of gas depends on the type of pollutant of concern: for example, CO is oxidised into CO_2 and the oxygen is reduced to water. Oxidation causes a flow of electrons to move from the working electrode to the counter electrode via the external circuit. A reduction reaction causes a flow of electrons in the opposite direction (namely, from the counter electrode to the working electrode). The electron with the electric circuit detects and amplify the current, which is then scaled according to the calibration given to the sensor to produce a reading in the relevant engineering units (typically in part per million (ppm).

Gaseous sensors vary widely in their minimum resolution and accuracy with a number collating the results of a wide range of studies (Paleologis *et al.* 2021 and Zhang and Srinivasan, 2020).

8.2.2. Metal Oxide Semiconductor Sensors

Metal oxide semiconductor sensors are relatively inexpensive compared to other sensing technologies and benefit from higher detection sensitivities and quicker response times to fluctuations in gas concentrations. For these reasons they have been utilised in monitoring trace levels of pollutants in air.

The principle of operation of this type of sensor is absorption of desorption of a gas on the surface of a metal oxide, which gives rise to changes in the conductivity of the material. In clean air, donor electrons in the metal oxide are attracted toward oxygen which is adsorbed on the surface of the sensing material, preventing electric current flow. In the presence of reducing gases, the surface density of adsorbed oxygen decreases as it reacts with the reducing gases. Electrons are then released into the metal oxide allowing current to flow freely to the sensor.

A comprehensive handbook on metal oxide semiconductor sensors available for air pollutant detection is published in “Semiconductor Gas Sensors, Second Edition, Elsevier Ltd, 2020. Although focused on outdoor air, performance and applications of electrochemical sensors were included in a WMO (2018) review.

8.2.3 Non-dispersive Infrared (NDIR) Sensors

When infrared radiation interacts with gas molecules infrared light is absorbed at a particular wavelength. Non-dispersive infrared sensors detect decreases in the transmitted infrared light onto a sensor resulting from the presence of a gas molecule which is proportionate to its concentration in air.

NDIR sensors consist of an infrared source, detector, optical filter, gas cell and electronics for signal processing. A single light source, dual wavelength type gas sensor has two detectors and two optical filters of different wavelengths which are placed in front of each detector. Infrared light that is absorbed by a target gas passes through the active filter with a particular bandwidth for the detection of the target gas. Infrared light that does not interact with the target gas passes through the reference filter. The difference between transmitted light intensities in these two bandwidths is converted into gas concentration. The dual wavelength sensor ensures stable measurements for a long period of operation as the aging effects of the light source or the gas cell are automatically compensated by output signals at the reference wavelength.

8.2.4 Photo-ionization Detection Sensors

PID sensors are used extensively for the measurement of VOCs. PIDs use an ultraviolet (UV) light source to fragment VOCs in the air sampled into positive and negative ions. The

PID measures the charge of the ionized gas, with the charge being a function of the concentration of VOCs. After ionization and measurement, the gas ions reform back to the original gas composition. PIDs do not therefore burn or change the composition of the gas sampled. Different lamps for ionisation are available which provide for measurement of different VOCs by delivering different wavelengths of light with different abilities to split apart different VOCs with different functionalities. Krypton; xenon and argon lamps are widely used each providing a slightly different set of VOCs to which the sensor responds. The response of a PID is dependant not only on the concentration of the VOC in air but the ease of ionisation of that compound. As a result the charge measured by the device can be influenced by the speciation of the air being sampled. PIDs typically express a measurement as a ‘total VOC’ concentration, but this rarely a well characterised absolute value. As such PIDs can provide a reasonable guide for trends and fluctuations in VOCs indoors, but do not generate data in a form that allows for comparison against individual VOC air quality guidelines.

At the present time small-scale/low-cost sensors or monitors for inorganic gases such as NO, NO_2 and O_3 in the ppb range, needed for assessing indoor air, are not routinely used, although they are now commonly used outdoors for indicative measurements of air quality.

