

Air Pollution Modelling from a BESS Fire at Lime Down

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Summary

This paper reviews the conclusions of the Applicant's modelling of concentrations of air pollutants from a BESS fire at Lime Down. The Applicant's modelling concluded that any risks were minimal. This paper doubts these conclusions, which are in fact a consequence of questionable model assumptions, incorrect use of pollutant standards and flawed model implementation. The principal points at issue are: (1) it is assumed that only one model container will catch fire, and there will be no spread to other containers. This ignores experience from other BESS fires, and a reasonable worst-case assumption requires that the fire will spread to all 270 battery containers. (2) The assessment is artificially restricted to a spread distance of 1 km, whereas a pollutant plume from a BESS fire is likely to spread much further, as shown for instance by the fire in a similar-sized BESS in California in 2025. (3) A more realistic assessment based on a 5 km spread would put many more receptors at risk, including nine villages, three primary schools and the M4. (4) Workplace exposure standards are wrongly used to assess the effects of particulate exposure, whereas there are ambient standards which could be used. (5) Pollutant concentrations were not assessed at the closest receptors, which include the main line railway to South Wales. (6) Particle concentrations were assessed using a diesel fire as a model, whereas there are data available to allow assessment using lithium-ion battery fires. This has led to an underestimation of particle concentrations by a factor of 2.5 to 4.6. (7) Not every potentially serious pollutant has been assessed, including very fine particles and heavy vapours. The modelling needs to be repeated using more realistic assumptions and standards, as it does not currently present a reasonable worst-case assessment of the likely significant environmental effects of this aspect of the proposed development at Lime Down.

1. Introduction

1. BESS thermal runaway events, or "fires", can release a wide range of toxic gases. The Lime Down ES includes a model assessment of the likely concentrations and effects of a BESS fire at Hullavington in the Lime Down Scheme. The topic is covered in Chapter 15 of the ES [APP-067]: methodology in 15.6.30 et seq. and results in 15.10.36 et seq. Appendix 15-2 [APP-239] covers the methods and results of modelling in more detail, Figure 15-5 maps the

modelling area [APP-166], and there is an outline Battery Safety Management Plan [APP-286]. The assessment concludes that concentrations of all the gases modelled are insignificant and not harmful to health at the closest receptors. In this paper we demonstrate that the modelling which has led to these conclusions is seriously flawed, and that the risks to people living locally and travelling in the area are likely to be considerably greater. This gives rise to the view that, contrary to para 15.10.42 of Chapter 15 of the ES, that assessed on a reasonable worst-case basis there are likely to be significant effects.

2. Background

2. Typically, for modelling to have credibility it will be underpinned by experimental or at least observational evidence. This section outlines the proposed BESS at Lime Down and how it relates to some BESS failure incidents.
3. The Lime Down BESS would be in a single compound next to the mainline railway north of Hullavington, as described in Chapter 3 of the ES [APP-055] Section 3.3. Appendix 3-1 [APP-182] has more technical detail. The BESS would consist of “approximately” 270 containers each of capacity 5015 kWh. Though the nameplate energy output of the Lime Down BESS on the PINS website is 1000 MWh, 270 containers at this capacity would deliver 1354 MWh of energy.
4. Failure incidents in grid-scale batteries systems are fairly uncommon. The (US) Electrical Power Research Institute (EPRI) maintains a database (see [1]) showing that, worldwide, there were 81 reported incidents between 2018 and 2024. Though infrequent, BESS fires have the potential to be very damaging, because they are difficult or impossible to extinguish and the consensus is that they should be allowed to burn out, which can take hours or days (see [1] for a review). While burning they can produce large quantities of toxic gases and particulates, possibly impacting human health, and spread toxic substances, including heavy metals and persistent organic pollutants, onto soils and waters.
5. The biggest and most extensively documented BESS fire occurred at Moss Landing, California, starting on 16 January 2025 [2]. The facility that caught fire was a 1200 MWh system (there is more than one system on site), so roughly the same size as that planned for Lime Down. The fire burned until 20 January, and briefly re-ignited in February. Between 55 and 80% of the

batteries were destroyed. The cause of the fire has not yet been revealed, and cleanup of the site continues. 1200 people were evacuated, and “shelter in place” orders were issued for people further away. An internationally important wetland is located adjacent to the facility, and a paper in *Nature Reports* [2] described concentrations of nickel, cobalt and manganese from the battery, and the presence of combustion particles, in the surface layers of the wetland after the fire. These contaminants rapidly dispersed due to rainfall and tidal washing. Though no toxic gases were detected at the time, monitoring was understandably directed towards places where most people live, which were fortunately not in the path of the pollutant plume. There are some anecdotal reports of people with rashes, sore throats and headaches [[news article](#)] and exacerbation of existing pulmonary diseases.

6. Photographs of the fire are instructive. Fig. 1 below shows the fire on the first night.



Smoke and flames are seen from Castroville as a fire at the Vistra battery storage plant burns in Moss Landing, Calif., on Thursday, Jan. 16, 2025. (Doug Duran/Bay Area News Group)

Figure 1: The Moss Landing fire the evening after it began.

7. The known height of the chimney stacks (150m) allows approximate scaling of the picture. The picture is taken from the South looking slightly west of North. The Pacific Ocean is just behind the chimney stacks. What appears to be a steady breeze is blowing onshore, and

carrying the polluted air inland. Heat from the fire is elevating the plume to about 500m, after which it starts to descend slowly. If the rate of descent was continued, it would hit the ground at 4 – 5 km from the fire. Although most of the plume is elevated, the photographs show some smoke and thus particulates closer to ground level, falling out of the plume due to gravity or turbulence. When the main plume reaches the ground, concentrations will be elevated, but there will also have been some dispersion which will tend to reduce concentrations, and very reactive species like HF may have been completely absorbed. What happens in a fire is very dependent on atmospheric conditions at the time. Wind direction is obviously very important, but also the stability of the atmosphere, and whether it is raining. More turbulent conditions can bring the plume to ground more quickly, and a temperature inversion can increase concentrations. To properly assess effects of a fire, this range of behaviour should be taken into account in modelling outcomes.

8. One important difference from the Lime Down proposal is that the Moss Landing BESS was in a building, which would possibly have aided spread but also hindered firefighting.

3. Methodology

9. The model used was ADMS-6 from Cambridge Environmental Research Consultants Ltd., a standard model used in the UK for these applications. Pollutants modelled were carbon monoxide (CO), formaldehyde, hydrogen chloride (HCl), hydrogen cyanide (HCN), hydrogen fluoride (HF), ammonia (NH₃), nitrogen dioxide (NO₂) and particulates (PM₁₀). Five years of meteorological data from RAF Fairford, 28 km distant, were used to calculate pollutant dispersion every hour. Fairford seems a reasonable choice with similar meteorological characteristics and environment to Hullavington. A fire in only a single BESS unit was simulated. Emissions data were derived from a test fire run by the Wärtsilä Corporation in the USA on one of its BESS modules. This produced concentration rather than emission data, and the emission data needed to run ADMS were derived by calibration. The batteries were lithium iron phosphate chemistry.
10. The concentrations were assessed at a range of receptors within 1 km of the BESS area at Hullavington. The maximum one hour mean pollutant concentration that would occur at each

receptor was calculated, based on the five years of meteorological data. The concentrations were compared to Acute Exposure Guidance Levels (AEGs) from the US Environmental Protection Agency, again standard practice in this type of study. AEGs are expressed as concentrations of a substance above which it is predicted that the general population, including susceptible individuals, could experience:

- a. Level 1 - Notable discomfort, irritation, or certain asymptomatic non-sensory effects. However, the effects are not disabling and are transient and reversible upon cessation of exposure
- b. Level 2 - Irreversible or other serious, long-lasting adverse health effects or an impaired ability to escape; and
- c. Level 3 - Life-threatening health effects or death.

11. Particulate matter is not a chemical and so does not have an AEG. Here the workplace exposure level standard from the Health and Safety Executive was used as the criterion.

12. Visibility reduction due to smoke was also assessed for local roads and the railway.

4. Results

13. All the modelled concentrations were well below Level 1. Visibility reduction at all sites was also assessed as insignificant.

5. Assessment of Model Validity

14. The Applicant's results are as a consequence of doubtful model assumptions, incorrect use of pollutant standards and flawed model implementation. A detailed critique is given below.

- a. The fire has been modelled assuming it can be confined to just one container. The reasonable worst case is however that the fire spreads to all containers. Though there may be safeguards to prevent this incorporated in the design, evidence supports that there is a realistic risk of their failure. They clearly failed to work at Moss Landing, and in a BESS fire at the Cirencester Solar Farm on 28 March 2025, a fire in a BESS container spread to an adjacent one, the damage in this case being limited by the fact that the BESS containers were scattered round the site in groups of three, unlike the situation

at Lime Down. As a matter of principle, risk assessments need to be based on a reasonable worst-case scenario. The history of BESS fires, which provides an evidential basis upon which to assess safeguards, gives insufficient reassurance that a fire would be confined to one container, and therefore a fire in *all* containers needs to be assessed. Given that there will be up to 270 containers, emissions have therefore been underestimated by up to a factor of approximately 270.

- b. Emissions have been modelled as though coming from an area source, and expressed in g/m^2 . A single container should be modelled as a point source. Modelling as an area source may give more pollutant dispersion than would really happen in practice, and thus lower pollutant concentrations. It would however be acceptable to model a fire in all 270 containers as an area source.
- c. The maximum pollutant travel distance modelled was 1 km. But given that thermal runaway fires are very hot, thermal uplift is likely to mean that under some circumstances the maximum concentration will occur further away than 1 km. The photograph of the Moss Landing fire (Fig. 1) shows this thermal uplift, with the pollutant plume first grounding at around 3-4 km from the fire. This is likely to be the point of maximum concentration, although there will also have been some dilution at this distance. Given that the Lime Down BESS will be of a comparable size to the Moss Landing facility that caught fire, Fig. 1 shows that artificially curtailing the analysis to 1 km is completely inadequate. ADMS will produce contour plots of pollutant concentrations round a source – this should be done with a realistic worst-case scenario as discussed in (1) above. Furthermore the Moss Landing experience clearly shows that particulate contamination occurred in the air and on the ground at several kilometres distance from the source. Such distances are also reflected in the scientific literature (2).
- d. Expanding the model envelope to a realistic distance brings many more receptors into the area at risk. Of particular note is Hullavington Primary School at 1.15 km and the whole of the large village of Hullavington at 1.0 to 1.5 km. The hamlet of Norton is at 1 to 1.2 km; the former Hullavington Airfield, now a Dyson Technology Campus, 2.5 to 3.6 km; the village of Corston, 3.7 km; Stanton St Quintin village and primary school is 3.8 km; Lower Stanton St Quintin, 3.8 km; Grittleton, 3.9 km; Leigh Delamere Village, 4.0 km; Leigh Delamere Motorway Services, 4.1 km; several km of the M4, 4.2-5.0 km; Pinkney, 4.2 km; Easton Grey, 4.3 km; the large village of Sherston, 4.4 km; Sherston

Primary School, 4.7 km; Alderton, 4.5 km; Rodbourne, 4.6 km. The Moss Landing Evacuation Order was for areas within 6 km of the fire, so using the same criteria, all these areas would have to be evacuated in the event of a BESS fire at Lime Down. The evidence from a real fire, as it is observed direct evidence of these matters, is much more compelling than modelling.

- e. The use of workplace exposure values to assess PM₁₀ concentrations is problematic. Workplace exposure limits are set by the Health and Safety Executive in the UK and are generally much higher (less stringent) than ambient concentration limits. The reasons are that workers are assumed to be healthy adults; exposure is limited to the working day; and workers are assumed to have consented to the conditions as part of their conditions of employment. WELs are also maximum values – exceedance can lead to prosecution and the normal expectation is that actual levels will be lower. They are not appropriate for assessing long-term exposure of populations which may contain vulnerable individuals, including old people and children, who are having the pollutants imposed on them without consent. Ambient environmental standards should be used in this case. While it is true that there are no short-term standards for exposure to particulate matter analogous to the AEGLs, there are short-term (24 hour) standards for particulate matter [[NAQ Objectives](#)]. For the UK, the 24-hour standard for PM₁₀ is 50 µg m⁻³ (compare the WEL of 4000 µg m⁻³ for 8h). However, it is a probabilistic standard, and exceedance is allowed up to 35 times a year in England and Wales (7 times a year in Scotland). The World Health Organisation [3] recommends a 99th percentile concentration (c.3 exceedances a year acceptable) of 45 µg m⁻³ for PM₁₀. The ADMS model results could be compared against these standards as it would be possible to generate a 24-hour mean with the fire continuing to burn. With the limited information available in the documentation, the modelled fire generates 1h exceedances of the UK standards for PM₁₀ at *every* receptor site if background is included (Appendix 15-2 Table 9).
- f. The closest receptors to the BESS are the railway and Hill Hayes Lane. But pollutant concentrations (except PM₁₀) are not modelled at these sites – only visibility. The modelled PM₁₀ maximum concentrations are much higher at these sites – over 1000 µg m⁻³ at two of the railway sites – implying that the concentrations of other pollutants would be high also. It is hard to see any justification for not assessing pollutant concentrations on the railway or at Hill Hayes Lane. Exposure for rail passengers would

be transitory (unless the train stopped for whatever reason) but Hill Hayes Lane has foot traffic and is no different from the PROWs in that respect.

- g. Particle concentrations have been modelled using a diesel fire as a model. This is unnecessary and unhelpful because there are enough data in the literature on particle emission rates from lithium-ion battery fires to estimate emissions directly. For instance, estimating particulate emissions using emission factor data from Claassen et al. (2024) [4] suggests PM₁₀ emissions from a single battery container with lithium iron phosphate chemistry would be between 17.5 and 32.9 g s⁻¹ depending on the state of the battery charge. The value used in this study was 7.06 g s⁻¹ (from Appendix 15-2, Tables 3 and 4) showing that emissions of PM₁₀ have been underestimated by a factor of between 2.50 and 4.66 for a 6-hour fire.
- h. ADMS is unlikely to be able to cope with situations such as a cloud of heavy vapour being produced in humid conditions and sinking into the adjacent railway cutting. This can affect HF particularly, but also other gases and fine particulates. This is likely to represent a real risk to the safety of workers and passengers on the railway if such a cloud eventuated.
- i. Emission rates for the study have been estimated using a convoluted calibration procedure. There are better established methods which would give more accurate results. For instance, using emission factors from Claassen et al. (2024) [4] shows that for a 6-hour fire affecting one container, the HF emission from lithium iron phosphate chemistry batteries would be between 9.1 g s⁻¹ and 18.9 g s⁻¹ depending on battery charge, as opposed to the 13.5 g s⁻¹ used in the study (from Appendix 15-2, Tables 3 and 4). The value used in the study comes within this range, but this does not represent a reasonable worst case. For instance, if the battery burnt out in 3h, the emission rate would be 37.8 g s⁻¹.
- j. Lithium-ion battery fires produce high concentrations of ultrafine particles (PM_{2.5}, PM_{1.0} and below) [4] [5], which can penetrate deep into the lungs and cross the blood-brain barrier. These have not been modelled in the study. ADMS has the capacity to model PM_{2.5}. There have also been reports [6] of organophosphorus compounds. Three of the chemicals are phosphonofluoridates or phosphorofluoridates. Their structural similarity to some organophosphorus compounds (which are highly toxic) The estimation of toxic emissions from Li-ion battery fires is a rapidly evolving field. A recent review by Bugryniec and colleagues [7] notes the importance of battery

chemistry; configuration; size; and state of charge on the results of experimental investigations. Nilsson and Tidblad [8] conclude that general conclusions cannot be drawn from the bulk of experimental studies of Li-ion battery fires, and call for a more systematic approach to the design of experiments. In these conditions of uncertainty, where the health and lives of people are being put in jeopardy, it is appropriate to use the Precautionary Principle when modelling possible outcomes, to ensure that risks are minimised and to enable an appropriate amount of emergency planning.

6. Conclusions

15. The modelling in the Environmental Statement of the air pollution effects of a fire in the Lime Down BESS is seriously flawed and leads to an impression of safety which is not properly made out. The reasons are discussed above. A reasonable worst-case scenario has not been modelled; the geographical limits of the model are too narrow; inappropriate standards are used for particulates; pollutant concentrations at the closest receptors have not been modelled; model parameterization is inaccurate; and not all potentially toxic species have been modelled. By analogy from past BESS fires, nine villages, including three village primary schools, and several smaller settlements are within reach of emissions from a serious fire, and something approaching 10,000 people may need evacuation. An objective assessment of risk, taking into account the most up to date and appropriately identified parameters is required. The evaluation is needed irrespective of the agreed fact that utility-scale BESS fires are fairly uncommon. The nuclear industry provides an analogy; radioactive releases from nuclear power stations are much rarer than BESS fires, yet there are comprehensive risk assessments around nuclear power stations, emergency plans are formulated and kept up to date, and there is a safety culture and organisational structure dedicated to ensuring that radioactive releases do not occur at all. Based on the current assessment provided by the Applicant, it fails to consider the reasonable worst-case and as a result fails to assess the likely significant effects of BESS fire.

7. References

1. Hutton, G. and I. Stewart, *Battery Energy Storage Systems*. 2025, House of Commons Library: London.
2. Aiello, I.W., et al., *Coastal wetland deposition of cathode metals from the world's largest lithium-ion battery fire*. Nature Scientific Reports, 2025. **15**(1): p. 42113.

3. WHO, *WHO global air quality guidelines: particulate matter (PM_{2.5} and PM₁₀), ozone, nitrogen dioxide, sulfur dioxide and carbon monoxide*. . 2021, Geneva: World Health Organisation.
4. Claassen, M., et al., *Characterization of Lithium-Ion Battery Fire Emissions—Part 2: Particle Size Distributions and Emission Factors*. Batteries, 2024. **10**(10): p. 366.
5. Pemnarth V., et al., *Characerization of particulate emissions from thermal runaway of lithium-ion cells*. ASME Journal of Electrochemical Energy Conservation and Storage. (2025) **22**, 031002-1 DOI: 10.1115/1.4065938.
6. Bergström, U., et al., *Vented gases and aerosol of automotive Li-ion LFP and NMC batteries in humidified nitrogen under thermal load*, in *Försvarets Forskningsinstitut FOI*. 2015.
7. Bugryniec, P.J., et al., *Review of gas emissions from lithium-ion battery thermal runaway failure—Considering toxic and flammable compounds*. Journal of Energy Storage, 2024. **87**: p. 111288.
8. Nilsson, E.J. and A. Ahlberg Tidblad, *Gas emissions from lithium-ion batteries: a review of experimental results and methodologies*. Batteries, 2024. **10**(12): p. 443.

The evidence that we have prepared and provided is true, and represent our true and professional opinions irrespective of by whom we are engaged.

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Research Briefing

23 June 2025

By 

Battery energy storage systems



Summary

- 1 Background: Targets and statistics
- 2 What permissions do BESSs need?
- 3 Concerns about the safety of BESSs
- 4 How is the safety of BESSs regulated?
- 5 Barriers to the development of BESSs
- 6 Further reading

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Summary

Battery energy storage systems (BESSs) use batteries, for example lithium-ion batteries, to store electricity at times when supply is higher than demand. They can then later release electricity when it is needed. BESSs are therefore [important for “the replacement of fossil fuels with renewable energy”](#).

The government has a legally binding [target to reduce the UK’s greenhouse gas emissions by 100% by 2050](#), compared with 1990 levels. This is known as the ‘net zero target’. To meet this target, the government has set the aim of achieving “[clean power by 2030](#)”. This means “being on track to achieving at least 95% of low carbon generation by 2030”.

Renewables, such as wind and solar power, rely on the weather to generate electricity. This means that they cannot adjust to demand from consumers as easily as fossil fuels and nuclear power can. Therefore a decarbonised power system will need to be [supported by technologies that can respond to fluctuations in supply and demand](#), including energy storage.

The government’s [Clean Power Action Plan](#) stated that it expected 23–27 GW of battery storage to be needed by 2030 to support clean power, up from 4.5 GW in 2024.

What permissions do BESSs need?

Installing a grid-scale BESS requires planning consent. Planning is a devolved matter, and decision-making rules differ across the UK.

In England and Wales, decisions on BESSs (regardless of their capacity) are made by local planning authorities. In Scotland and Northern Ireland, BESSs require consent from either ministers or the planning authority depending on their storage capacity.

Depending on its capacity, a BESS may also require a generating licence to operate. Generating [licences are issued by Ofgem in Great Britain](#) or by the Utility Regulator in Northern Ireland.

Concerns about the safety of BESSs

Although safety incidents for BESSs are rare, a common concern about BESSs is the [potential fire risk of lithium-ion batteries](#) (PDF). Lithium-ion batteries

can catch fire because of a process called “thermal runaway”. It can occur, for example, if part of a battery is damaged.

Understanding of thermal runaway has improved in recent years, leading to more flame-resistant batteries. BESS sites can be also [designed with safety features, such as fire suppression systems](#).

There is no reliable, publicly accessible record of the number of BESS fires that have occurred in the UK or elsewhere. Two documented incidents of a BESS fire in the UK include: [a fire at a BESS site in Liverpool in September 2020](#) (PDF) and a [fire at a BESS project under construction in Essex in February 2025](#).

How is the safety of BESSs regulated?

While there is no single piece of legislation that sets specific requirements for BESS sites, there is a framework of relevant legislation that applies. Individual batteries may have to adhere to [product safety regulations](#) (for example if second-life batteries are used) and grid-scale facilities will also have to comply with relevant [fire safety requirements](#) and [health and safety laws](#). The Health and Safety Executive has a [grid-scale battery energy storage systems](#) webpage that collates relevant existing legislation. The government also published new guidance on the [health and safety of grid scale electrical energy storage](#) in April 2024.

There are [concerns that existing regulations and guidance are not sufficient](#) to address the safety risks posed by BESSs. For example, there are calls for fire and rescue services to be made statutory consultees in planning applications for BESSs.

In response to concerns about the safety of BESSs, the government said [BESSs were covered by “a robust regulatory framework”](#). It published guidance in August 2023 which encourages developers and [local planning authorities in England to consult their local fire and rescue service](#) in preparing and deciding on planning applications for BESSs.

Barriers to the development of BESSs

The Commons Business and Trade Select Committee has raised concerns that [the UK has “insufficient domestic manufacturing capacity”](#) for batteries, and the Commons Foreign Affairs Select Committee has raised concerns that [“the UK is almost completely dependent on imports for critical minerals”](#), such as lithium, that are used in batteries.

The government set out how it intends to secure the UK’s battery supply chain and improve the resilience of the UK’s critical minerals supply in its [UK battery](#)

[strategy](#) (November 2023) and [critical minerals strategy](#) (July 2022). The Labour Government has said it plans to produce a new critical minerals strategy in 2025.

Barriers to the development of BESSs and other energy storage systems also include [high upfront capital costs, uncertain revenue streams](#) and delays to grid connections.

The government and Ofgem are implementing a [new 'cap and floor' revenue support scheme](#) for long duration electricity storage (LDES), which will include BESSs as well as other technologies. This will provide a guaranteed minimum revenue stream (the floor) and a limit on revenues (the cap) for long duration energy storage. Ofgem [aims to approve the first round of projects](#) by Q2 2026.

1

Background: Targets and statistics

Battery energy storage systems (BESSs) can store electricity during times when supply is high and demand is low, then release electricity when it is needed.

BESSs can support the use of renewable energy technologies, such as wind and solar power, because they reduce the supply issues that are associated with their intermittency. They can store the electricity that is generated when, for example, the sun is shining and the wind is blowing, and release it later when consumers need it.¹ This process of storing energy is also called “grid balancing”.² National Grid states that BESSs are “essential to speeding up the replacement of fossil fuels with renewable energy”.³

This briefing focuses on grid-scale battery energy storage solutions. For further information about domestic batteries, see [research commissioned by the government on domestic battery energy storage systems](#).

1.1

What are the governments targets on clean energy?

The government has a legally binding [target to reduce the UK’s greenhouse gas emissions by 100% by 2050](#), compared with 1990 levels.⁴ This is known as the ‘net zero target’. This will require reducing the UK’s reliance on fossil fuels and increasing its use of renewable energy sources, such as wind and solar power.

One of the government’s [five missions](#) in its Plan for Change for this Parliament is to “[make Britain a clean energy superpower](#)”.⁵ This includes an ambition to deliver clean power by 2030.⁶ This means “being on track to

¹ National Grid, [Energy explained: What is battery storage?](#), undated [accessed 14 March 2024]

² The Royal Society, [Large-scale electricity storage](#), September 2023; Department for Business and Trade (DBT), [UK battery strategy](#), last updated December 2023, p25

³ National Grid, [What is battery storage?](#), undated [accessed 14 March 2024]

⁴ More information on net zero, including the underpinning legislation, policies and progress, is set out in the Library research briefing [The UK’s plans and progress to reach net zero by 2050](#).

⁵ HM Government, [Plan for Change](#) (HTML) and [Plan for Change: Milestones for mission-led government](#) (PDF), 5 December 2024.

⁶ The government defines clean power to mean means power delivered by renewable energy and low-carbon sources (such as nuclear and future low carbon technologies such as gas with carbon capture and storage). DESNZ, [Clean Power 2030 Action Plan](#), 13 December 2024.

achieving at least 95% of low carbon generation by 2030”.⁷ These commitments were a key part of the Labour Party’s [2024 general election manifesto](#).⁸

In December 2024, the Department for Energy Security and Net Zero (DESNZ) published its [Clean Power 2030 Action Plan](#). This set out the government’s pathway to a clean power system and what it plans to do to achieve this. Further information is provided in the Library briefing paper, [Clean power targets](#) (28 March 2025).

1.2 What are the government’s targets for BESS?

The government’s [Clean Power Action Plan](#) stated that it expected 23-27 GW of battery storage to be needed by 2030 to support its clean power ambitions, up from 4.5 GW it said was installed in December 2024 – a “very significant level of increase”. The government said it expected most this storage to come from grid-scale batteries, with small-scale batteries also contributing.

1.3 What types of batteries are used in BESSs?

Energy storage systems for the grid must be able to accommodate daily and seasonal fluctuations in electricity demand.

Batteries are best suited for short-duration energy supply. This means that they can supply energy boosts to the grid for several hours. They are usually only used for short durations of energy supply because they are relatively expensive to install but are energy efficient (meaning that little charge is lost in the process of charging the battery).⁹

Pumped hydroelectric storage is a long-duration energy storage technology currently deployed on a large scale. It can store large amounts of energy that can be discharged to the grid over 1-2 days. There are other emerging long-duration energy storage technologies, such as compressed air, liquid air, gravitational storage, hydrogen and flow batteries, that are being developed, discussed below.¹⁰

⁷ HM Government, [Plan for Change: Make Britain clean energy superpower](#), accessed 17 April 2025.

⁸ The Labour Party, [Change: Labour Party Manifesto 2024](#), Make Britain clean energy superpower, June 2024

⁹ Bruno Cardenas and others, [Short-, Medium-, and Long-Duration Energy Storage in a 100% Renewable Electricity Grid](#), *Energies*, Vol 14, 2021, 8524

¹⁰ DESNZ, [Long duration electricity storage: scenario deployment analysis](#), January 2024.

Types of batteries used in BESSs

Lithium-ion batteries

Currently, lithium-ion batteries are the most common BESS being installed in the UK. According to a [2023 report by the research organisation, The Faraday Institution](#) (PDF), lithium-ion batteries accounted for over 90% of grid-scale installations in the UK from 2020 to 2024.¹¹

There are different types of lithium-ion batteries. They all follow the same principles in how they store energy (see box 1) but are made of different materials, have different costs, and can store different amounts of energy.

The two main types of lithium-ion batteries used for BESSs are lithium-iron-phosphate (LFP) and lithium-nickel-manganese-cobalt-oxide (NMC). The Faraday Institution estimates that lithium-iron-phosphate batteries made around 60% of lithium-ion BESSs in the UK market in 2022 because of their lower cost and higher efficiency.¹²

1 How do lithium-ion batteries work?

A battery is a device that stores electrical energy as chemical energy that can be released on demand. Batteries can be rechargeable or non-rechargeable:

- Non-rechargeable batteries are typically used in household devices such as torches.
- Rechargeable batteries can store and release energy many times. Rechargeable batteries, such as lithium-ion batteries, are used for applications that have greater energy demand such as mobile phones, electric vehicles and grid storage.

A lithium-ion battery stores energy by using electricity to move positively charged particles of lithium (lithium ions) from one side of the battery (the cathode) to the other (the anode). The lithium ions are stored in the anode when the battery is charged up. When electricity is needed, lithium ions move back to the cathode which discharges electricity to the device or grid.¹³

This process is repeated in a cycle of charging and discharging the battery.

¹¹ The Faraday Institution and Rho Motion, [Market and Technology Assessment of Grid-Scale Energy Storage required to Deliver Net Zero](#) (PDF), September 2023, p4

¹² The Faraday Institution and Rho Motion, [Market and Technology Assessment of Grid-Scale Energy Storage required to Deliver Net Zero](#) (PDF), September 2023

¹³ United States Department of Energy, [How Lithium-ion Batteries Work](#), February 2023; Australian Academy of Science, [Lithium-ion batteries](#), undated [accessed 15 April 2024]

Alternatives to lithium-ion batteries

Alternative batteries that can be used in BESS facilities include:

- Lead-acid batteries, a predecessor of lithium-ion, are relatively cheap and easy to recycle. However, they have a limited lifespan and store a low amount of energy-per-weight due to the high density of lead. They are therefore not commonly used anymore.¹⁴
- Flow batteries, also known as redox-flow batteries, are an emerging technology. They store energy in large tanks of liquid. The liquid is pumped into a battery to release the energy when required. Flow batteries can store energy for years and discharge energy over longer durations than lithium-ion batteries. They are also easy to maintain but, unlike lithium-ion batteries, are limited to grid-scale storage because of their size and weight.¹⁵
- Sodium-ion batteries work in a similar way to lithium-ion but use sodium ions instead of lithium ions. Sodium can be sourced from seawater, making it a cheaper raw material than lithium.¹⁶ However, because sodium is heavier than lithium, sodium-ion batteries are expected to be more appropriate for grid storage than mobile storage. Although sodium-ion batteries are not used on a large scale yet, in its [2023 UK battery strategy](#), the government stated that it expects to deploy this technology in the next ten years.¹⁷

Other types of energy storage systems

Some examples of long-duration energy storage include:

- Pumped hydroelectric storage: Electricity is generated when water is released from a reservoir and travels downhill through a turbine. The water is pumped uphill during lower energy demand. According to the [British Hydropower Association](#), the UK's pumped hydropower capacity was 2.8 GW or 32 GWh (see box 2 for an explanation of the difference).¹⁸
- Hydrogen storage: Excess electricity can be used to produce hydrogen by electrolysis. Hydrogen gas must be stored under very low temperatures and high pressures. Currently, there is one large-scale hydrogen storage site in the UK which has a capacity of 25 GWh. There are plans for three new sites which the government said could be operational by the late

¹⁴ Xiayue Fan and others, [Battery Technologies for Grid-Level Large-Scale Electrical Energy Storage](#), Transactions of Tianjin University, Vol 26, 2020, p92-103

¹⁵ ABC Science, [Vanadium redox flow batteries](#), February 2023

¹⁶ The Faraday Institution, [Sodium-ion Batteries: Inexpensive and Sustainable Energy Storage](#) (PDF), May 2021

¹⁷ DBT, [UK battery strategy](#), last updated December 2023

¹⁸ British Hydropower Association, [Pumped Storage Hydro](#), undated [accessed 22 April 2024]

2020s.¹⁹ Hydrogen can be converted back into electricity (usually using a fuel cell) or used directly in heating, transport or industry applications.²⁰

- Compressed air storage: Electricity can be used to compress air which is then stored at a high pressure. When the air expands to atmospheric pressure, it can be used to drive turbines to generate electricity. There are currently no compressed air energy storage systems in the UK.²¹

For further information about these energy storage systems, see the Royal Society's [briefing on large-scale electricity storage](#) and the Parliamentary Office of Science and Technology's (POST) note on [longer duration energy storage](#).

2 Watts and watt-hours

A watt (W) and its multiples (such as megawatt, MW, and gigawatt, GW) are units of power. They measure the amount of energy generated or consumed per unit of time. Watts can be used to measure the installed capacity, that is, the maximum amount of power that a battery can generate at any particular time.

A watt-hour (Wh) is a unit of energy. It measures the total amount of energy that a battery or other energy storage systems can store and provide.

To illustrate, a 100-watt battery provides 100W of power at any given moment. If that battery runs for 2 hours, it delivers 200 watt-hours of energy.

For an energy storage system, the amount of energy stored (watt-hour), and the power (watt, an energy per unit of time) that the system can deliver at any given time, are both important when comparing different technologies and applications.

Battery storage capacity in the UK was 6.8 GW and electricity output from battery storage was 1,811 GWh (or 1.8 TWh) at the end of 2024.²²

¹⁹ BEIS, [Proposals for hydrogen transport and storage business models](#): Analytical annex, September 2022

²⁰ The Royal Society, [Large-scale electricity storage](#), September 2023; POST, [longer duration energy storage](#), 20 December 2022.

²¹ The Royal Society, [Large-scale electricity storage](#), September 2023

²² DESNZ, [Digest of UK Energy Statistics \(DUKES\): electricity, Energy storage overview \(DUKES 5.16\)](#)

1.4

How many BESSs are there in the UK?

There is no definitive estimate of the number of operational BESS and their total capacity in the UK.

According to the government’s [renewable energy planning database](#), there were 124 operational BESS sites with a total capacity of 2,630 megawatts (MW) of electricity (2.6 gigawatts, GW) as of April 2025. However, this is likely to be an underestimate as facilities with a capacity below 1 MW that went through the planning system prior to 2021 were not recorded.

The government’s Clean Power Action Plan said that there was 4.5 GW of battery storage capacity in Great Britain in December 2024.²³ This is power capacity, which is the rate at which they can supply their stored energy; see [Box 2 in section 1.3](#) for definitions.

Renewable energy planning database

Data on the number of BESSs is also available in the government’s [renewable energy planning database](#) (REPD). The REPD tracks the progress of energy projects, including BESSs, through the planning system. Until 2021, the REPD only recorded projects with a capacity over 1 MW. Since 2021, it also includes projects with a capacity over 150 kilowatts (kW).²⁴ Therefore, BESSs that were going through the planning system before 2021 may not have been captured in the REPD.

Number and capacity of operational BESSs and BESSs under/awaiting construction, as of April 2025								
	Co-located with renewable energy		Stand-alone storage		Co-located with fossil fuel plant		Total	
	Capacity		Capacity		Capacity		Capacity	
	Number	(MW elec)	Number	(MW elec)	Number	(MW elec)	Number	(MW elec)
Operational	51	388	65	2,001	6	142	124	2,630
Under/awaiting construction	282	4,333	583	38,567	9	499	930	45,615

Note: The totals do not sum due to 58 BESSs not recording their storage type (stand alone or co-located).

Source: DESNZ, [Renewable Energy Planning Database: Quarterly extract: April 2025](#) [accessed 18 June 2025]

According to the REPD, as of April 2025, there were 124 operational BESS in the UK with a total capacity of 2.6 GW. Of the operational BESSs recorded in

²³ DESNZ, [Clean Power 2030 Action Plan](#), 13 December 2024.

²⁴ DESNZ, [Renewable Energy Planning Database: Quarterly extract](#), last updated 12 February [accessed 22 March 2024]

the REPD, 65 were stand-alone storage facilities and 51 were co-located with renewable energy projects (such as solar and wind farms).

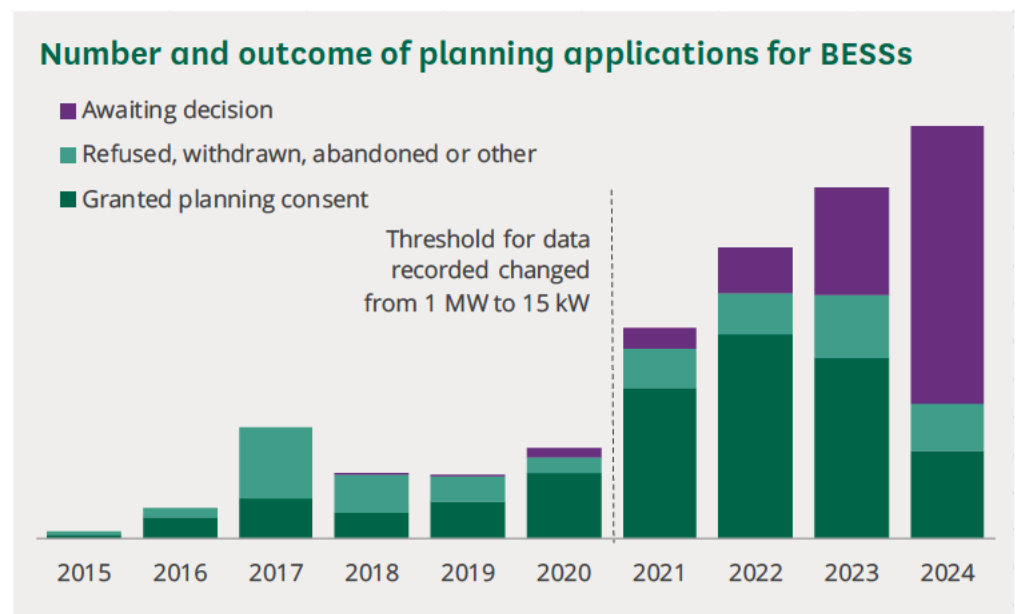
A further 930 BESSs recorded in the REPD have been given planning consent and are awaiting or under construction as of April 2025. Of these, 583 were stand-alone storage facilities and 282 were co-located with renewable energy projects.²⁵

A small number of BESSs recorded in the REPD are also co-located with fossil fuel plants (15 in total in the UK, of which six are operational and nine are under or awaiting construction).

Most BESSs recorded in the REPD (that are operational or awaiting or under construction) are located in England: 99 operational BESSs and 678 BESSs that are awaiting or under construction. This is followed by Scotland where 19 operational BESS are located and 190 BESSs that are awaiting or under construction.

The chart below shows the number of applications for BESSs that have been submitted for planning permission since 2015. The figures before and after 2021 cannot necessarily be compared, as the capacity threshold for recording projects in the REPD changed in 2021.

Of the applications submitted between 2015 and 2024 on which a decision has already been made, 70% were granted planning consent and 20% were refused consent, withdrawn or abandoned by developers. 517 applications for BESS that have been submitted are still awaiting a decision.



Source: DESNZ, [Renewable Energy Planning Database: Quarterly extract](#): January 2025 [accessed 30 May 2025]

²⁵ DESNZ, [Renewable Energy Planning Database: Quarterly extract](#), [Renewable Energy Planning Database \(REPD\): January 2024](#) (CSV), last updated 12 February 2024 [accessed 22 March 2024]

2

What permissions do BESSs need?

The installation of a grid-scale battery energy storage system (BESS) requires planning consent. Planning is a devolved matter, and decision-making rules for planning consent for BESSs differ across the UK.

In England and Wales, decisions on BESSs (regardless of their capacity and size) are made local planning authorities. In Scotland and Northern Ireland, the size of a project will determine whether a BESS requires consent from ministers or the local planning authority.

Depending on its capacity, a BESS may also require a generating licence to operate. Generating licences are issued by the Office for Gas and Electricity Markets (Ofgem), the independent energy regulator for Great Britain, or the Utility Regulator in Northern Ireland.

2.1

Planning permission

England

Until 2020, energy storage systems with a generating capacity over 50 MW were classed as ‘nationally significant infrastructure projects’ (NSIP) under the [Planning Act 2008](#). They required development consent from the Secretary of State for Business, Energy and Industrial Strategy (BEIS).²⁶ Energy storage systems with a generating capacity up to 50 MW required planning consent from the local planning authority.

Following two consultations,²⁷ the government decided to remove BESSs from the NSIP regime in 2020.²⁸ The government said the “time and cost of the NSIP regime is not proportionate to the planning impacts of these technologies”. It also said the 50 MW threshold had distorted decisions about sizing, leading to a large number of facilities with a capacity of around 49 MW.²⁹

The government explained that the role of energy storage in reaching its net zero target by 2050 had motivated it to remove BESSs from the NSIP regime:

Electricity storage is a key technology in the transition to a smarter and more flexible energy system and will play an important role in helping to reduce

²⁶ [Part 3 of the Planning Act 2008](#)

²⁷ BEIS, [Error! Hyperlink reference not valid.](#), October 2019; BEIS [Error! Hyperlink reference not valid.](#), last updated July 2020

²⁸ [Infrastructure Planning \(Electricity Storage Facilities\) Order 2020](#)

²⁹ [Explanatory Memorandum to the Infrastructure Planning \(Electricity Storage Facilities\) Order 2020](#)

emissions to net-zero by 2050. These changes will make it simpler for large scale storage facilities to seek planning permission, helping to bring forward larger projects supporting more efficient grid balancing and management of intermittent renewable generation.³⁰

All energy storage systems (except pumped hydro), regardless of their size and capacity, are now determined by local planning authorities in the first instance.³¹

How are planning applications decided?

A local planning authority will generally decide planning applications in line with its local plan which sets out its policies for the development and land use in its area. The local plan forms the basis for decision-making on planning applications unless ‘material considerations’ indicate otherwise.³²

There is no set list of material considerations. One material consideration is the [National Planning Policy Framework](#) (last updated December 2024) which sets out the government’s planning policies for England. It is supported by [planning practice guidance on renewable and low-carbon energy](#).

The National Planning Policy Framework states that the planning system should “support renewable and low carbon energy and associated infrastructure”.³³ It states that when determining planning applications, local planning authorities should “give significant weight to the benefits associated with renewable and low carbon energy generation and the proposal’s contribution to a net zero future”.³⁴

The government’s planning practice guidance notes the potential benefits of BESSs “to use energy more flexibly and de-carbonise our energy system cost-effectively”.³⁵

Information about the Secretary of State’s power to call-in planning decisions and to ‘recover’ planning appeals for their own determination, is provided in the Library briefing: [Call-in of planning applications \(England\)](#), 4 April 2024.

Where should BESS be sited?