8.3 Approaches for VOC assessment

A WHO (2020) report described analytically protocols for sampling and analysing those VOCs and SVOCs which are most commonly found in indoor settings (with particular focus on schools). In summary, for oxygenated VOCs (formaldehyde and acetaldehyde), the sampling protocol is defined in detail in ISO 16000-2:2004 and can be applied to the other aldehydes. The determination of formaldehyde can be done using a diffusive sampler with solvent desorption and high-performance liquid chromatography (HPLC) and the analytical procedure is available in ISO 16000-4:2011. This standard was reviewed and confirmed in 2017 and is still valid.

For VOCs, the detailed sampling protocol is given in ISO 16000-5:2007. General guidance for the sampling and analysis of volatile organic compounds (VOCs) in air is given in ISO 16017-2:2003 and is applicable to indoor, ambient and workplace air. This standard describes the analytical procedure by sorbent tube / thermal desorption (TD) / capillary gas chromatography, which is applicable to a wide range of VOCs, (including hydrocarbons, halogenated hydrocarbons, ester, glycol ethers, ketones and alcohols), ISO 16017-2:2003 was reviewed and confirmed in 2019 and is still valid.

If thermal desorption is not available, sorbent tube can be a low-cost alternative for analysing VOCs. The methodology can be found in EN 14662-5 and is specific for benzene, but it can be applied to a range of VOCs. Adsorbent methods, either passive or pumped are however limited to those VOCs that are sufficiently involatile that they can be quantitatively trapped during sampling. This often excludes very volatile VOCs in the range C_2-C_4. Given that species such as ethanol, propane and butane can be the most abundant VOCs in UK homes (see Table 6.3), this a notable deficiency in methods. An alternative is to collect whole air samples, using methods analogous to the TO-19 EPA methods used outdoors. This approach collects air into a stainless steel vessel which is then returned to a lab for analysis. This is non-selective and allows for the most volatile VOCs to be measured. An example of this method used in UK homes is reported in Heeley-Hill *et al.*, 2021.

SVOCs (classified by WHO as organic pollutants with boiling points ranging from 240–260 °C to 380–400 °C) can be found either in the air (gas and particulate phase) and surfaces (settled dust). The sampling strategy for polychlorinated biphenyls (PCBs), polychlorinated dibenzo-p-dioxins (PCDDs), polychlorinated dibenzofurans (PCDFs) and polycyclic aromatic hydrocarbons (PAHs) is given in ISO 16000-12:2008 (reviewed and confirmed in 2016). The sampling and preparation of sampling media for dioxin-like polychlorinated biphenyls (PCBs), polychlorinated dibenzo-*p*-dioxins (PCDDs) and polychlorinated dibenzofurans (PCDFs) in indoor air is given in ISO 16000-13:2008 (reviewed and confirmed in 2018). Passive sampling of SVOCs was initially developed to collect them in the gas phase; however, it has been widely used to report concentrations of SVOCs associated with particles (WHO, 2020).

For polycyclic aromatic hydrocarbons (PAHs), the sampling strategy is available in ISO 16000-12:2008 using a low-volume sampler (LVS). For the gas/particle phase, the protocol for the analysis is available in ISO 12884:2000 (for ambient air) with collection on sorbent-backed filters with gas GC-MS analysis. This standard was revised and confirmed in 2016 and is still valid. For the particle phase, ISO 16362: 2005 for ambient air determines the particle-phase polycyclic aromatic hydrocarbons by high performance liquid chromatography. The standard was reviewed and confirmed in 2016 and is still valid.

Very recently, the revised ISO 16000-6:2021 refers to organic compounds (VVOC, VOC, SVOC) in indoor air and in air sampled to determine emissions from products or materials used in indoor environments using test chambers and test cells. The method uses sorbent sampling tubes with subsequent thermal desorption (TD) and gas chromatographic (GC) analysis employing a capillary column and a mass spectrometric (MS) detector with or without an additional flame ionisation detector (FID). Depending on the sorbents used, many very volatile organic compounds (VVOC) and semi-volatile organic compounds (SVOC) can be analysed.

8.4 Current approaches for biological particles.

Current approaches to assaying indoor microbial bioaerosols comprise methods for firstly their capture and secondly their enumeration using culture-dependent and culture-independent approaches. Due to the highly complex and heterogeneous nature of indoor bioaerosols, there exist a broad range of techniques for their sampling, characterisation and quantification. These diverse methods are often reactively applied ad hoc when risks to health by bioaerosols are perceived. Due to this diversity of approaches, there is a need for standardised approaches that can lay down baseline expectations for indoor bioaerosol quantity and composition, and there is also a matched need for better understanding of the relationship between bioaerosol exposures and health impacts.