There are no specific policies for the siting of BESS. The National Planning Policy Framework (last updated in December 2024) advises local planning

³⁰ BEIS, [The planning system for electricity storage: Follow up consultation](#), last updated July 2020

³¹ An applicant can appeal the refusal of permission or the non-determination of an application with the Planning Inspectorate, an executive agency of the Department for Levelling Up, Housing and Communities. The decisions of LPAs and the Planning Inspectorate can also be challenged in court.

³² [Section 70\(2\) of the Town and Country Planning Act 1990](#); [Section 38\(6\) of the Planning and Compulsory Purchase Act 2004](#)

³³ Ministry of Housing, Communities and Local Government (MHCLG), [National Planning Policy Framework](#), December 2024, para 161.

³⁴ MHCLG, [National Planning Policy Framework](#), December 2024, para 168.

³⁵ MHCLG, [Renewable and low carbon energy](#), last updated August 2023, para 32.

authorities to identify “suitable areas” for renewable energy developments and supporting infrastructure.

The planning practice guidance states that “there are no hard and fast rules” on how authorities should identify suitable areas. It advises local authorities to consider potential impacts of developments on the local environment and the views of local communities.³⁶

The National Planning Policy Framework (NPPF) guides all development away from the ‘best and most versatile’ agricultural land, that is, [good to excellent-quality agricultural land](#).³⁷ It states that elements of many renewable energy projects are not “appropriate” development for green belt land.³⁸ Developers would need to demonstrate “very special circumstances” for a project to proceed. Very special circumstances may include the wider environmental benefits associated with increased production of energy from renewable sources.³⁹

There is no mandatory minimum distance that BESS sites should be from other buildings. [Guidance on grid-scale BESS by the National Fire Chiefs Council](#) (PDF, November 2022) recommended a minimum of 25 metres from occupied buildings and site boundaries.⁴⁰

Who is consulted on planning applications?

Before deciding a planning application, a local authority must publicise the application to allow local residents and other interested parties to express their views on the proposed development.⁴¹ A local authority must take these views into account when deciding a planning application.

In August 2023, the government updated its planning practice guidance to encourage developers and local planning authorities to consult their local fire and rescue service on planning applications for BESSs.⁴² However, fire and rescue services are not statutory consultees, so local planning authorities are not required to consult them.

The government also updated its planning practice guidance to refer local planning authorities and developers to [guidance on grid-scale BESS by the National Fire Chiefs Council](#) (PDF). For further information on the guidance, see section 4.2 of this briefing.

³⁶ MHCLG, [Renewable and low carbon energy](#), last updated August 2023, para 32

³⁷ MHCLG, [National Planning Policy Framework](#), December 2024.

³⁸ MHCLG, [National Planning Policy Framework](#), December 2024, para 160.

³⁹ MHCLG, [National Planning Policy Framework](#), December 2024, para 160.

⁴⁰ National Fire Chiefs Council (NFCC), [Grid Scale Battery Energy Storage System planning – Guidance for FRS](#) (PDF), last updated October 2023.

⁴¹ [Articles 15 and 33 of the Town and Country Planning \(Development Management Procedure\) \(England\) Order 2015](#)

⁴² MHCLG, [Renewable and low carbon energy](#), last updated August 2023, paras 34-35

Wales

Between 2015 and 2019, all energy storage systems with a capacity between 10 MW and 350 MW required consent from Welsh ministers under the Welsh Government's regime for 'developments of national significance'. BESSs in Wales with a capacity over 350 MW were governed by the UK Government's NSIP regime.⁴³

In 2019, the Welsh Government changed planning rules so planning decisions on energy storage systems (except pumped hydro) with a capacity up to 350 MW were made by local planning authorities. In 2020, the UK Government removed energy storage (except pumped hydro) from the NSIP regime.⁴⁴ Therefore, planning decisions on all BESSs, regardless of their capacity, are now made by local planning authorities in Wales (as is the case in England).

How are planning applications decided?

Local planning authorities in Wales will decide planning applications for BESS in line with [Future Wales](#) (the Welsh Government's spatial plan), [Planning Policy Wales](#) (which sets out national planning policies), and relevant policies set out in the local development plan for the area.

Planning Policy Wales notes the importance of energy storage to manage the transition to a low-carbon economy, including to manage the intermittency of supply from renewable energy sources and to meet increases in demand from the electrification of heat and transport. Future Wales states there is "a need to consider large-scale energy storage [...] to provide grid balancing".⁴⁵

Planning Policy Wales advises local planning authorities to "optimise energy storage" and support proposals for storage facilities "where possible".⁴⁶

Scotland

In a [letter to chief planning officers in 2020](#), the Scottish Government clarified that BESSs should be "treated as any other generating station" in Scotland.⁴⁷ This means that, depending on their capacity, BESS need either consent from planning authorities or from Scottish ministers:

- Planning authorities are responsible for determining applications for electricity generating stations with a capacity up to 50 MW.

⁴³ [Section 19 of the Planning \(Wales\) Act 2015](#), which added [section 62D to the Town and Country Planning Act 1990](#); [Section 15 of the Planning Act 2008](#)

⁴⁴ [Developments of National Significance \(Specified Criteria, Fees and Fees for Deemed Applications\) \(Wales\) \(Amendment\) Regulations 2019](#); [Infrastructure Planning \(Electricity Storage Facilities\) Order 2020](#)

⁴⁵ Welsh Government, [Planning Policy Wales](#), last updated February 2024, para 5.7.12; Welsh Government, [Future Wales: The National Plan 2040](#), February 2023, p99

⁴⁶ Welsh Government, [Planning Policy Wales](#), last updated February 2024, para 5.7.7; para 5.7.12

⁴⁷ Scottish Government, [Battery storage consents: Chief Planner letter August 2020](#), August 2020

- Scottish ministers are responsible for determining applications to construct and operate electricity generating stations with a capacity over 50 MW under [section 36 of the Electricity Act 1989](#).
 - If they grant consent under section 36, Scottish ministers can also grant planning permission at the same time (so a developer does not need to make a separate planning application).⁴⁸

How are planning applications decided?

Planning authorities and Scottish ministers will decide planning applications in line with the area's local development plan and the Scottish Government's [National Planning Framework 4](#) (NPF4).

The NPF4 provides that planning authorities should support proposals for “all forms” of renewable and low-carbon technologies, including BESSs. However, it advises that they should ensure that proposals maximise economic impact and minimise certain other impacts (including on the surrounding landscape, the environment, and nearby communities).⁴⁹

The NPF4 also states that planning authorities should place “significant weight” on the contribution of a development to renewable energy and greenhouse gas emissions targets.

Northern Ireland

In Northern Ireland, depending on their size, developments fall into one of three categories: ‘local’, ‘major’ or ‘regionally significant’.

- Local and major developments are decided by the local planning authority.
- For regionally significant developments, applicants are required to consult Northern Ireland Department for Infrastructure. If the department decides that a development is “significant to the whole or a substantial part of Northern Ireland”, it (rather than the local planning authority) will decide the planning application.
- For major and regionally significant developments, applicants must hold a pre-application consultation with the local community.⁵⁰

In a [letter to chief planning officers in 2020](#), the Department for Infrastructure said that it considered that “electricity storage development falls within the meaning of an ‘electricity generating station’ [...] for the purposes of planning in Northern Ireland”.⁵¹ Electricity generating stations with a capacity over

⁴⁸ [Section 36 of the Electricity Act 1989](#); Section 57 of the Town and Country Planning (Scotland) Act 1997

⁴⁹ Scottish Government, [National Planning Framework 4](#), February 2023, Policy 11

⁵⁰ [Schedule of the Planning \(Development Management\) Regulations \(Northern Ireland\) 2015](#); [Section 26 of the Planning Act \(Northern Ireland\) 2011](#)

⁵¹ Department for Infrastructure, [Seventh Chief Planner's Update issued to Councils](#), December 2020

5 MW are major developments, and electricity generating stations with a capacity over 30 MW are classed as potentially regionally significant.

The decision to treat BESSs as electricity generating stations to decide whether they are local or major developments was challenged under judicial review. The court found that BESSs could not be considered to fall “wholly within” the class of electricity generating stations. Whether BESS are local or major developments will therefore depend on their site area exceeds one hectare.⁵²

How are planning applications decided?

In determining planning applications, the Department for Infrastructure and local planning authorities in Northern Ireland are guided by the department’s [Strategic Planning Policy Statement](#) and the [Regional Development Strategy for Northern Ireland 2035](#). Neither contain specific policies for decision-making on BESS.

The Strategic Planning Policy Statement advises local planning authorities to support “development that generates energy from renewable resources” if the development will not result in “an unacceptable adverse impact” on other considerations (such as visual amenity, landscape character, public safety or residential amenity).⁵³ The Strategic Planning Policy Statement does not set out whether BESSs are considered a form of renewable energy development.

2.2

Generating licence from Ofgem

Under the [Electricity Act 1989](#), certain activities, including the generation of electricity in Great Britain, require a licence from Ofgem (the independent energy regulator). Licensees must comply with Ofgem’s [industry codes and standards](#) and [license conditions](#).

In general, operators of BESSs that have a net capacity of more than 100 MW must hold a generating licence. Operators of BESSs that have a net capacity of less than 100 MW and that do not exceed 50 MW of power at any time are likely exempt from the requirement to hold a generating license.⁵⁴

Until 2023, the Electricity Act 1989 included no specific definition of, or reference to, electricity storage. The government established a definition of electricity storage through the [Energy Act 2023](#) and clarified that electricity storage, such as BESSs, is a distinct subset of generation within the Electricity Act 1989.⁵⁵

⁵² [ABO Wind NI Limited and Energia Renewables Company 1 Limited](#) [2021] NIQB 96; Cleaver Fulton Rankin, [BESS and the impact on the NI energy sector](#), undated [accessed 18 March 2024]

⁵³ Department for Infrastructure, [Strategic Planning Policy Statement](#), September 2015, para 6.224

⁵⁴ [Electricity \(Class Exemptions from the Requirement for a Licence\) Order 2001](#)

⁵⁵ [Section 213 of the Energy Act 2023](#), which amended [Section 4 of the Electricity Act 1989](#)

Ofgem has treated electricity storage as a subset of electricity generation for regulation purposes since 2020.⁵⁶ The change made by the Energy Act 2023 formalises that approach. The government said the change removes “current ambiguities” and provides “clarity and certainty” on how electricity storage is treated within the existing and possible future frameworks.⁵⁷

For further information about the changes introduced by the Energy Bill (now Act), see section 3.4 of the Library briefing, [Energy Bill \[HL\] 2022-23, parts 4-6: Electricity and gas markets](#).

Northern Ireland

Energy policy is a devolved matter in Northern Ireland. The Utility Regulator is responsible for issuing and enforcing licences to those who want to generate or supply electricity in Northern Ireland.⁵⁸

Like Ofgem, the Utility Regulator treats BESSs as a type a generating station and expects them to obtain a generating licence.⁵⁹ In Northern Ireland, BESSs which do not exceed 10 MW of power at any point in time are exempt from the requirement to hold a generating license.⁶⁰

⁵⁶ Ofgem, [Decision on clarifying the regulatory framework for electricity storage: Changes to the electricity generation licence](#), October 2020

⁵⁷ [Explanatory Notes to the Energy Bill 2022-23](#), para 48

⁵⁸ Northern Ireland Authority for Utility Regulation, [Licence to supply electricity \(Northern Ireland\)](#), undated [accessed 24 April 2024]

⁵⁹ Utility Regulator, [Decision to grant an electricity generation licence to Belfast Energy Storage Company Limited](#), November 2021; Utility Regulator, [Decision to grant an electricity generation licence for Heron Storage Limited](#), January 2023;

⁶⁰ [Electricity \(Class Exemptions from the Requirement for a Licence\) Order \(Northern Ireland\) 2013](#)

3

Concerns about the safety of BESSs

Although safety incidents for battery energy storage systems (BESSs) are rare, a common concern about the rollout of BESSs that use lithium-ion batteries is the potential fire risk.⁶¹

This section of the briefing sets out the potential causes and consequences of fires at BESS facilities and the number of BESS fires that have occurred. It also discusses how the risk of fire can be minimised. Section 4 looks at how the safety of BESSs is regulated in the UK, concerns about the regulatory framework and the government's response.

3.1

Concerns about fire risks

The research organisation, The Faraday Institution, explains that lithium-ion battery fires are very rare. It notes that “even with billions of lithium-ion cells in circulation, there are very few safety incidents involving them”. It estimates that only one in 40 million battery cells experience failure that results in fire.⁶²

A [scientific paper on the challenges for grid-scale battery energy storage](#) explains that fires can occur at BESS facilities because they contain millions of battery cells. If one cell catches fire and is not shielded or insulated from the rest of the battery, it can cause adjacent cells to catch fire (sometimes called “thermal propagation” or “thermal runaway”).⁶³

Potential causes of battery fires

Lithium-ion batteries can catch fire because of a process called “thermal runaway”. A [scientific review on the safety of lithium-ion batteries](#) explains that thermal runaway occurs when part of a battery is damaged or has a manufacturing fault. This causes the internal components of the battery to decompose, which produces heat. Higher temperatures can cause further decomposition, which in turn produces more heat. Without external cooling, this cycle can result in fire.⁶⁴

⁶¹ Gallagher, [Fire risk at battery energy storage systems](#), August 2021

⁶² The Faraday Institution, [Improving the Safety of Lithium-ion Battery Cells](#) (PDF), July 2023

⁶³ Huang Y and Li J, [Key Challenges for Grid-Scale Lithium-Ion Battery Energy Storage](#) [online via Wiley Online Library], *Advanced Energy Materials*, Vol 12, Issue 48, 2020

⁶⁴ Yuqing Chen and others, [A review of lithium-ion battery safety concerns](#), *Journal of Energy Chemistry*, August 2021, p83-99

The underlying causes of lithium-ion battery damage are stress or misuse. Examples of stress or misuse, which can result in thermal runaway and fire, include:

- Mechanical stress: Defects can be caused during manufacture or external damage (for example, damage caused by natural disasters).
- Thermal stress: If a battery is overheated (for example, due to an external fire), this can lead to fire.
- Electrical failure: A battery could be overcharged due to a fault in the battery management system (the electronic system that monitors and controls charging and discharging of the battery). This can result in too much energy being put into the battery.⁶⁵

A [study commissioned by the government](#) notes that some batteries used in BESSs are ‘second-life batteries’ that were previously used in electric vehicles. The study highlights that second-life batteries could pose a greater risk than new batteries because they could have experienced damage or contain cells from different sources, which could result in electrical failure.⁶⁶

Possible consequences of battery fires

Battery fires also can be difficult to extinguish because batteries contain a lot of fuel to sustain a fire. During thermal runaway, gases are produced which can further fuel the fire and/or lead to an explosion.

The gases produced are toxic, which can pose a risk to site workers and emergency responders if inhaled.⁶⁷ There are also concerns that toxic metals and chemicals can leach into the water used to fight the fire which could then end up in water systems, posing a risk to wildlife.⁶⁸

3.2

How many BESS fires have occurred?

There is no reliable, publicly accessible record of the number of BESS fires that have occurred in the UK or elsewhere.

⁶⁵ The Faraday Institution, [Improving the Safety of Lithium-ion Battery Cells](#) (PDF), July 2023, p2; Huang Y and Li J, [Key Challenges for Grid-Scale Lithium-ion Battery Energy Storage](#) [online via Wiley Online Library], Advanced Energy Materials, Vol 12, Issue 48, 2020

⁶⁶ Office for Product Safety and Standards (OPSS), [Safety of second life batteries in battery energy storage systems](#), January 2023

⁶⁷ Jens Conzen and others, [Lithium ion BESS hazards](#), Journal of Loss Prevention in the Process Industries, Vol 81, February 2023; OPSS, [Safety of second life batteries in battery energy storage systems](#), January 2023, p22

⁶⁸ Ian S. Mylenbusch and others, [Hazards of lithium-ion BESS, mitigation strategies, minimum requirements, and best practices](#), American Institute of Chemical Engineers, June 2023

Incidents in the UK

There have been at least two documented incidents of BESS fires in the UK: [a fire at a BESS site in Liverpool in September 2020](#) and a [fire at a BESS project under construction in Essex in February 2025](#).⁶⁹

A [research paper on the incident by the Electric Power Research Institute](#) (EPRI), an organisation based in the United States which conducts research on the generation, delivery, and use of electricity, notes that BESS design has improved since the Liverpool BESS was installed in 2018:

Some newer system designs use smaller, modularized cabinets with a few racks of batteries. The system layout limits damage because of thermal runaway and allows a more targeted first responder approach in the event of a fire.⁷⁰

Incidents in the rest of the world

An [online database by EPRI](#) estimates that (as of 17 April 2024) 85 BESS fires have occurred since 2011.⁷¹ However, the EPRI database relies on news articles and does not verify their validity, so the list may not be accurate or exhaustive.

Many battery fires recorded in the EPRI database occurred in South Korea: it estimates 38 fires between 2017 and 2022.⁷² The South Korean Ministry of Industry found that the fires were caused by errors during installation, a lack of protection from the environment and a lack of safety systems.⁷³

Despite this number of fires, a [report by EPRI states that the global failure rate for grid-scale BESSs has decreased significantly](#) since 2018, from almost 10 failure events per GW of storage per year, to less than one failure event per GW per year since 2020.⁷⁴

⁶⁹ [Fire at 20MW UK battery storage plant in Liverpool](#), Energy Storage News, September 2020 [accessed 16 April 2024]; [Fire at Statera BESS project in England brought under control, handed back to site management](#), Energy Storage News, 24 February 2025 [accessed 1 May 2025].

⁷⁰ Electric Power Research Institute (EPRI), [Carnegie Road Energy Storage System Failure Response, Recovery, and Rebuild Lessons Learned](#), April 2023, p10

⁷¹ EPRI, [BESS Failure Event Database](#), undated [accessed 10 April 2024]

⁷² EPRI, [BESS Failure Event Database](#), undated [accessed 10 April 2024]

⁷³ Jens Conzen and others, [Lithium ion BESS hazards](#), Journal of Loss Prevention in the Process Industries, February 2023, 104932

⁷⁴ EPRI, [Technology Innovation Spotlight: Lithium Ion Battery Fires in the News](#), October 2023, p2

3.3

How can BESS fires be prevented?

A [scientific review on lithium-ion batteries highlights](#) that the understanding of thermal runaway has improved in recent years, leading to better and more flame-resistant battery designs.⁷⁵

Research has also found that batteries with lithium-iron-phosphate (LFP) cathodes, which are considered safer than those with lithium-nickel-manganese-cobalt-oxide (NMC) cathodes because they are less prone to thermal runaway, make up a greater percentage of the BESS market (see section 1.1 for an overview of cathode chemistry).⁷⁶

A report by the Faraday Institution also notes that the next-generation of batteries used in BESSs (sodium-ion batteries, flow batteries and solid-state batteries) might be safer than currently-used lithium-ion batteries because they are less prone to thermal runaway.⁷⁷

In addition, BESSs have a battery management system to ensure they operate safely and efficiently. A battery management system will usually ensure that batteries are kept within their safe ranges of voltage, current and temperature.⁷⁸

A [scientific paper on the best practices for BESSs](#) also notes that the design of the BESS facilities can also help ensure their safety. In BESS facilities, battery packs can be spaced out and have insulation between the cells to prevent the spread of fire.⁷⁹ BESSs can be also designed with safety features, for example, cooling systems, gas and fire detectors, fire control and suppression systems, explosion control and water supply for emergency services in case a fire does occur.⁸⁰

⁷⁵ Huang Y and Li J, [Key Challenges for Grid-Scale Lithium-Ion Battery Energy Storage](#) [online via Wiley Online Library], Advanced Energy Materials, Vol 12, Issue 48, 2020

⁷⁶ The Faraday Institution and Rho Motion, [Market and Technology Assessment of Grid-Scale Energy Storage required to Deliver Net Zero](#) (PDF), September 2023, p47; Ian S. Mylenbusch and others, [Hazards of lithium-ion BESS, mitigation strategies, minimum requirements, and best practices](#), American Institute of Chemical Engineers, June 2023

⁷⁷ The Faraday Institution and Rho Motion, [Market and Technology Assessment of Grid-Scale Energy Storage required to Deliver Net Zero](#) (PDF), September 2023, p47

⁷⁸ Volts Energies, [BMS in Focus](#), March 2023; BEIS, [Domestic battery energy storage systems](#), October 2020, p6

⁷⁹ Ian S. Mylenbusch and others, [Hazards of lithium-ion BESS, mitigation strategies, minimum requirements, and best practices](#), American Institute of Chemical Engineers, June 2023

⁸⁰ Fire Industry Association, [Guidance on Lithium-Ion Battery Fires](#), January 2021

4

How is the safety of BESSs regulated?

4.1

Overview

While there is no single piece of legislation that sets specific requirements for BESS sites, there is a framework of relevant legislation that applies, set out in more detail in the following sections.

The operators of battery energy storage systems (BESSs) will be subject to a [range of health and safety legislation](#). This includes a general duty for employers, set out under the [Health and Safety at Work etc Act 1974](#) (HSWA), to take reasonable measures to ensure that employees and members of the public are not harmed by the work activity. Operators will also likely be subject to further legislation, including legislation governing the use of electricity, and the management of dangerous substances and explosive atmospheres, depending on the nature of the site. Individual batteries may have to adhere to product safety regulations, for example when second-life batteries are used (see below). The government published guidance on the [health and safety of grid scale electrical energy storage](#) in April 2024.