Traditionally microbial diversity in homes has been studied using culture-based methods from surface samples that is complemented by a trend to use nucleic-acid barcoding approaches for microbial characterisation (Barberan *et al.* 2015; Fu *et al.* 2020). However, while informative, these studies have limited to ability to determine the amount and type of indoor bioaerosols as dust/surface samples do not describe what microbial components are airborne and respirable, and here specific techniques are needed. A further complication is that only a subsample of microbes that are recovered from indoor air are culturable, for example the ratio of total fungi to viable fungi has been estimated to be as low as 100:1 (Toivola *et al.* 2002).

Widely used methods for sampling airborne microbiota is either by impaction, impingement and filtration. The simplest approach is to let gravity deposit particles onto settle plates (passive air sampling). Settle plates are an inexpensive method for detecting whether microbial spores are present, for example as contaminants in hospital theatres (Napoli *et al*.,2012) and their simple usage has allowed them to be used in a UK-wide Citizen Science projects to monitor *Aspergillus fumigatus* bioaerosols in homes (Shelton, Fisher, and Singer 2020). However, to directly monitor bioaerosols that are within the inhalable/respirable size range as defined by ISO conventions (ISO 7708; 1995), active filtration of air is required through the use of machines. Impactors are widely used for sampling spore-forming bacteria and fungi, and rely on actively drawing air currents directly over collection surfaces. Collection is then followed by their enumeration either directly by culture as colony forming units per m^3 of air (CFU / m^3) or onto adhesive tapes for subsequent microscopy or nucleic-acid characterisation. There exists a range of widely used commercially available devices that are used in indoor environments ranging from homes to hospitals including the widely-used 6-stage Andersen cascade impactor which can separate culturable bioaerosols depending on their size distribution (Ghosh *et al.*, 2015).

Impingement methods use the same approach as impaction, except that particles are collected into a liquid rather than onto a solid medium followed by enumeration as culturable CFU/m^3 or downstream nucleic-acid characterisation. Filtration methods rely on actively drawing defined volumes of air through filters of various types with a typical pore size of 0.8 µm, which is then processed in liquid to release collected material for culturable organisms or their nucleic acids. While most widely used for capturing and enumerating spore forming bacteria and fungi, all three methods (impaction, impingement and filtration) can be adapted to detect airborne viruses (Bekking *et al*. 2019) and have widely been used for assessing exposure to viable SARS-CoV-2 – containing aerosols in a variety of indoor settings (Lednicky *et al.* 2020; Borges *et al.* 2021).

Increasingly, there are a broad range of emerging technologies that combine both the capture and enumeration steps into real-time characterisation of biological aerosol particles (Huffman *et al*. 2020). This includes particle counters that characterise biological particles in real time through their optical scattering and ultraviolet fluorescence properties rather than culturing (Crawford *et al.* 2020). However, these methods mostly need further technical validation standardisation before widespread use is possible.

Generally speaking, there is a lack of internationally standardised quantitative methods for assessing exposure to indoor bioaerosols (Ferguson *et al*. 2019). Where relevant, approaches have been developed driven by the need to assess exposure to specific high-risk bioaerosols in occupational settings; these are detailed in Table 8.2. For example, exposure limits to indoor LPS endotoxins have been set at 90 endotoxin units / m^3 according to the Dutch NEN-NE14032 methodology (DECOS 2010), while wood (3-5 mg m^{-3}; 8-hr average; (HSE 2021)) and grain dust (10 mg m^{-3}; 8-hr average; (HSE 2013)) exposures are similarly prescriptive. Developed for outdoor composting environments, the Environment Agency M9 protocol (Environment Agency 2018) details a standardised approach for monitoring the thermophilic bioaerosols *Aspergillus fumigatus* and mesophilic bacteria, and the principles of this protocol are of relevance to assessment of these organisms in indoor air. However, there remains a pressing need to develop UK-relevant standardised methodologies and baselines in order to understand ‘normal’ levels of exposure to bioaerosols in domiciles. Owing to their complex polymicrobial nature (fungi, bacteria and viruses) as well as the occurrence of high-risk individuals, this is a challenging task.