The government has said grid-scale facilities may also have to comply with requirements out in the [Fire Safety Order 2005](#) (which governs the fire safety of premises during operation and use).⁸¹ There is no guidance by the government or regulatory bodies on whether fire safety requirements set in building regulations (which govern the fire safety of buildings during construction) apply to grid-scale BESSs.⁸² As explained in section 2.1, local authorities are encouraged, but not required to consult fire and rescue services when considering planning applications for BESS sites.

A [gap analysis report on health and safety standards for electricity storage technologies](#) commissioned by the government (June 2020) explains that industry standards published by the British Standards Institute and other international bodies provide guidance for how developers can ensure the safe installation and operation of BESSs.⁸³ There are no statutory provisions that require BESSs to adhere to industry standards, but they can help developers demonstrate compliance with the law.

⁸¹ PQ 31536 [[Batteries: Fire Prevention](#)] 12 July 2021

⁸² North Yorkshire Fire & Rescue Service, [Battery Energy Storage Systems](#), undated; Essex County Fire & Rescue Service, [Fire service guidance for BESS developers](#), undated [accessed 18 March 2024]

⁸³ DESNZ and BEIS, [Electricity storage health and safety standards: Gap analysis](#), last updated August 2023

Concerns about the regulatory framework

Concerns about the fire safety of BESSs and the adequacy of the existing regulatory framework, have been raised in the UK Parliament by MPs from all parties. For example, there were debates on the safety of BESS facilities in June 2022, July 2023 and most recently on 5 June 2025.⁸⁴ Battery energy storage systems were also raised during committee stage debates of the [Product Regulation and Metrology Bill 2024-25](#), the [Planning and Infrastructure Bill 2024-25](#), and there have been several private members' bills on the topic.⁸⁵ Similar concerns about the fire safety of BESSs have also been raised in the Scottish Parliament on a number of occasions.⁸⁶

MPs have called for a stronger, national regulatory framework that specifically addresses the unique risks posed by BESS sites, arguing that general performance-based regulations, which require site owners to ensure risks are managed, are insufficient.⁸⁷ MPs have highlighted challenges for local authorities to manage these applications without specific guidance. There are calls for measures including:

- mandatory requirements for certain safety features such as thermal containment measures and fire suppression
- for fire services to be made statutory consultees for planning applications
- for a mandatory minimum distance from residential premises
- for clearer guidance from the Health and Safety Executive
- for stronger environmental permitting regulations.⁸⁸

Government response

In response to the [debate on safety regulations for BESSs in June 2025](#), Energy Minister Miatta Fahnbulleh said that the government was “confident that the safety risks posed by grid-scale batteries are relatively small and well

⁸⁴ [HC Deb 5 June 2025](#), [Battery Energy Storage Sites: Safety Regulations]; HC Deb [[Solar Farms and Battery Storage](#)] 8 June 2022; HC Deb [[Lithium-ion Battery Storage Facilities: Regulation](#)] 4 July 2023; PQ HL7632 [[Batteries: Fires](#)] 3 May 2023.

⁸⁵ Private members' bills include Lord Redesdale's [Lithium-ion Battery Safety Bill \[HL\]](#) introduced to the House of Lords on September 2024; the [Battery Energy Storage Systems \(Fire Safety\) Bill](#), a presentation bill introduced to the House of Commons by Anna Sabine in October 2024 and Dame Maria Miller's Ten-Minute Rule Bill on [Lithium-Ion Battery Storage \(Fire Safety and Environmental Permits\)](#) introduced to the House of Commons in September 2022.

⁸⁶ SP WA 7 February 2024, [S6W-25333](#); SP WA 4 January 2024, [S6W-24333](#).

⁸⁷ [HC Deb 5 June 2025](#), [Battery Energy Storage Sites: Safety Regulations].

⁸⁸ [HC Deb 5 June 2025](#), [Battery Energy Storage Sites: Safety Regulations]; [HL Deb 20 November 2024](#), [43GC \[Product Regulation and Metrology Bill \[HL\]\]](#); [Battery Energy Storage Systems \(Fire Safety\) Bill](#) (October 2024); [Lithium-Ion Battery Storage \(Fire Safety and Environmental Permits\)](#) (September 2022).

managed”.⁸⁹ She pointed to the framework of legislation that is relevant to BESS sites.

The Minister mentioned some areas where the government could look to strengthen processes:

- on planning, she highlighted that planning practice guidance in England encourages developers to engage with fire services, and stated “there is scope to strengthen the process and build on it” to address some of the concerns that Members raised.⁹⁰
- On health and safety, the Minister pointed to existing regulations that together put in place protections against some of the issues raised, and said “there is scope to think about how we bring this together in a way that is accessible and enforceable”.⁹¹
- On environmental permits, she confirmed the government would consult later in June on whether to include batteries in the environmental permitting regulations.⁹²
- On the minimum distance between BESS sites and occupied buildings, she pointed to [National Fire Chiefs Council guidance](#) recommending a distance of at least 25 meters from occupied buildings, and said “we can look at how we can build on that going forward”.⁹³

The following sections set out more detail on specific aspects of the current regulatory framework applying to BESSs.

4.2 Product safety

A study commissioned by the government noted in 2023 that BESSs might use second-life lithium-ion batteries, for example, batteries that were previously used in electric vehicles or as domestic batteries. If second-life batteries are used in BESSs, the study notes that their safety likely falls under the scope of the UK General Product Safety Regulations 2005.⁹⁴

Like all other products that are placed on the market and are intended for, or will likely be used by, consumers, domestic batteries and batteries used in electric vehicles must comply with the [UK General Product Safety Regulations 2005](#). The 2005 Regulations require producers to ensure that products placed on the market in the UK are safe for consumers. Products must “not present

⁸⁹ [HC Deb 5 June 2025](#), c566 [Battery Energy Storage Sites: Safety Regulations].

⁹⁰ [HC Deb 5 June 2025](#), c567 [Battery Energy Storage Sites: Safety Regulations]

⁹¹ [HC Deb 5 June 2025](#), c567 [Battery Energy Storage Sites: Safety Regulations]

⁹² [HC Deb 5 June 2025](#), c567 [Battery Energy Storage Sites: Safety Regulations]

⁹³ [HC Deb 5 June 2025](#), c568 [Battery Energy Storage Sites: Safety Regulations]

⁹⁴ OPSS, [Safety of second life batteries in battery energy storage systems](#), January 2023

any risk or only the minimum risks” compatible with their use “under normal or reasonably foreseeable conditions of use”.⁹⁵

For further information about the 2005 Regulations, see [government guidance for Great Britain](#) and [separate guidance for Northern Ireland](#).

However, the study commissioned by the government also notes that “there was some uncertainty [...] in terms of the codes, standards and regulations” that govern the testing of second-life batteries, because the re-purposing batteries is “a nascent global market”.⁹⁶ Further, if the batteries used in BESSs are not second-life batteries that were previously used in products covered by the [UK General Product Safety Regulations 2005](#), it is likely that the batteries are outside the scope of the 2005 Regulations.

Government response to concerns about product safety

In response to concerns about the safety of lithium-ion batteries used in energy storage facilities, the government said that it considered BESSs to be covered by “a robust regulatory framework”:

Grid-scale lithium-ion battery energy storage systems are covered by a robust regulatory framework, which requires manufacturers to ensure that products are safe before they are placed on the market and installed correctly, and that any safety issues found after products are on the market or after installations are dealt with.⁹⁷

In response to parliamentary questions in spring 2023, the government said it was undertaking a review of batteries regulations and, as part of this review, it will “consider safety risks associated with all batteries during their lifecycle, including safety concerns associated with lithium-ion batteries”.⁹⁸

The government also said it would consult on batteries regulations by the end of 2023.⁹⁹ In evidence provided to the Lords Environment and Climate Change Committee, the government said the consultation would be published in 2024. It said the consultation would cover the [Batteries and Accumulators \(Placing on the Market\) Regulations 2008](#) and the [Waste Batteries and Accumulators Regulations 2009](#).¹⁰⁰ At the time of publication of this briefing, a consultation has not yet been published.¹⁰¹

⁹⁵ [Regulation 2 of the General Product Safety Regulations 2005](#)

⁹⁶ OPSS, [Safety of second life batteries in battery energy storage systems](#), January 2023

⁹⁷ HL Deb [[Energy Bill \[HL\]](#)] 28 March 2023, c240-241.

⁹⁸ PQ 182649 [[Batteries: Storage](#)] 25 April 2023; PQ HL7592 [[Waste Management: Fires](#)] 2 May 2023

⁹⁹ PQ 182649 [[Batteries: Storage](#)] 25 April 2023; PQ HL7592 [[Waste Management: Fires](#)] 2 May 2023

¹⁰⁰ Lords Environment and Climate Change Committee, [Written evidence: EV strategy: Rapid recharge needed](#) (PDF), November 2023, HL 51 2023-24, Ev ELV0144 [Department for Environment, Food and Rural Affairs, Defra]

¹⁰¹ Defra published a [post-implementation review of the Batteries and Accumulators \(Placing on the market\) Regulations 2008](#) in May 2024, which also stated that it would consult on a review of the UK Battery Regulations (para 10.6).

Concerns about the safety of lithium ion batteries was raised during parliamentary debates on the [Product Regulation and Metrology Bill 2024-25](#), mostly regarding fire risk of consumer products such as e-bikes and e-scooters. The bill does not contain provisions on BESSs specifically. It makes provision about the marketing and use of products in the UK. In response to concerns raised about the fire risk of lithium ion batteries, the government said that there are already strict product safety requirements relating to lithium ion batteries, [but that the bill would enable the government to make regulations to regulate the risks more effectively](#).¹⁰² The [Library briefing on the bill](#) provides further discussion.

3 Batteries regulations

- The [Batteries and Accumulators \(Placing on the Market\) Regulations 2008](#) restrict the use of certain hazardous substances (namely mercury, cadmium, and lead) in batteries and require batteries to be labelled with a 'crossed out wheeled bin' symbol.
- The [Waste Batteries and Accumulators Regulations 2009](#) require battery producers to take back, collect, and recycle automotive, waste industrial, and portable batteries. Retailers and distributors that sell at least 32 kg of portable batteries per year must also provide a take-back service.

For further information about the requirements the regulations place, see [government guidance for manufacturers on placing batteries on the market](#) and [guidance on the collection, treatment and recycling of waste batteries](#).

Both the 2008 Regulations and the 2009 Regulations are derived from EU law but were retained and remain in place following the UK's exit from the EU.

The EU has since updated its Batteries Regulations. The updated regulations require stationary BESSs with a capacity greater than 2 kilowatts per hour to provide evidence that they have undergone certain safety tests, including for thermal runaway.¹⁰³ For further information, see [information published by the European Commission on the Batteries Directive](#).

¹⁰² [HC Deb 839, 8 October 2024](#).

¹⁰³ [Regulation \(EU\) 2023/1542 of the European Parliament and of the Council, 12 July 2023](#); European Commission, [Batteries](#), undated [accessed 28 March 2024]

4.3

Fire safety

Fire safety during construction: Building regulations

Fire safety requirements for new buildings in England and Wales are set out in [Part B of Schedule 1 of the Building Regulations 2010](#). Part B requires buildings to be designed and constructed to limit the spread of fire within and between buildings, to have systems that give early warning of fire, and to allow fire and rescue services to access a building and assist them in fighting a fire.¹⁰⁴

Whether the fire safety requirements set out in building regulations apply to BESSs will depend on whether the modules (stacks of batteries) are housed in a building and, if they are housed in a building, whether the building is exempt:

- [Regulation 2](#) states that building regulations apply to “any permanent or temporary building but not any other kind of structure or erection”.
- Under [Schedule 2](#), detached buildings are exempt from the building regulations if people do not normally enter them or they only enter intermittently to inspect or maintain plants or machinery.¹⁰⁵

Fire safety requirements for new buildings in Scotland and Northern Ireland are set out in the [Building \(Scotland\) Regulations 2004](#) and in the [Building Regulations \(Northern Ireland\) 2012](#) respectively. In Scotland and Northern Ireland, detached buildings are also exempt if people do not usually enter them or only intermittently enter inspect or maintain plants or machinery.¹⁰⁶

There is no guidance in any part of the UK on whether fire safety requirements set in building regulations apply to grid-scale BESSs. CROSS (Collaborative Reporting for Safer Structures), a body which is supported by the Institutions of Structural and Civil Engineers, states that “large-scale BESS are likely to be outside of the scope of ADB [Approved Document B]”.¹⁰⁷ Approved Document B is published by the government to support Part B of the Building Regulations 2010 in England.

Fire safety during operation and use

The fire safety of non-domestic premises is governed during operation by the [Regulatory Reform \(Fire Safety\) Order 2005](#) in England and Wales. The Order designates the person in control of a premises (for example, the employer in a

¹⁰⁴ [Part B of Schedule 1 of the Building Regulations 2010](#)

¹⁰⁵ [Regulation 2 of the Building Regulations 2010](#); [Schedule 2 of the Building Regulations 2010](#)

¹⁰⁶ [Schedule 1 of the Building \(Scotland\) Regulations 2004](#); [Schedule 2 of the Building Regulations \(Northern Ireland\) 2012](#)

¹⁰⁷ Collaborative Reporting for Safer Structures, [Battery Energy Storage System concerns](#), May 2023

workplace) as the ‘responsible person’ for fire safety. The responsible person does not have to be an individual; it can be an organisation.¹⁰⁸

Responsible persons have two main duties under the Fire Safety Order:

- to carry out and regularly review fire risk assessments to identify what is needed to prevent fires and protect the people in and around a building (such as employees, visitors, and contractors) in the event of a fire.
- to put in place and maintain fire safety measure. Examples of fire safety measures include, “where reasonably necessary”, fire detectors and alarms, fire suppression systems (such as sprinklers or gas extinguishing agents), and firefighting equipment (such as fire extinguishers).¹⁰⁹

The Fire Safety Order imposes further duties on persons who are responsible for premises where ‘dangerous substances’ are present. There is no defined list of which substances are considered dangerous, but they include explosive materials, flammable gases, and any other substances that create a risk as a result of their “physico-chemical or chemical properties”.¹¹⁰

In premises where ‘dangerous substances’ are present, the Fire Safety Order also provides that responsible persons must eliminate or reduce related risks “so far as is reasonably practicable”. Where they cannot eliminate risks, they must put in place measures to control these risks and mitigate the potentially detrimental effects of a fire. Responsible persons must also share information on their emergency procedures with local fire and rescue services so they can prepare their own response procedures and precautionary measures.¹¹¹

The UK Government has indicated the Fire Safety Order applies to grid-scale BESSs.¹¹² However, it has not explicitly said whether lithium-ion or other batteries used in BESS are ‘dangerous substances’ for the purposes of the Fire Safety Order. The government has said that there are statutory requirements to notify fire services for ‘large-scale’ battery sites.¹¹³ The [Dangerous Substances \(Notification and Marking of Sites\) Regulations 1990 \(NAMOS\)](#), requires the operators of sites that hold 25 tonnes or more of a dangerous substance to notify their local fire and rescue service in writing, and to display signs.¹¹⁴

¹⁰⁸ [Article 3 of the Regulatory Reform \(Fire Safety\) Order 2005](#); Home Office, [A guide for persons with duties under fire safety legislation](#), September 2023, paras 2.2-2.23

¹⁰⁹ [Part 2 of the Regulatory Reform \(Fire Safety\) Order 2005](#)

¹¹⁰ [Article 2 of the Regulatory Reform \(Fire Safety\) Order 2005](#); Home Office, [A guide for persons with duties under fire safety legislation](#), September 2023, paras 20.1-20.13

¹¹¹ [Article 12 of the Regulatory Reform \(Fire Safety\) Order 2005](#); [Article 16 of the Regulatory Reform \(Fire Safety\) Order 2005](#)

¹¹² PQ 31536 [[Batteries: Fire Prevention](#)] 12 July 2021; PQ 31537 [[Health and Safety: Batteries](#)] 12 July 2021

¹¹³ PQ 31536 [[Batteries: Fire Prevention](#)] 12 July 2021.

¹¹⁴ DESNZ, [Grid scale electrical energy storage systems: health and safety](#), 18 April 2024.

The fire safety of for non-domestic premises in Scotland and Northern Ireland is governed by the [Fire Safety \(Scotland\) Regulations 2006](#) and the [Fire Safety Regulations \(Northern Ireland\) 2010](#) respectively.

The Fire Safety Order is enforced by local fire and rescue services in England and Wales. For further information about their powers, see [guidance on the enforcement of, and sanctions for non-compliance with the Fire Safety Order](#). Fire safety regulations in Scotland and Northern Ireland are enforced by the Scottish Fire and Rescue Service and Northern Ireland Fire and Rescue Service respectively.

Government response to concerns about fire safety

In August 2023, in response to concerns about the fire safety of BESSs, the government updated its planning practice guidance to encourage developers and local planning authorities in England to consult their local fire and rescue service in preparing and deciding on planning applications for BESSs.¹¹⁵

Similar guidance has not been published by the devolved administrations.

The updated guidance also refers local planning authorities and developers to [guidance on grid-scale BESS by the National Fire Chiefs Council](#) (PDF). The guidance advises operators of BESSs to install “an effective and appropriate method” to detect faults and thermal runaway and fire suppression systems to prevent or limit spread of fire. The National Fire Chiefs Council also advises developers to consider designing their site to prevent of the spread of fire and to provide safe access and water supply for fire and rescue services.¹¹⁶

The National Fire Chiefs Council is consulting [revised guidance for fire and rescue services on BESS](#). It published a draft of its new guidance in July 2024 and the consultation closed on 22 August 2024.

4.4 Health and safety

Operators of battery energy storage systems (BESSs) will be subject to a range of health and safety legislation. This includes a general duty for employers, set out under the [Health and Safety at Work etc Act 1974](#) (HSWA), to take reasonable measures to ensure that employees and members of the public are not harmed by the work activity. The legislation is not prescriptive about which risks must be considered or how they should be mitigated. It is for employers to consider how legislation applies to their activities and take the necessary measures to mitigate risks. Other relevant health and safety legislation is set out below.

¹¹⁵ DLUHC and MHCLG, [Renewable and low carbon energy](#), last updated August 2023, paras 34-35

¹¹⁶ National Fire Chiefs Council (NFCC), [Grid Scale Battery Energy Storage System planning – Guidance for FRS](#) (PDF), last updated October 2023

In Great Britain, enforcement for health and safety law is shared between the Health and Safety Executive (HSE), local authorities and specific industry regulators.

The Health and Safety Executive has a [Grid-scale battery energy storage systems](#) webpage, which sets out relevant health and safety legislation for BESSs in Great Britain as including:

- The [Health and Safety at Work Act 1974](#), which places legal duties on employers to manage risks to employees, other workers and the public.
- The [Management of Health and Safety at Work Regulations 1999](#), which requires employers to identify potential risks to employee health and safety and appropriately manage them. The Act is not prescriptive about which risks must be considered, it is for employers to assess their specific situation and take the necessary measures to mitigate risks. For further information on how employers should assess and manage risks, see [guidance by the HSE](#).
- The [Electricity at Work Regulations 1989](#), which requires all work activities that uses or might be affected by electricity to be done safely. All foreseeable risks must also be assessed and minimised as far as possible. For further information, see the [HSE's guidance on the 1989 Regulations](#).
- The [Dangerous Substances and Explosive Atmospheres Regulations 2002](#), which requires employers to identify the risks posed by dangerous substances (which are substances that may be explosive, flammable or oxidising) in the workplace. Employers must put in place measures to remove risks to the safety of employees and others or, if it is not possible to fully remove risks, mitigate them and limit the effects of potential incidents involving the dangerous substances.
- The [Construction Design and Management Regulations 2015](#), which sets requirements to ensure construction projects are carried out in a way that secures health and safety, from conception to completion.
- The [Dangerous Substances \(Notification and Marking of Sites\) Regulations 1990 \(NAMOS\)](#), which requires the operators of sites that hold 25 tonnes or more of a dangerous substance to notify their local fire and rescue service in writing, and to display signs.

Similar health and safety regulations exist in Northern Ireland.¹¹⁷

¹¹⁷ [Management of Health and Safety at Work Regulations \(Northern Ireland\) 2000](#); [Electricity At Work Regulations \(Northern Ireland\) 1991](#); [Dangerous Substances and Explosive Atmospheres Regulations \(Northern Ireland\) 2003](#)

Government response

The government said the HSE considered the current regulatory framework to be “sufficient and suitably robust in relation to lithium-ion batteries”.¹¹⁸

The government said it had set up an industry-led electricity storage health and safety governance group in 2018 to ensure “an appropriate, robust and future-proofed health and safety framework is sustained as [...] electricity storage deployment increases”. In March 2023, the government said it was working with this group to develop guidance for grid-scale storage.¹¹⁹

The government published guidance on the [health and safety of grid scale electrical energy storage](#) in April 2024.

4.5

Which regimes do not apply to BESS?

Exclusion from the Control of Major Accident Hazard (COMAH) Regulations

The [Control of Major Accident Hazards \(COMAH\) Regulations 2015](#) intend to prevent and mitigate the effects of major accidents involving dangerous substances which can cause serious damage or harm to people and/or the environment. The ‘dangerous’ substances which are covered by the [COMAH are set out in Schedule 1](#).¹²⁰

If dangerous substances listed in Schedule 1 are kept or used on a site and meet a certain threshold, the COMAH requires site operators to take certain steps to prevent major accidents. They must notify the relevant ‘competent authority’ (usually, the HSE) of the details of their operation, prepare a major accidents prevention policy and develop a safety management system. For further information, see the [HSE’s guide to the COMAH](#).