The clear links between dampness, exposure to moulds and their ensuing health impacts has led to extensive investigation (reviewed by (Eduard 2009)) leading to the WHO publishing guidelines for Dampness and Mould (WHO 2009). While not validated for routine use, the USA-developed quantitative Environmental Relative Moldiness Index (ERMI) assesses fungal burden as an ERMI scale ranging from -10 (lowest) to +30 (highest) mould contamination (Vesper *et al*. 2018). Based on mould specific quantitative PCR (MSQPCR), ERMI analyses the presence of 36 indicator-moulds, and positive values of the index have been epidemiologically linked to clinical syndromes such as asthma ((Vesper and Wymer 2016)). However, as no health-based exposure limits are specified for moulds, then interpretation of measurements by various methods are largely qualitative with respect to health outcome.

Table 8.2 Summary of methods for the measurement of air biological pollutants indoors.

Bioaerosol	Monitoring method	Indoor/Outdoor	Standard exposures	Country
Endotoxin; lipopolysaccharides LPS	NEN-EN14031 method	Indoor air sample	Exposure limit: 90 endotoxin units EU / m^{-3}	Netherlands
Mould	Mould-specific qPCR	Indoor settled dust	Environmental Relative Moldiness Index (ERMI)	USA
Mould (Aspergillus fumigatus), mesophilic bacteria	Impaction, filtration, impingement	Outdoor	Colony Forming Units CFU / m^3	UK & EU

Wood dust	IOM inhalable sampler	Indoor	2-5 mg m^{-3}	USA, EU
Grain dust	Gravimetric analysis	Indoor / outdoor	10 mg m^{-3}; 8-hr average	UK & EU

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Appendix 1

Some Factors Determining Indoor Pollutant Concentrations

Determination of the concentration of a pollutant at a location indoors depends on many factors. It requires details of the location and time dependence of the sources and sinks of the pollutant, and details of the many factors which determine the airflow and turbulence and hence dispersion of the pollutant. These data may then be used together with complex equations to determine the flow, dispersion and chemical reactions taking place[3]. Because of the complexities involved and computational requirements, such studies are typically limited in scope and not able to examine sensitivities to the wide range of different input parameters, nor are they easy to interpret.

An alternative simplified approach, which is widely used and is straightforward to apply to a wide range of conditions, is a box or zonal model which uses as its basis conservation of mass together with the simplifying assumption that pollutant is well mixed in each room[4] [5] [6]. This is a good assumption if the timescale for mixing in the room (τ_m) is short compared to the timescales for variations in the inner source emission rate and indoor sinks and timescales for air exchange (τ_e), since then the contribution of unmixed pollutant (i.e. the concentrations arising from the 'plume' of material recently released indoors or entrained from outside) relative to well mixed pollutant is small. Since $\tau_m \sim L/\sigma$, where L is the length scale of the room volume V and σ the rms turbulent velocity, and $\tau_e \sim V/\alpha_e$, where α_e is the rate of exchange of outdoor air, then for typical values, for example L=6m, V= 90m^3, σ=0.1m/s and one air exchange per hour, τ_m ~ 60s and τ_e ~ 3600s.

In this appendix we use this simplified approach to derive some approximate equations for the indoor concentration and indoor outdoor (I/O) ratio dependent on the sources, sinks, ventilation rate and outdoor concentration. The key processes and parameters are represented in Figure A1 below.

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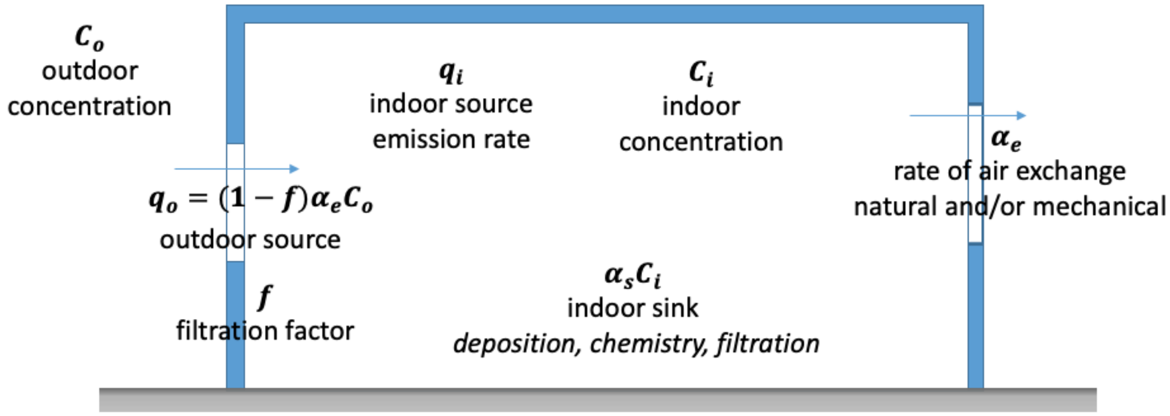

Figure A1. Pictorial representation of the key processes and parameters that influence tha concentrations of air pollution found indoors.

In order to make the analysis straightforward we consider a single room of a building with indoor–outdoor air exchange but no room-to-room exchange (i.e. box model rather than zonal model). This indoor exchange could be included, but will not change the essence of the analysis presented here. The (uniform) indoor concentration of a pollutant, C_i, in the room volume can be estimated from the following factors: indoor source(s) emission rate; outdoor source(s), governed by the rate of ingress of outdoor pollutant; and sinks caused by chemical reactions (also sources), deposition and ventilation. If we also make the assumption that both chemical loss (or gain) and deposition are proportional to the concentration, which means that any non-linear chemistry is not considered, then from the conservation of mass we obtain:

$$V\frac{dC_i}{dt} = q_i + q_o - (\alpha_e + \alpha_s)C_i \qquad \text{(A1)}$$

Where,

q_i is the indoor source emission rate;

q_o is the source strength of pollution entering the room from outdoors; for each inlet this equates to $(1 - f)\alpha_e C_0$ where C_0 is external concentration at the outdoor air inlet(s), and *f* is a filtration factor (i.e. the fraction of pollutant that is filtered on entering the building);

α_e the rate of exchange of outdoor air (m^3/sec), equivalent to *EV*/3600 where *E* is the number of air exchanges per hour in the room.

α_s the equivalent rate of volume loss or gain due to indoor sinks (sources) e.g. chemistry, deposition and indoor filters. The actual rate of loss $\alpha_s C_i$ is assumed to be proportional to C_i.

If we also assume that the source emission rates and rates of air exchange are constant in time then this equation can be solved analytically. For $C_i = 0 \; at \; t = 0$ it has solution

$$C_i = \frac{(q_i+q_o)}{(\alpha_e+\alpha_s)}\left\{1 - exp\left(-\frac{(\alpha_e+\alpha_s)}{V}t\right)\right\} \tag{A2}$$

For small t this reduces to

$$C_i = \frac{(q_i+q_o)}{V}t \tag{A3}$$

showing the concentration initially increases with t when the sources are 'turned on'. For steady state conditions the indoor concentration is given by:

$$C_i = \frac{(q_i+q_o)}{(\alpha_e+\alpha_s)} \tag{A4}$$

This formula provides useful linkages between steady state indoor and outdoor concentrations and source strengths, and ventilation (air exchange) rate and sinks. Since q_o is proportional to C_0, A4 can be used to estimate indoor outdoor ratios:

$$\frac{C_i}{C_o} = (1-f)\frac{1+q_i/[(1-f)\alpha_e C_o]}{1+\alpha_s/\alpha_e} \tag{A5}$$

If the indoor source q_i is turned off at $t = 0$ and the concentration is already in equilibrium (A4), then

$$C_i = \frac{q_i}{(\alpha_e+\alpha_s)}\exp\left(-\frac{(\alpha_e+\alpha_s)}{V}t\right) + \frac{q_o}{(\alpha_e+\alpha_s)} \tag{A6}$$

(A6) can be used to show how indoor concentrations return to equilibrium with outdoor concentrations, for example the fall in CO_2 concentrations when people leave a room they have been occupying.

Table A1 shows some examples of the application of equation (A4) for the steady state conditions to which the formula applies: i.e. indoor source strength, outdoor concentration constant on the time scale of air exchange which is also assumed to be constant.