The COMAH applies to Great Britain. Similar regulations apply in Northern Ireland, where they are enforced by the Health and Safety Executive for Northern Ireland.¹²¹

Government response

When asked why grid-scale lithium-ion BESSs are not covered by the COMAH, the government responded that lithium-ion batteries are articles rather than substances for the purposes of the COMAH. The government also said the HSE

¹¹⁸ PQ 31537 [[Health and Safety: Batteries](#)] 12 July 2021; HC Deb [[Topical Questions](#)] 27 February 2024, c126

¹¹⁹ HL Deb [[Energy Bill \[HL\]](#)] 28 March 2023, c240-241

¹²⁰ [Control of Major Accident Hazards \(COMAH\) Regulations 2015](#); HSE, [Control Of Major Accident Hazards Regulations 2015](#), undated [accessed 16 April 2024]

¹²¹ Health and Safety Executive for Northern Ireland, [Control of Major Accident Hazards Regulations](#), undated [accessed 16 April 2024]

considered that “the current regulatory framework is sufficient and suitably robust”:

The Control of Major Accident Hazards Regulations 2015 (COMAH) apply to dangerous substances as classified by the Classification, Labelling and Packaging Regulations 2008. Lithium-ion batteries are considered to be articles, rather than substances, and are therefore outside of the scope of the COMAH.

The Health and Safety Executive considers that the current regulatory framework is sufficient and suitably robust in relation to lithium-ion batteries and battery energy storage systems.¹²²

Environmental permits

Under the [Environmental Permitting \(England and Wales\) Regulations 2016](#), certain ‘regulated facilities’ (such as landfill sites, combustion plants and waste incinerators) require an environmental permit. In general, a permit is needed for activities that could pollute the air, water or land, increase flood risk, or adversely affect land drainage.¹²³

For further information about which installations and activities require an environmental permit, see government [guidance on environmental permitting](#) and on [the definition of ‘regulated facilities’](#).

Environmental permits are issued by the Environment Agency in England and Natural Resources Wales in Wales. A permit will set out which mandatory that operators have to comply with to protect the health of local communities and the local environment. The Environment Agency and Natural Resources Wales have powers to inspect regulated facilities to ensure operators comply with their permit. If they are not compliant, they can take enforcement action.

Similar regulations apply in Scotland and Northern Ireland.¹²⁴

Government response

In response to a question in the House of Lords in May 2023, the government explained that BESS were not classed as ‘installations’ (one type of ‘regulated facilities’) for the purposes of environmental permitting regulations. It said it was “in discussions” with the Environment Agency “on the issue”.¹²⁵

In a debate in September 2023, the Parliamentary Under-Secretary of State for Energy Security and Net Zero, Andrew Bowie, acknowledged “the concerns

¹²² PQ 29036 [[Health and Safety: Batteries](#)] 7 July 2021

¹²³ [Environmental Permitting \(England and Wales\) Regulations 2016](#); Defra, [Environmental permitting guidance: Core guidance](#), last updated August 2020

¹²⁴ Scottish Environment Protection Agency, [Environmental Authorisations \(Scotland\) Regulations 2018](#), undated [accessed 16 April 2024]; Northern Ireland (NI) Business Info, [Introduction to environmental permits and licences](#), undated [accessed 16 April 2024]

¹²⁵ PQ HL7632 [[Batteries: Fires](#)] 3 May 2023; Environment Agency, [Order Granting Development Consent for the West Burton Solar Project: Responses to the Examining Authority’s Written Questions](#) (PDF), last updated January 2024

surrounding the [...] environmental impact of battery energy storage at grid scale”. He said that the government would consult on including BESSs in environmental permitting regulations “at the earliest opportunity”.¹²⁶

The government’s December 2024 Clean Power Action Plan stated that Defra would consult on [including grid-scale batteries within the Environmental Permitting Regulations by June 2025](#).¹²⁷ At the time of publication of this briefing, a consultation has not yet been published.

¹²⁶ HC Deb [[Energy Bill \[Lords\]](#)] 5 September 2023, c322-323

¹²⁷ DESNZ, [Clean Power Action Plan](#), December 2024.

5

Barriers to the development of BESSs

Barriers to the development of battery energy storage systems (BESSs) in the UK include a lack of manufacturing capacity for batteries in the UK and the UK's reliance on international markets for critical minerals that are used in batteries (such as lithium). Development might also be hindered by the cost of establishing BESSs and delays to grid connections.

5.1

Investment in, and funding for, BESSs

BESS companies and investors have reported difficult investment conditions in the UK because current market arrangements do not provide sufficient ways to earn revenue.¹²⁸

In a [report on long-duration energy storage](#) (March 2024), the House of Lords Science and Technology Select Committee said the commercial development of long-duration energy storage required financial support because of high upfront capital costs and uncertain revenue streams:

Long-duration storage facilities can take 7–10 years to build and require upfront capital investment. Developers need a clear business case, supporting infrastructure such as grid connections, and financial support in order to invest. [...] Commercial development of long-duration storage needs financial support mechanisms for investors due to high upfront capital costs, long lead times for planning approvals and construction, and uncertain revenue streams.¹²⁹

In the report, the committee also expressed concern that a focus on lithium-ion storage projects meant that policymakers, investors and regulators had neglected the need for other longer-duration technologies.¹³⁰

In a 2021 call for evidence, the government sought views on the [barriers to deployment of large-scale and long-duration electricity storage](#) and how these might be addressed. It found that high upfront capital costs, long build

¹²⁸ Energy Storage News, [UK's battery storage assets subject to 'weak revenue environment', says Gresham House](#), 1 February 2024

¹²⁹ Lords Science and Technology Committee, [Long-duration energy storage: Get on with it](#) (PDF), 13 March 2024, HL 68 2023-24, p3-4

¹³⁰ Lords Science and Technology Committee, [Long-duration energy storage: Get on with it](#) (PDF), 13 March 2024, HL 68 2023-24, para 31

times, lack of a track record of the technology, lack of revenue certainty, and lack of market signals were some of the main barriers.¹³¹

Cap and floor scheme

In its response to the call for evidence (August 2022), the government said that it would work with Ofgem, the energy regulator, to develop “an appropriate policy to enable investment in large-scale, long-duration electricity storage”.

In 2024 the government consulted on [proposals to enable investment in long duration electricity storage \(LDES\)](#). The consultation set out plans to introduce a ‘cap and floor’ financial support mechanism to overcome investment barriers.¹³² This will provide a guaranteed minimum revenue stream (the floor) and a limit on revenues (the cap) for long duration energy storage.¹³³

Ofgem published its [Long Duration Electricity Storage technical document](#) (March 2025), which sets out the final details of the scheme, including how it will operate, eligibility criteria and potential capacity needed. This sets out that there will be a twin track approach for projects deliverable in 2030 and 2033. The first application window is expected to have a capacity range of 2.7 to 7.7 GW. Eligible LDES projects will need to be capable of delivering 8 hours of continuous power and have a capacity of 100 megawatts (MW) or 50 MW depending on the type of technology. The scheme will have a duration of 25 years. The scheme will be funded by through electricity network charges on energy suppliers. The [Planning and Infrastructure Bill 2024-25](#) includes provisions requiring Ofgem to introduce the scheme.

The first window for applications opened in April 2025; Ofgem aims to approve the first round of projects for Q2 2026.¹³⁴

Competitive funding for LDES

In 2021, the government launched a [competition to support the development of innovative longer duration energy storage projects](#).¹³⁵ It provided over £69 million of capital funding to support a range of energy storage technologies (but not widely deployed technologies such as lithium-ion batteries) under two streams:

¹³¹ BEIS, [Facilitating the deployment of large-scale and long-duration electricity storage: call for evidence](#), last updated August 2022

¹³² DESNZ, [Long duration electricity storage: Proposals to enable investment](#), January 2024

¹³³ It would be a bespoke, short-medium term mechanism as the government’s wider [Review of Electricity Market Arrangements \(REMA\)](#) has ruled out a general cap and floor mechanism for renewable generation. For further information, see: DESNZ, [Review of Electricity Market Arrangements: Second Consultation](#), March 2024

¹³⁴ Ofgem, [Long Duration Electricity Storage: cap and floor application window 1](#), 8 April 2025.

¹³⁵ DESNZ and BEIS, [Longer Duration Energy Storage Demonstration \(LDES\) competition](#), last updated April 2023

- Stream 1 of the competition was for actual demonstration projects that would be completed in March 2025. Successful projects in Stream 1 were announced in February 2021 and included a vanadium flow battery and a hydrogen storage project.
- Stream 2 of the competition was for prototype demonstration projects that would be completed in September 2024. Successful projects in Stream 2 included flow batteries and copper/zinc batteries.¹³⁶

In its [UK battery strategy](#) (November 2023), the government also highlighted that government-backed finance options “to help UK companies develop and scale their business” were available through the British Business Bank, the UK Infrastructure Bank (now the National Wealth Fund) and UK Export Finance.¹³⁷ For further information, see, for example, a [strategy update on short duration battery energy storage systems published by the UK Infrastructure Bank](#) (PDF) in September 2023.

5.2 Manufacturing capacity

In its [report on batteries for electric vehicle manufacturing](#) (November 2023), the Business and Trade Committee said that “the UK faces a gigafactory gap, because of insufficient domestic manufacturing capacity”. Gigafactories are large-scale battery manufacturing sites that produce batteries mainly for the electric vehicle market but also for electrical storage applications.¹³⁸

Most rechargeable batteries are currently manufactured in East Asia.¹³⁹ There is only one existing gigafactory in the UK, in Sunderland. It is owned by Envision AESC UK, part of a Chinese-owned battery technology company that is headquartered in Japan.¹⁴⁰

In its report, the Business and Trade Committee called on the government to explain “how it will ensure the UK develops the capacity to build the battery supply needed [...] to achieve our targets for Net Zero”. It recommended that the government offer more financial support, address skills gaps and identify sites for gigafactories to attract investors and encourage the development of gigafactories.¹⁴¹

¹³⁶ DESNZ and BEIS, [Longer Duration Energy Storage Demonstration Programme: Successful projects](#), last updated April 2023

¹³⁷ DBT, [UK battery strategy](#), last updated December 2023

¹³⁸ Business and Trade Committee, [Batteries for electric vehicle manufacturing](#) (PDF), 14 November 2023, HC 196 2023-24, pp3-5

¹³⁹ DBT, [UK battery strategy](#), last updated December 2023

¹⁴⁰ Business and Trade Committee, [Batteries for electric vehicle manufacturing](#) (PDF), 14 November 2023, HC 196 2023-24, pp4-5

¹⁴¹ Business and Trade Committee, [Batteries for electric vehicle manufacturing](#) (PDF), 14 November 2023, HC 196 2023-24, pp4-5

In its response to the committee’s report, the government pointed to its [UK battery strategy](#) (published in November 2023).¹⁴² In the strategy, it said its aim was for the UK to “have a globally competitive battery supply chain that supports [...] the net zero transition” by 2030 and for the UK to become “a world leader in sustainable battery design and manufacture”.¹⁴³

There has been some recent investment in gigafactories in the UK, including:

- Envision AESC UK is building a second gigafactory in Sunderland which will have an initial capacity of 12 GWh.
- Tata Group announced in July 2023 that it would invest £4 billion in a gigafactory that is expected to commence production in 2026.¹⁴⁴

The Library briefing, [Electric vehicles and infrastructure](#) (July 2024) includes further information about battery manufacturing (see section 3.4).

5.3

Delays to grid connections

BESSs are connected to either the higher-voltage transmission system (132 kilovolts or above), which is operated mainly by NESO in Great Britain¹⁴⁵, or the lower-voltage distribution system (less than 132 kilovolts), which is operated by licensed distribution network operators (DNOs).¹⁴⁶

The [Energy Network Association](#) is the trade association for energy networks in the UK. It is also responsible for ensuring connections are compliant with technical and legal requirements. Ofgem, the independent energy regulator, is responsible for ensuring that developers looking for a new connection to the grid receive good service at a fair price.¹⁴⁷

Developers of all types of low-carbon energy infrastructure have reported long lead times for connection to the electricity transmission or distribution networks in Great Britain.

In January 2023, the head of hydrogen of the energy company Centrica told the then Business, Energy and Industrial Strategy (BEIS) Committee that, as a

¹⁴² DBT, [Batteries for electric vehicle manufacturing: Government Response to the Committee’s First Report of Session 2023-24](#) (PDF), 14 February 2024, HC 547 2023-24

¹⁴³ DBT, [UK battery strategy](#), last updated December 2023

¹⁴⁴ Business and Trade Committee, [Batteries for electric vehicle manufacturing](#) (PDF), 14 November 2023, HC 196 2023-24, paras 18-22

¹⁴⁵ National Grid ESO are the electricity system operator for Great Britain. Under the Energy Act 2023, National Grid ESO will become the [National Energy System Operator \(NESO\)](#), a new independent public corporation responsible for planning Britain’s electricity and gas networks and operating the electricity system, in summer 2024.

¹⁴⁶ National Grid, [Electricity Transmission: How can I connect?](#), undated [accessed 16 April 2024]

¹⁴⁷ Energy Network Association (ENA), [Connecting commercial generation to the electricity networks](#), undated [accessed 16 April 2024]; Ofgem, [A guide to electricity distribution connections policy](#), April 2024

result of grid constraints, “hardly any more grid-scale batteries” could be put “on the network”. The chief executive of Regen, a not-for-profit organisation which seeks to support the transition to net zero, told the BEIS Committee that there was a 15-year delay to putting renewable energy projects and batteries on the grid “in most parts of the country”.¹⁴⁸

In its [UK battery strategy](#), the government also noted that respondents to its call for evidence had raised concerns that long lead times to accessing the grid could also have knock-on effects on the “cost of manufacturing in the UK and represent notable barriers to investment”.¹⁴⁹

Government actions

To address the issue of delays in grid connections, the government appointed Nick Winser as the UK’s first Electricity Networks Commissioner in July 2022. In August 2023, Nick Winser published an [independent report on accelerating electricity transmission network deployment](#). It out recommendations to half the connection time of new projects to around seven years.¹⁵⁰

In response to the Nick Winser’ report, the government and Ofgem published an [action plan](#) in November 2023. The government said that it aimed for “a significant majority of projects” to be connected to the transmission network by the date they requested and to reduce average delays for projects from five years to six months. The government and Ofgem also said they would work with the electricity system operator and network companies to improve connections to the distribution network.¹⁵¹

Other steps to reduce grid connection delays include:

- In November 2023, Ofgem also announced a [new policy to clear ‘zombie projects’ and cut waiting time for electricity grid connections](#). The policy ended the existing ‘first-come, first-served’ system and allowed stalled or speculative projects to be forced out of the connections queue.¹⁵²
- NESO has a [five-point plan to improve the connection process](#) at the transmission level in the short term. In November 2023, NESO announced that the [connection dates of 10 gigawatts of battery projects had been accelerated](#) as part of this plan.¹⁵³

¹⁴⁸ Business, Energy and Industrial Strategy (BEIS) Committee, [Oral evidence: Decarbonisation of the power sector](#), 24 January 2023, HC 283 2022-23, Q215

¹⁴⁹ DBT, [UK battery strategy](#), last updated December 2023

¹⁵⁰ BEIS, [New Electricity Networks Commissioner appointed to help ensure home-grown energy for Britain](#), July 2022; DESNZ, [Accelerating electricity transmission network deployment: Electricity Networks Commissioner’s recommendations](#), last updated August 2023

¹⁵¹ DESNZ and Ofgem, [Electricity networks: Connections action plan](#), November 2023

¹⁵² Ofgem, [Ofgem announces tough new policy to clear ‘zombie projects’ and cut waiting time for energy grid connection](#), November 2023

¹⁵³ National Grid ESO, [Our five-point plan](#), undated [accessed 16 April 2024]

- In April 2025, [Ofgem approved NESO's plans to reform the connections process](#). This will prioritise connections for projects that are “[ready to progress, and which are needed to deliver clean power by 2030](#)”.
- The Energy Network Association have set out [plans to improve customer connections](#) to the grid at distribution level.¹⁵⁴

5.4

Critical minerals

The manufacture of batteries requires a reliable supply of a range of ‘critical minerals’ (such as lithium, cobalt, nickel and graphite). In a report on [the role of critical minerals in clean energy transitions](#) (May 2021), the International Energy Agency estimated that demand for minerals for grid-scale electricity storage would grow 30 times between 2020 and 2040.¹⁵⁵

Supplies of critical materials can be from domestic or international markets.

The possibility of extracting lithium in Cornwall is currently being explored. In its [report on batteries for electric vehicle manufacturing](#), the Business and Trade Committee said nickel and cobalt were also found in the UK, but it was not clear whether they could be extracted “at a commercial scale”.¹⁵⁶

Other minerals needed to make batteries do not exist in the UK, however. In its [report on critical mineral resilience](#) (December 2023), the Foreign Affairs Committee noted that “the UK is almost completely dependent on imports for critical minerals and mineral products”. It highlighted that China is currently the “dominant global player” in critical minerals supply chains.¹⁵⁷

International markets can be subject to volatility and social, economic and geopolitical issues.¹⁵⁸ In its [critical minerals strategy](#), the government said that its reliance on international markets meant critical minerals were “vulnerable to market shocks, geopolitical events and logistical disruptions”.¹⁵⁹

¹⁵⁴ ENA, [Connecting to the networks](#), undated [accessed 18 April 2024]

¹⁵⁵ International Energy Agency (IEA), [The Role of Critical Minerals in Clean Energy Transitions](#), May 2021, p104

¹⁵⁶ Business and Trade Committee, [Batteries for electric vehicle manufacturing](#) (PDF), 14 November 2023, HC 196 2023-24, para 39

¹⁵⁷ Foreign Affairs Committee, [A rock and a hard place: Building critical mineral resilience](#) (PDF), 15 December 2023, HC 371 2023-24, para 6

¹⁵⁸ For further discussion, see: POST briefing, [Mining and the sustainability of metals](#), February 2022; POSTnote, [Access to Critical Materials](#), September 2019.

¹⁵⁹ DBT and BEIS, [UK Critical Minerals Strategy](#), last updated March 2023

Government actions

The government published a [critical minerals strategy](#) in July 2022. It said the strategy aimed to:

- maximise the UK’s domestic supply of critical minerals (including through mining, refining and manufacturing) and accelerate the recovery, reuse and recycling of critical minerals in the UK.
- collaborate with international partners to diversify supply and improve the resilience of critical minerals supply to the UK.
- make international markets more transparent and responsible, including by boosting environmental, social and governance performance and reducing vulnerabilities.¹⁶⁰

The government also established the [Critical Minerals Intelligence Centre](#) in July 2022 to support government and industry to mitigate risks to the supply of critical minerals, domestically and internationally. The centre is led by the British Geological Survey and supported by the government.

In March 2023, the government published an update to its [critical minerals strategy \(titled ‘critical minerals refresh’\)](#). In the ‘refresh’, the government announced that it would:

- launch a Critical Minerals Task and Finish Group to investigate the critical mineral dependencies and vulnerabilities across UK industry sectors. The group published [a report in December 2023 that assessed environmental, social, and governance risks and supply risks](#) for critical materials across six sectors, including energy.¹⁶¹
- improve the UK’s collaboration with international partners, including through partnerships with Canada and South Africa and engagement through the Minerals Security Partnership, International Energy Agency and the G7.¹⁶²

In its [report on critical mineral resilience](#) (December 2023), the Foreign Affairs Committee raised concerns that the government’s critical minerals strategy was “too broad to be helpful as a guide to industry”. It also said the strategy did not “convey the sense of urgency and need for immediate, decisive action which is of paramount importance” to deliver the UK’s net zero commitments and improve its resilience to global supply chains.¹⁶³

¹⁶⁰ DBT and BEIS, [UK Critical Minerals Strategy](#), last updated March 2023

¹⁶¹ DBT, [Task and Finish Group: Industry resilience for critical minerals](#), December 2023

¹⁶² DBT and BEIS, [UK Critical Minerals Strategy](#), last updated March 2023

¹⁶³ Foreign Affairs Committee, [A rock and a hard place: Building critical mineral resilience](#) (PDF), 15 December 2023, HC 371 2023-24, p3

The committee called on the government to set “specific targets for priority sectors”, building on its critical minerals strategy and refresh. The committee also said a “detailed implementation plan” was needed.¹⁶⁴

In its response to the committee’s report, the government said its critical minerals strategy and refresh demonstrated its “commitment to improving the resilience of critical minerals supply chains”.¹⁶⁵

The Labour Government has said that the Department for Business and Trade will publish a new critical minerals strategy in 2025.¹⁶⁶ It commissioned an [analysis of UK critical minerals capabilities](#).¹⁶⁷

¹⁶⁴ Foreign Affairs Committee, [A rock and a hard place: Building critical mineral resilience](#) (PDF), 15 December 2023, HC 371 2023-24, paras 31-35

¹⁶⁵ Foreign, Commonwealth and Development Office (FCDO), [A rock and a hard place: Building critical mineral resilience: Government Response to the Committee’s First Report](#) (PDF), 8 March 2024, HC 583 2023-24

¹⁶⁶ [HC Deb 3 December 2024, c40WH](#).