We consider a room with volume 45 m^3 (typical living room 4.5 × 4 × 2.5m). The first two rows in the table are for an indoor source with zero outdoor background, and losses, either to ventilation or to indoor sinks, equivalent to one air change per hour. Even with a source emission rate of 1 $\mu g \; s^{-1}$ which is low (relative to source emission rates of outdoor sources typically considered), noteworthy concentrations are achieved because of the relatively small volume of air into which the emitted material disperses. For reference, the maximum plume concentration 2m downstream of a point with emission rate 1 µg/s in neutral conditions in a 1m/s flow depends on turbulence levels but is typically more than an order of magnitude lower than the 80 $\mu g/m^3$ in the room – this shows that it is the build-up of well mixed pollutant in the confined indoor space which determines almost all the concentration, not recently released unmixed pollutant.

The next seven rows of the Table consider a range of internal sources and sink strengths for an external concentration of 40 µg m^{-3} typical of NO_x or NO_2 for a ventilation rate ($\alpha_e = 0.0125$ m^3 s^{-1}) equivalent to one air change per hour. With no indoor source or losses, the concentration in the steady state is the same as that outdoors (C_i/C_0 =1). As anticipated indoor losses decrease the ratio whereas indoor sources increase the ratio. In the final row, the outdoor concentration is set close to the regional background for CO_2 and the indoor emission rate corresponds to CO_2 emissions from three people (~1 kg per day per person). In this case C/C_o = 4.6 which is in the range of elevated CO_2 concentrations observed indoors in rooms occupied by people.

C_o(µg m^{-3})	**α_e(m^3 s^{-1})**	**f**	**q_o (µg s^{-1})**	**q_i(µg s^{-1})**	**α_s(m^3 s^{-1})**	**C_i (µg m^{-3})**	**C_i/C_o**
0	0.0125	0	0	1	0	80	N/A
0	0	0	0	1	0.0125	80	N/A
40	0.0125	0	0.5	0	0	40	1
40	0.0125	0.5	0.25	0	0	20	0.5
40	0.0125	0	0.5	0	0.00625	26.7	0.67
40	0.0125	0	0.5	0	0.0125	20	0.5
40	0.0125	0	0.5	1	0	120	3
40	0.0125	0	0.5	1	0.00625	80	2
40	0.0125	0	0.5	1	0.0125	60	1.5
770 mg m^{-3} 420 ppm (CO_2)	0.0125	0	9.6 (mg s^{-1})	35 (mg s^{-1})	0	3570 mg m^{-3} 1950 ppm (CO_2)	4.6

Table A1. Example application of the steady state equation A4. Volume of room is 45 m^3; α_e= 0.0125 m^3 s^{-1} is equivalent to one air exchange per hour; α_s = 0.00625 $m^3 s^{-1}$ equivalent to loss of 50% of pollutant (due to deposition or chemical reaction) in one hour. C_i is the indoor concentration so C_i/C_0 is the indoor-outdoor ratio of concentrations.

Examples of Applying the Approximate Formulation to data from the Main Report

1. Household VOCs

Emissions of VOCs and their concentrations are discussed in Section 1.3 of the report. Specifically Figure 1.1 (b) shows concentrations of a range of VOCs measured over three-day periods in summer and winter in 60 UK homes. Concentrations of up to a few hundred micrograms per cubic metre were measured for the most abundant species (n-butane and propane). Figure 1.2 shows estimates of total UK emissions of VOCs from a range of aerosolised personal care products. Current emissions are estimated as approximately 80 kT of which more than 90% (~ 70kT) are estimated to be released in the home.

The broad consistency between the measured concentrations and the estimated emissions can be assessed by applying equation A4 as follows: assume 27.6 million households[7] and 70 kT emissions per annum and hence an average emission rate per household of 80 µg s^{-1}; assume an average dwelling surface area of 75 m^2 [8], room height 2.4 m and hence volume 180 m^3. Applying A4 and assuming negligible contribution from the outdoor concentrations and no indoor loss gives a concentration of 1600/*E* (µg m^{-3}), where *E* is the number of air exchanges per hour. (The assumption of no appreciable indoor chemical loss is reasonable for relatively slow reacting and dominant species such as butane and ethanol where OH would be the major oxidant. In an indoor environment these VOC would have a oxidation lifetime of the order 2-5 days). This gives concentrations broadly consistent with those measured for about two exchange rates per hour (i.e.~ 800 µg m^{-3}); lower rates of air exchange would be consistent with indoor losses.