¹⁶⁷ DBT, [UK critical minerals midstream and recycling capability report](#), 2 April 2025.

6

Further reading

Research on BESSs

- Parliamentary Office for Science and Technology (POST), [Longer duration energy storage](#), December 2022
- Royal Society, [Large-scale electricity storage](#), September 2023
- Faraday Institution and Rho Motion, [Market and Technology Assessment of Grid-Scale Energy Storage](#) (PDF), September 2023

Committee reports

- Business and Trade Committee, [Batteries for electric vehicle manufacturing](#), November 2023
- Lords Science and Technology Committee, [Long-duration energy storage](#), March 2024
- Lords Science and Technology Committee, [Role of batteries and fuel cells in achieving Net Zero](#), July 2021

Government reports

- DBT, [UK battery strategy](#), last updated December 2023
- DESNZ, [Long duration electricity storage: Scenario deployment analysis](#), January 2024
- DESNZ and BEIS, [Benefits of long-duration electricity storage](#), August 2022
- DESNZ and BEIS, [Electricity storage health and safety standards: Gap analysis](#), last updated August 2023
- OPSS, [Safety of second life batteries in battery energy storage systems](#), January 2023

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OPEN Coastal wetland deposition of cathode metals from the world's largest lithium-ion battery fire

Ivano W. Aiello^{1✉}, Charlie Endris¹, Steven Cunningham¹, Monique Fountain², Maxime M. Grand¹, Wesley Heim¹, Amanda S. Kahn¹ & Kerstin Wasson²

Fires at lithium-ion battery storage facilities pose emerging environmental risks that remain largely undocumented under real-world conditions. Following a major fire at the world's largest Battery Energy Storage System (BESS) in Moss Landing, California, we conducted rapid, high-resolution soil surveys to quantify metal fallout in adjacent estuarine wetlands. Field-portable X-ray fluorescence (FpXRF), validated by SEM/EDS, laboratory XRF, and ICP-MS, revealed a significant but transient surface enrichment of nickel (Ni), manganese (Mn), and cobalt (Co). This enrichment had Ni:Co mass ratios near 2:1 serving as a geochemical fingerprint of NMC-type cathode materials. The metals were confined to a shallow surface layer (<5 mm). Surface concentrations declined rapidly following precipitation and tidal inundation. The fallout's thin, transient and patchy distribution would have eluded standard coring methods but was detected through spatially intensive FpXRF sampling, highlighting the importance of rapid detection and the mobilization of metals into wetland ecosystems. These findings underscore the need for adaptive environmental monitoring following battery fires and raise critical considerations for ecosystem protection and infrastructure as energy storage systems expand.

Rapid growth of distributed energy storage systems in recent years reflects the global need to store power from renewable energy sources and to regulate electrical systems^{1–3}. Lithium-ion batteries (LIBs) are the most widely used type of electrochemical energy storage, as they offer high energy and power density compared to other battery technologies⁴. However, electrochemical energy storage and the use and disposal of LIBs involves inherent risks, such as thermal runaway⁵ which can lead to the release of potentially toxic compounds from battery materials⁶, and localized deposition of battery-associated metals in adjacent ecosystems⁷, with, potentially, long-term implications for terrestrial, aquatic, and human health.

Establishing robust environmental baselines in areas surrounding energy storage systems and achieving adequate spatial and temporal coverage to identify contamination after emergency release are both logistically difficult and often cost-prohibitive. In this context, portable and cost-effective technology such as X-ray fluorescence (FpXRF) offers a means of collecting high-density data, serving as a valuable complement to traditional laboratory-based analytical methods.

On 16 January 2025, a large fire engulfed the largest lithium-ion battery (LIB) Battery Energy Storage System (BESS) in the world, burning actively for at least 2 days. This was followed by a smaller reignition on 18 February 2025. Owned by Vistra Corporation, the BESS is in Moss Landing, California, immediately adjacent to Elkhorn Slough, a Ramsar site recognized as a wetland of international importance⁸. The fire affected the core of the facility (Phase 1) which had a capacity of 300 MW/1200 megawatt-hours (MWh) and was equipped with LG Energy Solution's TR1300 battery rack systems⁹. The fire destroyed approximately 75% of the facility¹⁰ and produced a smoke plume visible from tens of kilometers away, depositing ash and soot across the surrounding area (Fig. 1a). Due to potential toxicity, including possible exposure to hydrogen fluoride, evacuation orders and road closures were issued. Residents were permitted to return 2 days after the fire began¹¹.

Controlled experiments show Li-ion battery fires emit metal-bearing aerosols (notably Ni–Co–Mn) and other toxicants, which can deposit downwind⁶. Three days after the fire, we rapidly mobilized to assess whether surface soils at Hester Marsh, a wetland restoration area within the Elkhorn Slough National Estuarine Research Reserve (ESNERR), only a few km from the Moss Landing facility had been affected by the fallout material from the smoke plume. Coincidentally, we had collected baseline surface soil elemental data in the same area for other research purposes with an FpXRF in 2023 (Fig. 2).

¹Moss Landing Marine Laboratories, San Jose State University, 8272 Moss Landing Rd., Moss Landing, CA 95039, USA. ²Elkhorn Slough National Estuarine Research Reserve, 1700 Elkhorn Road, 95076 Royal Oaks, CA, USA. ✉email: iaiello@sjsu.edu

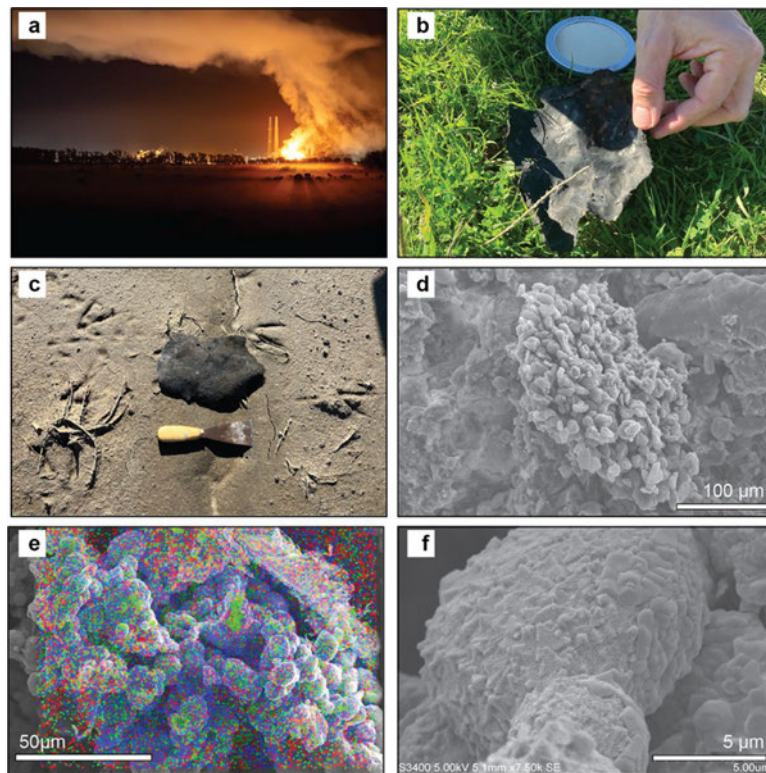


Fig. 1. (a) Photo of the battery fire and the smoke plume on January 16th, 2025. The picture is looking south towards the smokestacks of the old Moss Landing power plant and shows the smoke plume hovering Elkhorn Slough and Hester marsh to the east (Photo credit: Mike Takaki). (b–c) Field photographs showing burned battery fragments from the Vistra battery facility fire collected near transect T12 (B) and transect T8 (C). (d) Scanning electron microscope (SEM) images of cathode material aggregate composed of multiple Nickel Manganese Cobalt (NMC) microparticles; (e) Energy-dispersive X-ray spectroscopy (EDS) elemental map highlighting the spatial distribution of nickel (Ni, red), manganese (Mn, blue), and cobalt (Co, green). (f) A SEM close-up of a single NMC particle.

The Moss Landing battery facility is located within a complex and vulnerable landscape. It sits adjacent to Elkhorn Slough, one of California's largest estuaries, near the town of Moss Landing, and is surrounded by intensively farmed agricultural land. The fallout from the fire's smoke plume raises serious concerns about contamination of soils, water, and vegetation in this region.

Here, we report on the extent and dynamics of cathode metal contamination in estuarine soils immediately following the world's largest lithium-ion battery fire. By combining rapid, high-resolution field surveys with laboratory validation, we tracked the deposition and short-term fate of battery-derived metals in a sensitive wetland ecosystem. Our findings provide rare real-world evidence of the environmental footprint of large-scale battery fires, underscore the value of having a baseline near industrial sites that pose contamination risks, and demonstrate the utility of FpXRF as a practical tool for rapid and spatially intensive environmental monitoring.

Specifically, we test whether the Moss Landing fire deposited a thin surface veneer of battery-associated metals in adjacent wetlands that differ relative to 2023 baseline conditions and whether composition is consistent with NMC cathode material, using a high-density FpXRF survey validated with SEM/EDS, LpXRF, and ICP-MS.

Study area and methodology

Elkhorn Slough is a tide-dominated estuary that in the past 150 years has lost significant vegetated marsh area¹². At Hester Marsh, extensive diking and draining caused the area to subside and degrade to unvegetated mudflat. In 2018, ESNERR initiated a restoration project to reestablish healthy marsh ecosystems through soil addition, creating a high elevation marsh plain that is only inundated by the highest tides.

To assess relationships between marsh plant health and soil composition, soil property analyses including elemental analysis with a portable Hitachi XMET 8000 XRF (pXRF), were conducted in 2023 along ten permanent transects also monitored for vegetation. These compositional data serve as a baseline for elemental concentrations in soils prior to the 2025 battery fire (Table 1). Following the 16 January 2025 fire at the Moss Landing battery storage facility, three of the original transects were resampled at high spatial and temporal resolution between 21 January and 23 February 2025 (Tables S1 and S2).

During the 2023 survey, surface and subsurface (~5–10 mm depth) samples were collected to compare elemental concentrations above and below the shallow redox boundary characteristic of these tidal marsh soils.

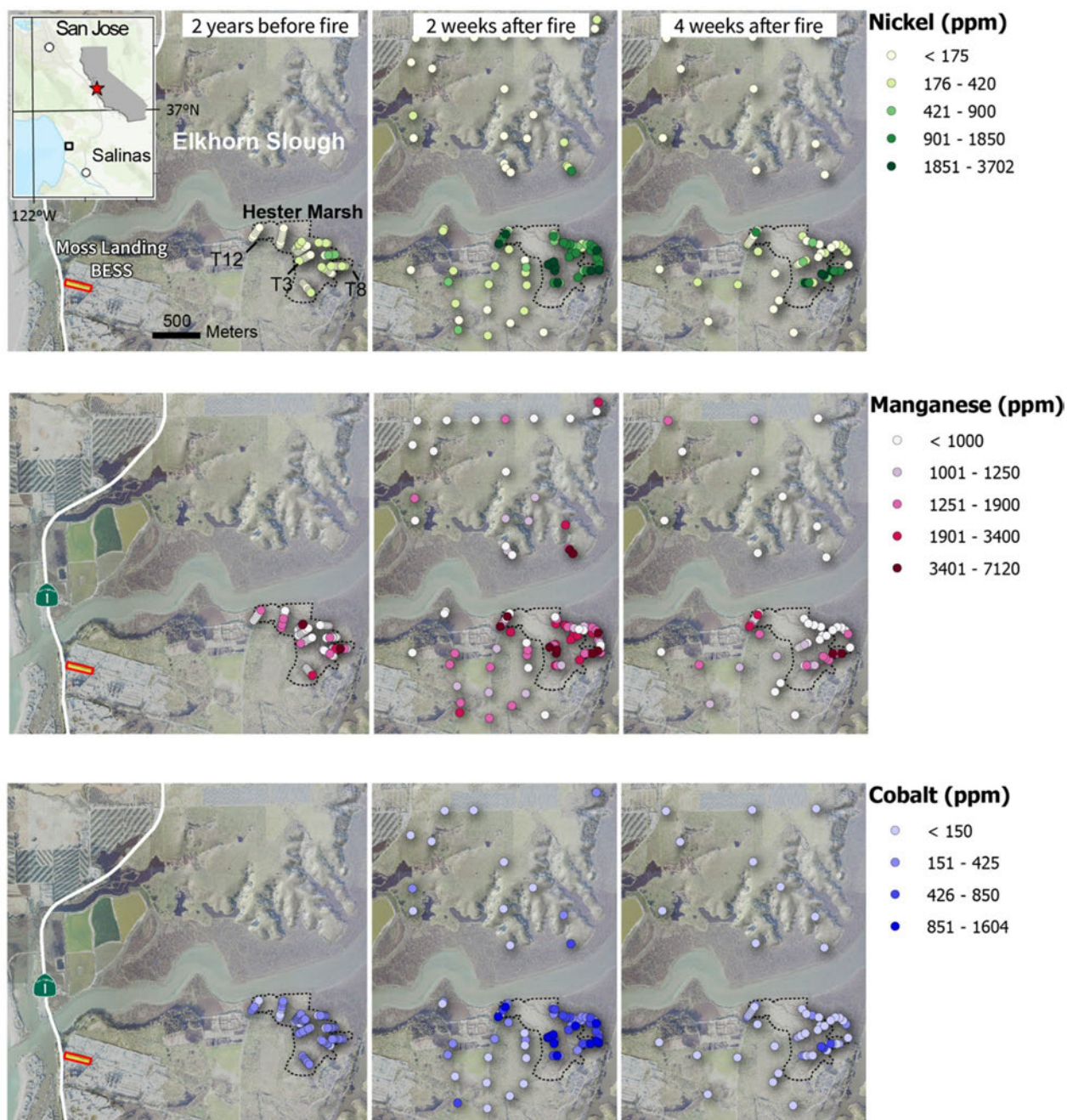


Fig. 2. Spatial distribution of nickel (Ni), cobalt (Co), and manganese (Mn) concentrations (ppm) in soils across three survey periods. The Hester Marsh restoration area is outlined with a dashed line and includes the locations of transects T12, T3, and T8. Peak concentrations were detected within this zone, approximately 1–3 km downwind of the Moss Landing battery facility. Color scales are consistent across all time points for each element to allow temporal comparison. Point classification for each element was done using the “Natural Breaks” (Jenks) method. The map was generated using ArcGIS Pro v3.4.2 (<https://pro.arcgis.com/en/pro-app/la-test/get-started/download-arcgis-pro.htm>).

Additional measurements were taken both outside the transects and beyond the boundaries of Hester Marsh during two post-fire survey periods: post-fire#1 (21 January–12 February 2025) and post-fire#2 (18 February–27 March 2025) (see Supplementary Sect. 1). These post-fire surveys encompassed a broader area, including nearby grasslands within the surrounding watershed (Fig. 2). This approach incidentally enabled differentiation of recent fire-related metal deposition from background levels and allowed detection of a transient, spatially patchy signal. All FpXRF measurements across all surveys have been conducted on bare, relatively dry soils to minimize moisture-related biases.

Survey period	Sites (n)	Area (km ²)	Surface												Subsurface†											
			Ni (ppm)				Mn (ppm)				Co (ppm)				Ni (ppm)				Mn (ppm)				Co (ppm)			
			min	max	median	MAD	min	max	median	MAD	min	max	median	MAD	min	max	median	MAD	min	max	median	MAD	min	max	median	MAD
Pre-fire (2023)	97	0.4	52	246	109	47	228	2193	713	315	72	298	183	59	52	232	116	43	324	2378	834	303	59	214	107	43
Post-fire#1 2025*	135	5.6	52	3702	441	517	386	7119	1488	783	40	1604	309	296	50	303	93	44	264	2119	926	435	40	218	91	49
Post-fire#2 2025^	64	5.6	52	339	100	49	45	3207	871	551	8	833	107	68	1	329	100	49	45	3207	871	551	8	265	97	56

Table . Summary statistics including minimum and maximum values, median, and median absolute deviation (MAD) of nickel (Ni), manganese (Mn), and Cobalt (Co) concentrations (ppm) in surface and subsurface soils measured by field-portable XRF (FpXRF) across three survey periods. Values represent minimum, maximum, and mean concentrations for each metal. †Subsurface measurements are from a few mm below the surface; *Post-fire#1 survey conducted Jan 21–Feb 12; ^Post-fire#2: Feb 18–Mar 27 (surface samples were measured in triplicate during Post-fire#2).

Soil samples collected in the field were analyzed using the pXRF in the lab (LpXRF) and with inductively coupled plasma mass spectrometry (ICP-MS) (Table S3).

Detailed laboratory procedures, including sample preparation, organic carbon analysis, and instrument protocols, environmental data, as well as statistical methods used for data analysis (non-parametric pairwise tests and regression analysis) and interpretation are provided in Supplementary Sect. 2.

Rain and tide data were retrieved from the Moss Landing weather station operated by Moss Landing Marine Laboratories and wind data from the ESNERR meteorological station (Tables S4 and S5).

Results

Detection and mapping of the cathode metals

Fragments of ash and burned or charred material were found scattered across Hester Marsh soils (Fig. 1b, c) in the days to weeks after the fire, providing clear physical evidence of fallout from the battery fire. The comparison between the 2023 and the 2025 post-fire data revealed a marked increase in concentrations of three metals: nickel (Ni), manganese (Mn), and cobalt (Co).

Notably, surface Ni and Co co-varied on log–log axes, with post-fire Ni: Co ratios averaging 2:1, consistent with NMC532 cathode chemistry. This fingerprint supports attribution of the (Ni, Mn, Co) metal spike to battery fire fallout.

Further analysis of selected samples using scanning electron microscopy (SEM) and energy-dispersive X-ray spectrometry (EDS) indicated that the elevated concentrations of Ni, Mn, and Co were linked to the presence of micron-sized metallic particles like those used as cathode materials in Nickel Manganese Cobalt (NMC) batteries. At finer scales, cathode-derived NMC microparticles were identified and elementally mapped in surface soil samples using SEM/EDS (Fig. 1d, e, f), consistent with the fracture and ejection of individual grains from NMC cathodes, a behavior previously observed in laboratory combustion tests⁶. These findings confirm the presence of fire-related battery material on the soil surface of nearby wetlands.

Geochemical evidence from FpXRF further supports the extent and magnitude of contamination. Although Ni, Mn, and Co displayed high spatial variability during the post-fire#1 survey, concentrations increased significantly relative to pre-fire values, with maximum Ni rising by an order of magnitude and Co by a factor of five (Table 1).

Overall, by the time the post-fire#2 survey was conducted, about 1 month after the battery fire, the median concentrations had decreased. The post-fire#1 subsurface data were statistically indistinguishable from the surface and subsurface 2023 pre-fire data. In contrast, surface concentrations of (Ni, Mn, Co) measured during the post-fire#1 survey were significantly elevated compared to pre-fire levels ($p < 0.001$, Mann–Whitney U test; Table S6), clearly indicating that the deposition associated with the fire was initially confined to the top layer of soil.

Figure 2 show that the post-fire#1 increase in metal concentrations (Ni, Mn, Co) in surface measurements was not uniform but clustered in distinct hotspots within Hester Marsh. Hester Marsh was also the area where the post-fire#2 survey recorded the most substantial decrease in metal concentrations. However, a few locations continued to show elevated levels, which explains why the maximum values of Ni and Co in the post-fire#2 survey remained high (Table 1). In contrast to the surface measurements, subsurface data showed no significant changes in either mean or maximum concentrations between the pre- and post-fire surveys. This further confirms that the sharp post-fire increase in cathode metal concentrations was confined to the topmost layer of the soil.

Although the FpXRF measurement along three permanent transects included the concentrations of all three cathode elements: Ni, Mn, and Co, we focused primarily on Ni as a tracer of battery fire fallout, as Ni is dominated by a single oxidation state (Ni^{2+}) across a broad range of redox and pH conditions. This makes it less sensitive to post-depositional remobilization compared to Mn and Co, both of which exhibit variable redox behavior in estuarine settings¹³.

Mn is strongly influenced by fluctuations in redox potential and organic matter, and its concentrations often vary independently of anthropogenic inputs¹⁴. Co also exhibited substantial redox sensitivity and, notably, a large proportion of Co measurements were non-detects, especially in pre-fire and subsurface samples (Table S7), due to concentrations below the portable XRF instrument's relatively high detection limit (Ni $\approx$ 50 ppm, Mn $\approx$ 45 ppm, Co $\approx$ 40 ppm). Box plots depicting the temporal trends of surface Ni concentrations at three permanent transects show that, following the fire, the median Ni concentration increased by two to threefold compared to pre-fire levels (Fig. 3). Over the month-long survey period, both the median and interquartile range of concentrations declined, with a substantial drop to near pre-fire values observed in early February coincident with rainfall in the area. Notably, Ni concentrations rose again at all transects during the surveys conducted in the second half of February.

On log–log axes (Fig. 4a), surface Ni and Co show clear bivariate associations. The distribution of $\log_{10}(\text{Ni}/\text{Co})$ (Fig. 4b) shows that post-fire values average near the 2:1 reference (0.301), while pre-fire values average < 0 indicating a substantial change in surface soil metal composition following the fire.

Comparative elemental analysis: field versus lab

To compare FpXRF results with laboratory measurements, we collected 51 samples (24 subsurface and 27 surface) from soils that had previously been analyzed in the field with FpXRF after the battery fire. Aliquots of these samples were analyzed for moisture content, organic carbon content, and elemental composition using LpXRF (5–6 g) and ICP-MS (~ 0.25 g). The water content of the samples ranged from approximately 7 to 43% by weight, while organic carbon concentrations varied between 0.1% and 2.4% by weight.

Because regressions of XRF against the ICPMS reference had non-zero intercepts and modest R^2 , we quantified method bias as the geometric mean of the per-sample ratios (i.e., FpXRF/ICPMS, LpXRF/ICPMS, and for completeness FpXRF/LpXRF, see Table S8). In subsurface samples, both XRF methods overestimated Ni

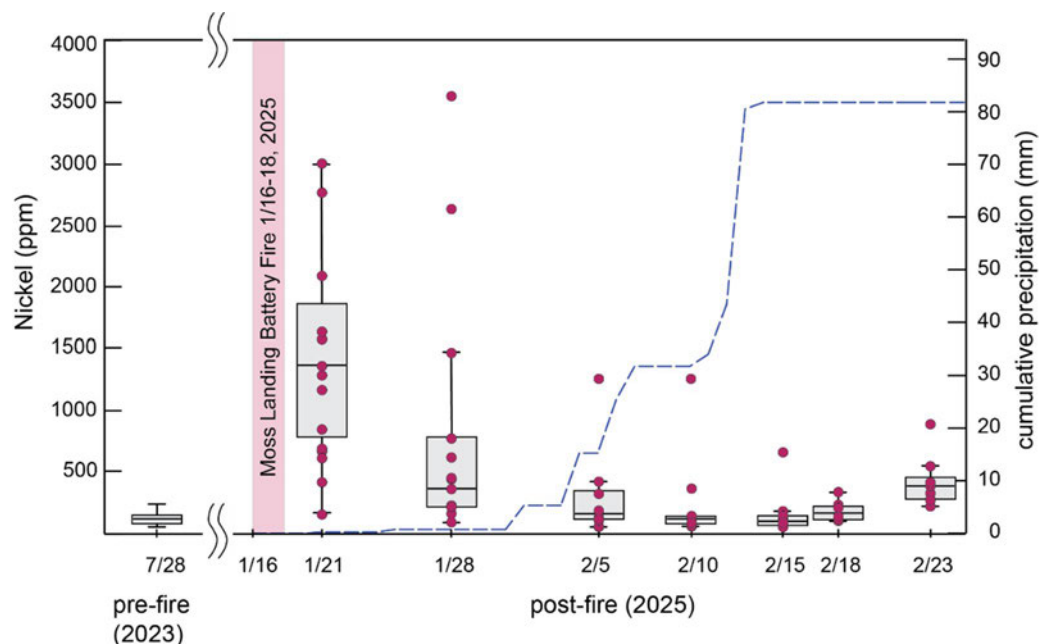


Fig. 3. Temporal variability in surface nickel (Ni) concentrations (ppm) along three permanent transects measured before the battery fire in 2023, and between January and February 2025 using field-portable X-ray fluorescence (FpXRF). Box plots represent the distribution of Ni concentrations at the benchmarks along each transect measured during each sampling date. The datapoints are represented with red circles (the pre-fire survey data were not included because they were indistinguishable and overlapping given their low values). The dashed line shows cumulative precipitation data recorded at the Moss Landing Marine Laboratories weather station (Latitude: 36.80040° N, Longitude: 121.78842° W). A transient spike in Ni concentrations occurred immediately after the 16 January 2025 battery fire, followed by a rapid decline, likely associated with rainfall and tidal flushing in early February. A smaller secondary increase was observed in late February, coinciding with the 18 February 2025 reignition event.

relative to ICPMS by roughly threefold (FpXRF/ICPMS = 3.09, 95% CI 2.74–3.48; LpXRF/ICPMS = 2.81, 95% CI 2.61–3.02), while the two types of XRF measurements were in reasonable agreement (FpXRF/LpXRF = 1.10, 95% CI 0.97–1.25). In surface samples, both FpXRF and LpXRF exhibited a larger positive bias (FpXRF/ICPMS = 5.40, 95% CI 4.18–6.99; LpXRF/ICPMS = 2.33, 95% CI 1.93–2.81). As we describe below, the stronger disagreement between FpXRF and laboratory measurements at the surface is best explained by dilution of a thin, metal-rich veneer during laboratory homogenization (which mixes surface material with underlying soil), whereas in-situ FpXRF interrogates the veneer more directly.

As observed with FpXRF data, LpXRF measurements of subsurface samples showed no significant linear association between Co and Ni. In contrast, surface samples showed coherent Ni–Co covariation across methods; Ni: Co ratios were near 2:1, consistent with Fig. 4b and the Ni-to-Co ratio observed in the post-fire#1 FpXRF survey data (Table S9).

Depth distribution of cathode metals

The NMC microparticles primarily occurred as aggregates of varying shape and size, often ~100 µm or larger (Fig. 1d, f). This suggests a minimum thickness for the deposition layer of approximately 100 µm, comparable to the critical detection depths for Ni, Mn, and Co in XRF analysis, defined as the depth beyond which less than 1% of the original fluorescent signal reaches the detector.

A rough estimate of the thickness of this contaminated layer can be derived by comparing FpXRF measurements with LpXRF results from sliced surface samples for which lab-based measurements using both LpXRF and ICP-MS yielded lower concentrations of Ni, Mn, and Co compared to those obtained via FpXRF.

Our hypothesis was that FpXRF and LpXRF should approximately yield similar concentrations (i.e., FpXRF/LpXRF ≈ 1) only when the thickness of the lab-analyzed sample approaches the depth of the metal-enriched layer. If the sample is thicker, it will include subsoil not affected by the fire, diluting the signal and resulting in FpXRF/LpXRF > 1.

A regression analysis of sample thickness versus the FpXRF/LpXRF ratio revealed a moderate positive linear association ($R^2 = 0.30$, $p = 0.0129$) that should be interpreted with caution. However, the ratio approaches one for samples between ~2 and 5 mm thick, which we interpret as the approximate depth to which fire-related cathode metals were initially incorporated into the soil (Figure S3).

This finding is consistent with our surface method comparison against the ICPMS reference: as the effective field sampling depth increases, the measured signal is increasingly diluted by pre-fire baseline material, leading

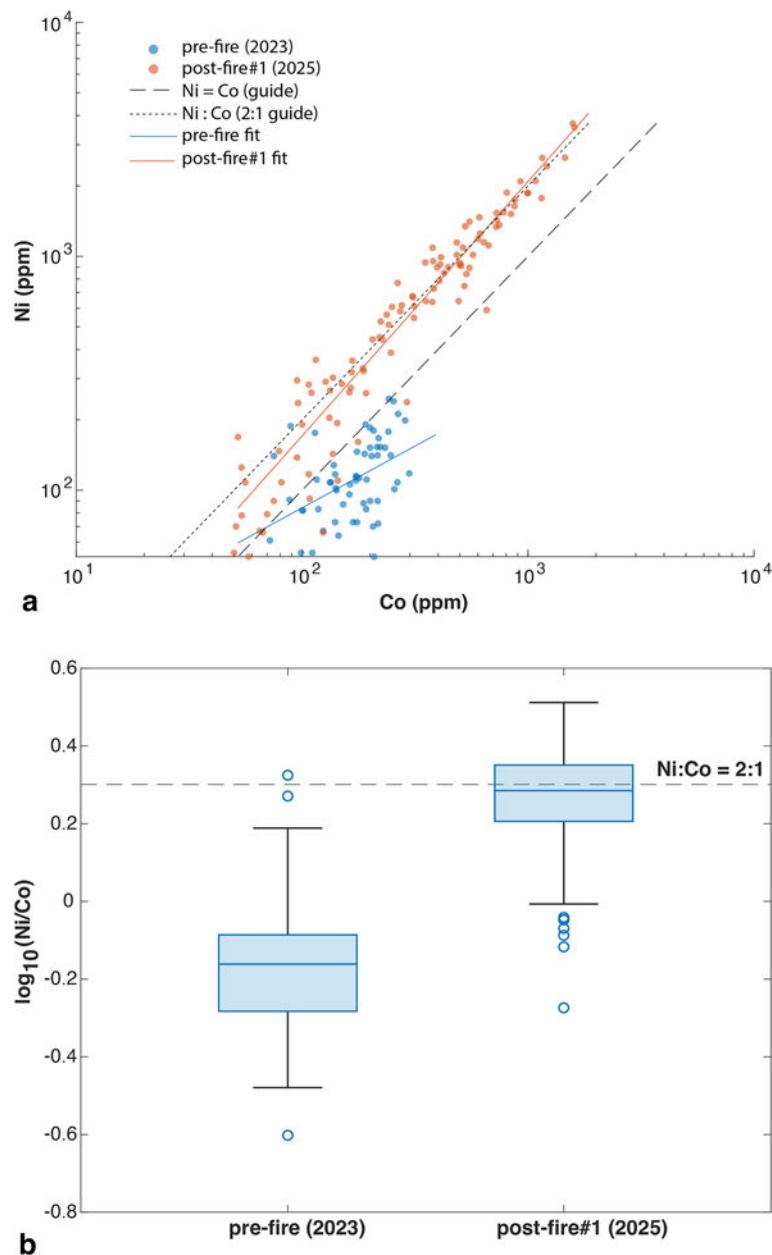


Fig. 4. (a) Log-log scatter of Co versus Ni (ppm; FpXRF) for pre-fire (2023, blue) and post-fire#1 (Jan–Feb 2025, orange). Guidelines show Ni = Co (1:1) and Ni: Co = 2:1. Robust log-log fits: pre $b = 0.45$ (95% CI 0.25–0.65); post $b = 0.85$ (0.80–0.89). (b) Distributions of $\log_{10}(\text{Ni}/\text{Co})$ for the same samples; dashed line at 0.301 marks Ni: Co = 2:1. Post-fire medians lie near 2:1, whereas pre-fire values are < 0 .

to underestimation of battery fire-related surface contamination (see Supplementary Sect. 2 for a detailed discussion of pXRF detection depth and matrix effects).

Discussion

Rapid detection of cathode metals with FpXRF

The sharp increase in (Ni, Mn, Co) metal concentrations detected in the surface soils of Hester Marsh between late January and early February 2025 is clearly attributable to the deposition of particulate matter from the smoke plume generated by the nearby battery storage facility fire at Moss Landing just days earlier. This interpretation is supported by multiple lines of evidence, including visible ash residues and soot, the presence of cathode-derived microparticles in surface soils, and distinctive geochemical patterns.

The key to early detection of cathode metal fallout immediately after the Moss Landing battery fire was the use of FpXRF. While field measurements were not as accurate as lab measurements, they played a pivotal role in rapidly observing that maximum concentrations of the three metals increased by an order of magnitude after the fire, monitoring how quickly they decreased, and assessing how patchy the battery metal fallout was across

the landscape. This key information could have been completely missed if we had relied only on a handful of samples taken in space and time. Metal co-variation patterns were consistent across methods (FpXRF, LpXRF, ICP-MS), while absolute levels differed.

While the use of FpXRF offers substantial advantages in responding quickly to environmental emergencies like battery fires, it also comes with limitations. These are especially pronounced in wetland soils, where moisture content, organic matter, and textural variability can significantly influence the accuracy of XRF readings. Light elements such as Ni, Mn, and Co are particularly susceptible to overestimation when measured with XRF, compared to more precise ICP-based methods^{15–18}. Our comparison of field and laboratory measurements confirms this pattern: both FpXRF and LpXRF overestimated Ni concentrations by more threefold relative to ICP-MS in the subsurface where samples do not include the fallout deposit.

Nevertheless, the methods showed an acceptable level of reproducibility, supporting the reliability of XRF for rapid environmental assessment. Importantly, although absolute concentrations may be overestimated, the change in surface concentrations before and after the fire was evaluated using the same FpXRF method, allowing for robust spatial comparison and comparison of relative differences over time.

A fingerprint for the cathode material

On log–log axes, Ni scales with Co in surface soils (Fig. 4a). Using ordinary least squares (OLS), the pre-fire fit yields $b = 0.4750b$ and $a = 0.9913$ ($R^2 = 0.175$), indicating a heterogeneous pre-fire ambient signal. The post-fire (survey 1) yields $b = 1.1023$ and $a = 0.0061$ ($R^2 = 0.912$); the point cloud and fitted line lie close to the $Ni = 2 \cdot Co$ guideline across the observed range, motivating a ratio view.

Figure 4b shows that the distribution of $\log_{10}(Ni/Co)$ shifts from pre-fire values < 0 ($Ni:Co < 1$) to post-fire values near the 2:1 reference ($\log_{10} 2 = 0.301$). This composition is consistent with the NMC532 cathode chemistry used in lithium-ion batteries⁹.

Subsurface samples remained near pre-fire levels and did not exhibit the post-fire ratio shift, indicating enrichment confined to a surface veneer. Notably, maximum surface concentrations of all three metals increased by several fold relative to pre-fire levels (Table 1). Most post-fire#1 surface samples analyzed with ICP-MS had Ni concentrations above 50 ppm, values that exceed thresholds associated with toxicity risks to plants and aquatic organisms²⁰.

The observed changes in surface concentrations of Ni, Mn, and Co across Hester Marsh and surrounding areas over time indicate the potential for rapid remobilization of these transition metals into estuarine soils and downstream waters (Fig. 2). FpXRF transect data from February–March 2025 show that surface Ni concentrations dropped to near baseline within weeks of the fire, following early February rain and tidal inundation (Fig. 3).

While variability in Ni concentrations declined over time, indicating redistribution of the cathode metals, at the higher elevation transect T12 (~1 km from the fire), higher Ni levels persisted for about 10 days before declining, while the lower elevation transects T3 and T8 showed an earlier decrease, likely due to January tidal flooding that immersed only the areas with the lowest relief.

The transect data and a simple comparison between columns 2 and 3 in Fig. 2 illustrate that had FpXRF sampling been delayed by even a few days, most of the early evidence for surface deposition in Elkhorn Slough would likely have been lost. Timely field deployment was essential for capturing the initial contamination signal before environmental processes such as rainfall and tidal flushing remobilized the metals.

Our results emphasize the high degree of spatial variability in the distribution of battery-associated metals on the soil surface. The highest concentrations observed after the fire were within the unvegetated portions at Hester Marsh (Fig. 2). Concentrations were also highly variable at finer spatial scales, between samples collected 10–20 m apart along the transects.

Boxplots of Ni concentrations over time along three transects (Fig. 3) show that the interquartile range, a measure of variability excluding outliers, increases with the median concentration. This relationship suggests that spatial heterogeneity is greatest where concentrations are highest, a pattern we interpret as evidence of the clumped distribution of cathode metal-bearing particles or ash. At the microscale, this clumping is represented by aggregates of NMC microparticles (Fig. 1d–f); at the macroscale, it is reflected in the scattering of ash and burned material fragments observed throughout the study area, up to approximately 3 km from the battery storage facility (Fig. 1b, c). Larger clumps result in higher localized concentrations and contribute to measurement variability, reinforcing the importance of repeated sampling at multiple spatial scales. This multi-scale capability is one of the key advantages of FpXRF over conventional discrete sampling and ensuing analysis via ICP-MS.

The relationship between the FpXRF/LpXRF concentration ratio and sample thickness used to estimate the critical depth of Ni enrichment following the battery fire shows that the ratio approaches 1.0 when sample thicknesses range between ~2 and 5 mm, suggesting that most of the deposited Ni was confined to the uppermost few millimeters of soil. Thicker samples diluted this surface signal, consistent with a sharp depositional pulse and limited vertical mixing. This interpretation aligns with both SEM imaging of NMC microparticle aggregates and the shallow critical escape depth of Ni in soil matrices.

Tracking the environmental footprint of cathode material

The cathode material detected in Elkhorn Slough soils following the battery fire was, at least initially, airborne. As a first approximation, the spatial distribution of cathode metals observed in surface soils during the post-fire#1 survey reflects the deposition pattern of battery-derived particulates that settled from the smoke plume. However, this interpretation likely oversimplifies the dynamics. Prior studies show that ground-level deposition often diverges from the plume's direction due to complex atmospheric behavior, as additional factors might be influencing spatial variability including plume height, particle size and shape, and sorptive properties of the soil surface^{21,22}.

Our reconstruction of the (Ni, Mn, Co) metal distribution from the post-fire#1 survey reveals that surface concentrations of contaminants do not increase with proximity to the battery fire site. Instead, concentrations peak in the Hester Marsh wetland, located approximately 1–3 km east of the facility (Fig. 2). This offset may be explained by westerly surface winds on the day of the fire, which prevailed for approximately 40% of the time, likely directing the smoke plume eastward over Hester Marsh (Fig. 1a). In addition to wind direction, local soil characteristics may have played a role in the retention of the metal particles following deposition. Hester Marsh soils are predominantly fine-grained and clay-rich, with high sorptive capacity, and higher potential for retention of transition metals that can increase their persistence in surface layers^{23,24}.

Regardless of the factors that controlled the initial distribution and retention of material deposited from the battery fire, the spike in transition metal concentrations at the surface of the wetland was short-lived. By the time the post-fire#2 survey was conducted, most of the cathode metals accumulated on the surface soils had been remobilized, except for a few persistent hotspots in some of the depressional salt pans within Hester Marsh (Fig. 2).

Natural processes such as rainfall and tidal inundation likely played a major role in the resuspension and remobilization of the contaminants. Precipitation was relatively intense during the first 2 weeks of February 2025 (Fig. 3), and the lower portions of Hester Marsh experienced repeated inundation during high-tide events, facilitating the physical transport and dispersal of deposited cathode metals. Percolation through the soils appears to have been limited since the post-fire#2 and post-fire#1 subsurface concentrations were statistically indistinguishable.

The drop in surface concentration of cathode metals resulting from the battery fire strongly suggests that the metals have been washed into downstream portions of the estuarine ecosystem. Their transport and fate throughout the estuary and potentially into adjacent open coastal ecosystems remain unknown. They may have settled in tidal channels, become buried in sediments, or undergone chemical transformations driven by redox cycling²⁵.

These processes could affect both metal mobility and, over time, pose a threat to higher trophic levels through bioaccumulation. Ni, Mn, and Co are all known to be toxic to humans as well as to aquatic and terrestrial organisms²⁰, and Mn toxicity is a major constraint limiting plant growth and production¹⁶. Co can have lethal or sublethal effects on reproduction in fish and crustaceans and has some bioaccumulation potential through adsorption to plant roots²⁶. These risks are particularly acute at Hester Marsh, where an \$18 M investment to restore tidal wetlands through soil augmentation raised marsh platforms to elevations intended to sustain native plant growth and survival under future flooding²⁷.

Mass budget for cathode metal deposition at Hester Marsh

The initial mass of Ni, Mn, and Co originating from the burned batteries that settled on Hester Marsh can be estimated based on the difference between metal concentrations measured at the surface during the pre-fire and the post-fire#1 surveys (see Table S10 for full calculations). To calculate the mass budget, the Hester Marsh area (1200,000 m²) was overlain with a 200 × 200 m grid composed of 30 cells, each covering a 40,000 m² area. For 10 of the 30 cells that contained both pre-fire and post-fire#1 FpXRF measurements, we computed the paired mean difference (post-fire#1—pre-fire) and the standard errors. The concentrations of the three metals were then converted to mass assuming a surface contamination depth of approximately 0.1 mm (which approximates the critical detection depth for these metals) and a dry density of 1500 kg/m³ for consolidated clay.

We estimate that the minimum deposited mass of cathode metals within the upper 100 µm is ~ 17 kg ± 4 kg per 200 × 200 m cell. However, as discussed earlier, the actual thickness of the soil layer containing the deposited metals is likely greater than the effective depth, and can be ~ 5 mm thick, which corresponds to a mass of cathode metals of ~ 855 kg ± 199 kg per cell. Extrapolation to the entire Hester Marsh area yields an estimated total mass of cathode metals of ~ 25 metric tons (25676 kg ± 5981 kg).

This estimate should be interpreted with caution. First, FpXRF measurements overestimate Ni concentrations by a factor of ~ 3, and the metal concentrations are expected to decline with depth due to dilution and limited vertical mixing. However, the values are also conservative, as they only include deposition on bare soil and exclude potential accumulation on vegetation, which may represent a larger surface area in marsh and upland settings than the exposed soil itself.

To put these figures in perspective, a 1 MW industrial lithium-ion battery manufactured by LG weighs 1.6 metric tons, with cathode materials accounting for approximately 35% of the total mass. This equates to roughly 1900 metric tons for the entire 1200 MWh storage capacity of the Moss Landing facility. If, as reported by Monterey County officials¹⁰, approximately 75% of the batteries were destroyed in the fire, then an estimated ~ 1400 metric tons of cathode material could have been involved in the event and potentially entrained into the smoke plume. Therefore, our estimates of the total mass of (Ni, Mn, Co) metals deposited on the soils of the Hester Marsh extrapolated to a 5 mm cathode metal deposit accounts only for < ~ 2% of the total battery material that may have been released during the Moss Landing battery fire.