2. Indoor/Outdoor ratios

In Section 3.1.2 of the report there is a discussion of Indoor to Outdoor (I/O) ratios for NO_2 and $PM_{2.5}$ for around 1.6 million spatially-referenced London dwellings estimated by Taylor et al (2019) using a building physics metamodel (based on large number of simulations of EnergyPlus). Key assumptions in their modelling were that fractional losses per hour due to deposition were 0.87 for NO_2 and 0.19 for $PM_{2.5}$, whilst 10% of outdoor $PM_{2.5}$ was filtered in the heating season. With these assumptions the I/O ratios for NO_2 were calculated to be: mean: 0.4, range: 0.3 - 0.6, and for $PM_{2.5}$ (mean: 0.6, range: 0.5 - 0.7); similar I/O ratios have also been predicted by the INDAIR/EXPAIR modelling framework *(*Dimitroulopoulou *et al.*, 2006).

Note that from A5 that ratio I/O < 1 implies that the rate of ingress of outdoor air pollutant ($(1-f)\alpha_e C_0$) is greater than the indoor source emission rate.

Using A5 with similar assumptions to Taylor *et al.* for filtration and deposition (*f* = 0.05 on average for $PM_{2.5}$, fractional losses per hour due to deposition of 0.87 for NO_2 and 0.19 for $PM_{2.5}$), then I/O ratios of 0.4 (NO_2) 0.6 ($PM_{2.5}$) imply air exchange rates per hour of 0.58 and 0.33 for NO_2 and $PM_{2.5}$ respectively. In reality these exchange rates must be the same: for a typical dwelling volume of 180 m^3 and an outdoor NO_2 concentration of 20 µg m^{-3}, it requires an indoor source of NO_2 of 0.15 µg s^{-1} to reduce the consequent exchange rate for NO_2 to

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[7] ONS: Families and households in the UK:2018 https://www.ons.gov.uk/peoplepopulationandcommunity/birthsdeathsandmarriages/families/bulletins/familiesandhouseholds/2018

[8] Which, 2018 https://www.which.co.uk/news/2018/04/shrinking-homes-the-average-british-house-20-smaller-than-in-1970s/

0.33 air exchanges per hour, the value corresponding to $PM_{2.5}$. Introduction of $PM_{2.5}$ sources implies even lower exchange rates and increased indoor NO_2 emissions. Such rates of air exchange consistent with those presented for new builds in Table 4.1 of the main report.

3. Primary Particles emitted indoors

At section 3.1.3 of the report there is a discussion of indoor particle concentrations. Vu et al. (2017) measured nanoparticle emissions arising from five indoor sources. Maximum number concentrations generated in an apartment were 9.38×10^4, 1.46×10^5, 2.89×10^4, 2.25×10^5 and 1.57×10^6 particles cm^{-3} for particles released from vacuum cleaning, soap cleaning spray, smoking, incense burning and cooking (frying) activities, respectively. Many studies have reported particle number counts exceeding 10^5 cm^{-3}, with modal particle diameters from 20 – 150 nm diameter (Abdullahi *et al.,* 2013).

Assuming no significant contribution from outdoor sources or loss from sinks on the timescale of the emissions then from A4 the particle emission rate can be estimated approximately from

$$q_i = \alpha_e c_i$$

which gives for an average dwelling (volume 180 m^3)

$$q_i = 0.05 E c_i$$

For the range of particle number concentrations given above (~10^{11} to 10^{12} particles m^{-3}), this equates to particle release rates in the range 5×10^9 to 5×10^{10} particles s^{-1} for one air exchange per hour.

4. CO_2 in schools

Figure 3.3 in section 3.2.1 of the report shows time series of CO_2 from a Naturally-ventilated Secondary School Classroom. These show significant variation during the day associated with changes in occupancy of the rooms and changes in air ventilation; typical values are about 1000 ppm in winter and 650 ppm in summer. By making some simple assumptions we can show that the measured concentrations are broadly consistent with equation A4 for reasonable air exchange rates. We assume a classroom area of 60 m^2[9], height 2.4 m and hence volume 144 m^3; outdoor CO_2 concentration of 420ppm and 30 pupils in the class room each emitting CO_2 at 5.8 mg s^{-1} equivalent to 0.5 kg day^{-1} (half a typical adult value, see section 1.7). Then from A4 the concentrations of 1000 ppm in winter equate to about 2.5 air exchanges per hour. The measured concentrations show that exchange rates in summer are higher. After school hours in winter the fall off is broadly consistent with equation A6 with one air exchange every few hours.

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[9] Building Bulletin 98; Briefing Framework for Secondary School Projects. http://www.educationengland.org.uk/documents/pdfs/2004-building-bulletin-98-sec.pdf

Appendix 2

Table A1: Estimates of UK emissions (kilotonnes) of NMVOC in 2019 occurring indoors in domestic residences and other buildings based on sectorial data from the NAEI. See Section 6.1

ktonnes	**Indoor residential**		**Indoor other**	
	Emissions	**± *Range***	**Emissions**	**± *Range***
Domestic combustion	0.1	*0.1*	0.0	*0.0*
Non-aerosol products - cosmetics and toiletries	11.4	*0.6*	0.0	*0.0*
Non-aerosol products - household products	11.0	*1.4*	2.8	*1.4*
Non-aerosol products - domestic adhesives	4.3	*0.2*	0.0	*0.0*
Non-aerosol products - paint thinner	5.5	*1.4*	5.5	*1.4*
Aerosols - cosmetics and toiletries	42.4	*2.2*	0.0	*0.0*
Aerosols - household products	5.7	*0.7*	1.4	*0.7*
Decorative paint - retail decorative	12.2	*1.5*	0.0	*0.0*
Decorative paint - trade decorative	0.0	*0.0*	10.2	*1.3*
Cigarette smoking	0.1	*0.0*	0.0	*0.0*
Total indoor emissions	92.6	*± 3.5*	19.9	*± 2.4*
% UK totals occurring indoors	11.3%	*±0.4%*	2.4%	*±0.3%*

Table A2: Estimates of UK emissions (kilotonnes) of NO_x in 2019 occurring indoors in domestic residences and other buildings based on sectorial data from the NAEI. See Section 6.1

ktonnes	**Indoor residential**		**Indoor other**	
	Emissions	**± *Range***	**Emissions**	**± *Range***
Domestic combustion	0.97	*0.39*	0.00	*0.00*
Cigarette smoking	0.05	*0.01*	0.00	*0.00*
Total indoor emissions	1.02	*± 0.39*	0.00	*± 0.00*
% UK totals occurring indoors	0.10%	*± 0.04%*	0.00%	*± 0.00%*

Table A3: Estimates of UK emissions (kilotonnes) of $PM_{2.5}$ in 2019 occurring indoors in domestic residences and other buildings based on sectorial data from the NAEI. See Section 6.1

ktonnes	**Indoor residential**		**Indoor other**	
	Emissions	***± Range***	**Emissions**	***± Range***
Domestic combustion	0.06	*0.02*	0.00	*0.00*
Cigarette smoking	0.68	*0.08*	0.00	*0.00*
Total indoor emissions	0.74	± *0.09*	0.00	± *0.00*
% UK totals occurring indoors	0.66%	± *0.08%*	0.00%	± *0.00%*

Table A4: Estimates of UK emissions (kilotonnes) of NH_3 in 2019 occurring indoors in domestic residences and other buildings based on sectorial data from the NAEI. See Section 6.1

ktonnes	**Indoor residential**		**Indoor other**	
	Emissions	***± Range***	**Emissions**	***± Range***
Non-aerosol products - household products	0.97	*0.12*	0.24	*0.12*
Cigarette smoking	0.10	*0.01*	0.00	*0.00*
Infant emissions from nappies	0.02	*0.01*	0.00	*0.00*
Adult breath & sweat	0.58	*0.19*	0.19	*0.19*
Total indoor emissions	1.67	± *0.11*	0.43	± *0.11*
% UK totals occurring indoors	0.59%	± *0.04%*	0.15%	± *0.04%*