Conclusions and implications for future battery fire response

To our knowledge, this study represents the first field-based documentation of battery-associated metal fallout following a large-scale lithium-ion battery fire and offers a framework for assessing future events of this kind. Use of field instrumentation enabled immediate collection of hundreds of measurements, critical given the spatial patchiness of battery metal aggregates in an extensive fallout layer in the vicinity of the fire and given the rapidity with which the metals were transported downstream by tides and rain. As battery energy storage systems continue to expand in scale and density, the risk of both localized and widespread contamination will increase even as safety protocols improve.

This incident also calls attention to the limitations of standard environmental sampling protocols. Conventional soil sampling depths, such as the commonly used top ~ 6 cm of soil²⁸, may fail to detect thin, spatially heterogeneous deposition layers. The patchy nature of ash deposition observed in this study suggests that sampling strategies must be adaptive and designed to capture contamination at multiple spatial scales and depths. This is especially critical in the first few days following an event, since, over time, rainfall, tides, and wind can rapidly redistribute surface-bound contaminants.

Environmental response frameworks must also consider the potential offset between fire origin and deposition zones. In this case, the most significant contamination occurred not adjacent to the site of the fire, but several kilometers downwind. This spatial offset highlights the need for evacuation protocols and monitoring networks that integrate plume dispersion models, meteorological data, air quality monitoring and ground-based measurements of deposition.

Finally, findings from controlled laboratory battery burns provide additional context for interpreting field observations. Previous experiments have demonstrated that thermal decomposition of cathode materials can release substantial quantities of (Ni, Mn, Co) metals and other toxicants⁶. These studies confirm that NMC-based batteries, when subjected to fire conditions, can emit airborne particles capable of traveling significant distances before settling onto the landscape. Field studies such as this one are essential to understanding how such deposition events unfold under real-world conditions.

Together, these results emphasize the need for proactive planning, site-specific risk assessment, and rapid, multi-scale environmental monitoring in the aftermath of battery fires. As battery technologies evolve, so too must the frameworks we use to track and mitigate their potential environmental impacts.

Data availability

All data used to generate the figures are available through Figshare at <https://figshare.com/s/32fb2899e519353f923b> **, **.

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References

- Koohi-Fayegh, S. & Rosen, M. A. A review of energy storage types, applications and recent developments. *J. Energy Storage* **27**, 101047. <https://doi.org/10.1016/j.est.2019.101047> (2020).
- Gutsch, M. & Leker, J. Global warming potential of lithium-ion battery energy storage systems: A review. *J. Energy Storage* **52**, 105030. <https://doi.org/10.1016/j.est.2022.105030> (2022).
- Gür, T. M. Giga-ton and tera-watt scale challenges at the energy-climate crossroads: A global perspective. *Energy* **290**, 129971. <https://doi.org/10.1016/j.energy.2023.129971> (2024).
- Armand, M. et al. Lithium-ion batteries—current state of the Art and anticipated developments. *J. Power Sources* **479**, 228708. <https://doi.org/10.1016/j.jpowsour.2020.228708> (2020).
- Xie, J., Li, J., Wang, J., Jiang, J. & Shu, C. M. Fire risk assessment in lithium-ion battery warehouse based on the bayesian network. *Process. Saf. Environ. Prot.* **176**, 101–114. <https://doi.org/10.1016/j.psep.2023.06.005> (2023).
- Barone, T. L. et al. Lithium-ion battery explosion aerosols: Morphology and elemental composition. *Aerosol Sci. Technol.* **55**, 1183–1201. <https://doi.org/10.1080/02786826.2021.1938966> (2021).
- Mrozik, W., Rajaeifar, M. A., Heidrich, O. & Christensen, P. Environmental impacts, pollution sources and pathways of spent lithium-ion batteries. *Energy Environ. Sci.* **14**, 6099–6121. <https://doi.org/10.1039/D1EE00691F> (2021).
- Lopez, R. D. *Ramsar Wetlands of the North American West Coast and Central Pacific: An Atlas* (CRC, 2024). <https://doi.org/10.1201/9781003046394>
- Vistra Corp. Vistra completes expansion of battery energy storage system at its flagship California facility. <https://investor.vistracorp.com/2021-08-19-Vistra-Completes-Expansion-of-Battery-Energy-Storage-System-at-its-Flagship-California-Facility> (Accessed 29 September 2025).
- San Francisco Chronicle. Fire at California battery plant brings safety assurances, quest for answers. San Francisco Chronicle (18 Jan 2025). <https://www.sfgate.com/california/article/moss-landing-fire-20042587.php> (Accessed 29 September 2025).
- County of Monterey. Moss Landing Power Plant Fire. <https://www.readymontereycounty.org/emergency/2025-moss-landing-vistra-power-plant-fire> (Accessed 29 September 2025).
- Van Dyke, E. & Wasson, K. Historical ecology of a central California estuary: 150 years of habitat change. *Estuaries* **28**, 173–189. <https://doi.org/10.1007/BF02732853> (2005).
- Yang, Z., Guo, W., Fan, Y., Lin, C. & He, M. High-resolution profiles of iron, manganese, cobalt, cadmium, copper and zinc in the pore water of estuarine sediment. *Int. J. Environ. Sci. Technol.* **10**, 275–282. <https://doi.org/10.1007/s13762-012-0110-2> (2013).
- Bartlett, R. J. Manganese redox reactions and organic interactions in soils. In *Manganese in Soils and Plants* (eds Graham, R. D., Hannam, R. J. & Uren, N. C.) 59–73 (Springer, Dordrecht, 1988). https://doi.org/10.1007/978-94-009-2817-6_5
- Sikora, A. L., Maguire, L. W., Nairn, R. W. & Knox, R. C. A comparison of XRF and ICP-OES methods for soil trace metal analyses in a mining-impacted agricultural watershed. *Environ. Monit. Assess.* **193**, 92. <https://doi.org/10.1007/s10661-021-09275-9> (2021).
- Li, W. et al. Comparative study on the determination of heavy metals in soil by XRF and ICP-MS. *J. Phys. : Conf. Ser.* **2009** (012075). <https://doi.org/10.1088/1742-6596/2009/1/012075> (2021).
- Ge, L., Lai, W. & Lin, Y. Influence of and correction for moisture in rocks, soils and sediments on in situ XRF analysis. *X-Ray Spectrom.* **34**, 28–34. <https://doi.org/10.1002/xrs.782> (2005).
- Wilson, P., Cooke, M., Cawley, J., Giles, L. & West, M. Comparison of the determination of copper, nickel and zinc in contaminated soils by X-ray fluorescence spectrometry and inductively coupled plasma spectrometry. *X-Ray Spectrom.* **24**, 103–108. <https://doi.org/10.1002/xrs.1300240305> (1995).
- ECHA. Lithium nickel cobalt manganese oxide. European Chemicals Agency. <https://echa.europa.eu/> (Accessed 29 September 2025).
- Eisler, R. Nickel hazards to fish, wildlife, and invertebrates: A synoptic review. US Geological Survey, Patuxent Wildlife Research Center. <https://restservice.epri.com/publicdownload/000000003002030360/0/Product> (Accessed 29 September 2025).
- Lejon, C. et al. Lagrangian plume rise and dispersion modelling of the large-scale lithium-ion battery fire in Morris, USA, Air Qual. Atmos. Health **17**, 2077–2089 (2024). <https://doi.org/10.1007/s11869-023-01443-9> (2021).
- Goodrick, S. L., Achtemeier, G. L., Larkin, N. K., Liu, Y. Q. & Strand, T. M. Modelling smoke transport from wildland fires: A review. *Int. J. Wildland Fire* **22**, 83–94. <https://doi.org/10.1071/WF11116> (2013).

23. Shand, C. A. & Wendler, R. Portable X-ray fluorescence analysis of mineral and organic soils and the influence of organic matter. *J. Geochem. Explor.* **143**, 31–42. <https://doi.org/10.1016/j.gexplo.2014.03.005> (2014).
24. Kravchenko, E. et al. Ecological and health risk assessments of heavy metal contamination in soils surrounding a coal power plant. *J. Hazard. Mater.* **484**, 136751. <https://doi.org/10.1016/j.jhazmat.2024.136751> (2025).
25. Barrio-Parra, F. et al. Environmental risk assessment of Cobalt and manganese from industrial sources in an estuarine system. *Environ. Geochem. Health* **40**, 737–748. <https://doi.org/10.1007/s10653-017-0020-9> (2018).
26. ATSDR. Toxicological Profile for Cobalt. <https://www.atsdr.cdc.gov/toxprofiles/tp33.pdf> (Accessed 29 September 2025).
27. Haskins, J. et al. UAV to inform restoration: A case study from a California tidal marsh. *Front. Environ. Sci.* **9**, 642906. <https://doi.org/10.3389/fenvs.2021.642906> (2021).
28. US EPA. Method 6200: Field portable X-ray fluorescence spectrometry for the determination of elemental concentrations in soil and sediment. <https://www.epa.gov/sites/default/files/2015-06/documents/6200.pdf> (Accessed 29 September 2025).

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Author contributions

I.W.A. conceived the study, led the writing and contributed to all aspects of this work; CE performed field work, data collection and geospatial interpretations; SC performed lab analyses; MF contributed to study design, site access and data collection; MG contributed to data interpretation; WH performed lab analysis and data interpretation; AK contributed to data interpretation; KW contributed to study design, site access and data interpretation. All authors contributed to manuscript revisions and approved the final version.

Declarations

Competing interests

The authors declare no competing interests.

Additional information

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WHO global air quality guidelines

Particulate matter (PM_{2.5} and PM₁₀),
ozone, nitrogen dioxide, sulfur dioxide
and carbon monoxide



World Health
Organization

071

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Foreword

Clean air is fundamental to health. Compared to 15 years ago, when the previous edition of these guidelines was published, there is now a much stronger body of evidence to show how air pollution affects different aspects of health at even lower concentrations than previously understood. But here's what hasn't changed: every year, exposure to air pollution is still estimated to cause millions of deaths and the loss of healthy years of life. The burden of disease attributable to air pollution is now estimated to be on a par with other major global health risks such as unhealthy diets and tobacco smoking.

In 2015, the World Health Assembly adopted a landmark resolution on air quality and health, recognizing air pollution as a risk factor for noncommunicable diseases such as ischaemic heart disease, stroke, chronic obstructive pulmonary disease, asthma and cancer, and the economic toll they take. The global nature of the challenge calls for an enhanced global response.

These guidelines, taking into account the latest body of evidence on the health impacts of different air pollutants, are a key step in that global response. The next step is for policy-makers around the world to use these guidelines to inform evidence-based legislation and policies to improve air quality and reduce the unacceptable health burden that results from air pollution.

We are immensely grateful to all the scientists, colleagues and partners around the world who have contributed time and resources to the development of these guidelines. As with all WHO guidelines, a global group of experts has derived the new recommendations based on a robust and comprehensive review of the scientific literature, while adhering to a rigorously defined methodology. This process was overseen by a steering group hosted and coordinated by the WHO European Centre for Environment and Health.

Although the burden of air pollution is heterogeneous, its impact is ubiquitous. These guidelines come at a time of unprecedented challenges, in the face of the ongoing COVID-19 pandemic and the existential threat of climate change. Addressing air pollution will contribute to, and benefit from, the global fight against climate change, and must be a key part of the global recovery, as prescribed by the WHO Manifesto for a healthy recovery from COVID-19.

A guideline is just a tool. What matters is that countries and partners use it to improve air quality and health globally. The health sector must play a key role in monitoring health risks from air pollution, synthesizing the evidence, providing the tools and resources to support decision-making, and raising awareness of the impacts of air pollution on health and the available policy options. But this is not a job for one sector alone; it will take sustained political commitment and bold action and cooperation from many sectors and stakeholders. The payoff is cleaner air and better health for generations to come.


WHO Director-General


WHO Regional Director for Europe

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WHO thanks all members of the steering group, guideline development group, systematic review team and external review group for their invaluable contributions in the guideline development process.

WHO, through its European Centre for Environment and Health, coordinated the development of these guidelines. The work was coordinated by [REDACTED], under the overall supervision of [REDACTED], Head of the European Centre for Environment and Health and [REDACTED], Director of the Division of Country Health Programmes, WHO Regional Office for Europe. [REDACTED], Director of the Department of Environment, Climate Change and Health, WHO headquarters provided invaluable advice and support throughout the process.

Members of the WHO steering group were [REDACTED]

[REDACTED] (see Annex 1, Table A1.1 for membership periods and affiliations).

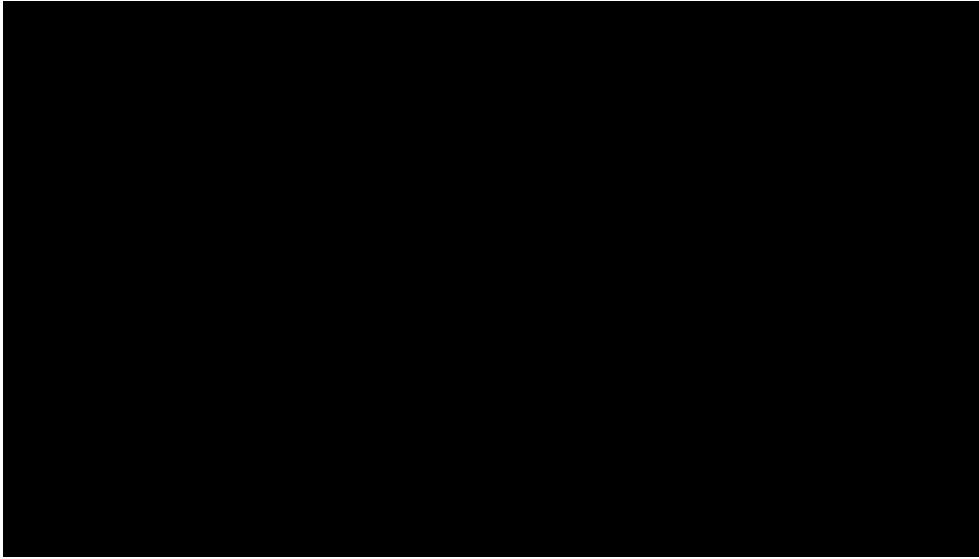
Members of the guideline development group were [REDACTED]

[REDACTED] (see Annex 1, Table A1.2 for membership periods and affiliations).

The systematic review team consisted of the following experts: [REDACTED],

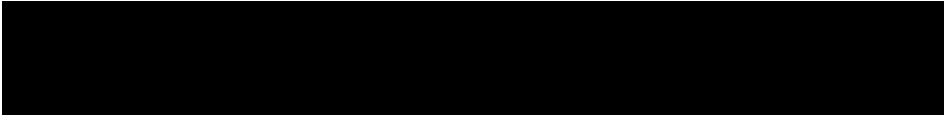
and [REDACTED] (see Annex 1, Table A1.3 for affiliations).

The external review group was composed of the following individual members, who participated at various stages of the guideline development process:



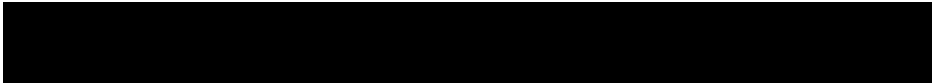
The following stakeholder organizations were also part of the external review group and, in particular, provided comments on the draft guideline document: Abu Dhabi Global Environmental Data Initiative, African Centre for Clean Air, Association for Emissions Control by Catalyst, Clean Air Asia, ClientEarth, Concawe, European Environment Agency, European Environmental Bureau, European Federation of Allergy and Airways Diseases Patients' Associations, European Respiratory Society, Health and Environment Alliance, International Society for Environmental Epidemiology, International Transport Forum and South Asia Co-operative Environment Programme (see Annex 1, [Table A1.6](#) for details). Representatives from the European Commission observed the meetings of the guideline development group (Frauke Hoss in 2016 and Thomas Henrichs in 2018–2020).

Special thanks are extended to the guideline development group co-chairs, the experts who contributed to several methodological working groups:



(see Annex 1, [Table A1.7](#) for details), and the external methodologists
(see Annex 1, [Table A1.4](#) for details).

In addition, WHO would like to express special gratitude to the members of the guideline development group who largely drafted the guideline document:



Glossary

Abatement. The reduction or elimination of pollution, which involves either legislative measures or technological procedures, or both.

Accountability research. Assessment of the effectiveness of interventions. Knowledge gained from such assessments can provide valuable feedback for improving regulatory or other action.

AirQ+. A software tool for health risk assessment of air pollution that looks at the effects of short-term changes in air pollution (based on risk estimates from time-series studies) and of long-term exposures (using the life-tables approach and based on risk estimates from cohort studies).

Air quality guidelines. A series of WHO publications that provide evidence-informed, non-binding recommendations for protecting public health from the adverse effects of air pollutants by eliminating or reducing exposure to hazardous air pollutants and by guiding national and local authorities in their risk management decisions. The current volume is the latest issue of the series.

Air quality guideline level. A particular form of a guideline recommendation consisting of a numerical value expressed as a concentration of a pollutant in the air and linked to an averaging time. It is assumed that adverse health effects do not occur or are minimal below this concentration level. For the purposes of this document, a long-term air quality guideline level is defined as the lowest exposure level of an air pollutant above which the guideline development group is confident that there is an increase in adverse health effects; the short-term air quality guideline level is defined as a high percentile of the distribution of daily values, for example the 99th percentiles equivalent to three to four days a year exceeding this value.

Air quality standard. A given level of an air pollutant (for example, a concentration or deposition level) that is adopted by a regulatory authority as enforceable. Unlike an air quality guideline level, a number of elements in addition to the effect-based level and averaging time must be specified in the formulation of an air quality standard. These elements include:

- measurement technique and strategy
- data handling procedures (including quality assurance/quality control)
- statistics used to derive, from the measurements, the value to be compared with the standard.

The numerical value of a standard may also include a permitted number of exceedances of a certain numerical value in a given time period.

Ambient air pollution. Air pollution in the outdoor environment, that is, in outdoor air, but which can enter or be present in indoor environments.

Averaging time. For the purposes of this document, the duration of the exposure with a given mean concentration associated with certain health effects.

Black carbon. An operationally defined term that describes carbon as measured by light absorption. As such, it is not the same as elemental carbon, which is usually monitored with thermal-optical methods.

Concentration–response function. A statistical function or model based on the results of epidemiological studies to estimate the relative risk from air pollution for a disease or health outcome (e.g. premature death, heart attack, asthma attack, emergency room visit, hospital admission) in a population per unit concentration of an air pollutant.

Dust storm (or sand storm). A mix of dust and/or sand particles that has been elevated to great heights by a strong, turbulent wind and can travel great distances and reduce visibility. Dust or sand readily penetrates into buildings, results in severe soiling and may also cause considerable erosion. The particles are usually lifted to greater heights in a dust storm than in a sand storm.

Good practice statement. A statement formulated when a guideline development group is confident that a large body of diverse evidence, which is hard to synthesize, indicates that the desirable effects of a particular course of action far outweigh its undesirable effects. In other words, there is high certainty that implementing a measure would be beneficial, without the need for conducting numerous systematic reviews and detailed assessments of evidence.

Hot spot. For the purposes of this document, an area where air pollution levels are higher than the average levels in the local environment.

Household fuel combustion. Air pollution generated by the inefficient combustion of fuels in the household environment that results in household air pollution and contributes to local ambient air pollution.

Integrated exposure–response function. Models that combine exposure and risk data for different sources of combustion-related pollution, such as outdoor air, second-hand tobacco smoke, active smoking and household air pollution.

Interim target. An air pollutant concentration associated with a specific decrease of health risk. Interim targets serve as incremental steps in the progressive reduction of air pollution towards the air quality guideline levels and are intended for use in areas where air pollution is high. In other words, they are air pollutant levels that are higher than the air quality guideline levels, but which authorities in highly polluted areas can use to develop pollution reduction policies that are achievable within realistic time frames. The interim targets should be regarded as steps towards ultimately achieving air quality guideline levels, rather than as end targets.

Particulate matter. A mixture of solid and liquid particles in the air that are small enough not to settle out on to the Earth's surface under the influence of gravity, classified by aerodynamic diameter.

Ultrafine particle. Particles of an aerodynamic diameter less than or equal to $0.1\ \mu\text{m}$ (that is, $100\ \text{nm}$). Owing to their small mass, their concentrations are most commonly measured and expressed in terms of particle number concentration per unit volume of air (for example, number of particles per cm^3).

Abbreviations

AAQS	ambient air quality standards
ACTRIS	Aerosol, Clouds and Trace Gases Research Infrastructure
APM	anthropogenic particulate matter
AQG level	air quality guideline level
BC/EC	black carbon or elemental carbon (an indicator of airborne soot-like carbon)
BenMAP-CE	Environmental Benefits Mapping and Analysis Program – Community Edition
CanCHEC	Canadian Census Health and Environment Cohort
CCAC	Climate and Clean Air Coalition
CEN	European Committee for Standardization
CI	confidence interval
CO	carbon monoxide
COMEAP	Committee on the Medical Effects of Air Pollutants
COPD	chronic obstructive pulmonary disease
CRF	concentration–response function
EEA	European Environment Agency
ERG	external review group
EU	European Union
FAO	Food and Agriculture Organization of the United Nations
GBD	Global Burden of Disease (study)
GDG	guideline development group
<i>Global update 2005</i>	Air quality guidelines – global update 2005. Particulate matter, ozone, nitrogen dioxide and sulfur dioxide
GRADE	Grading of Recommendations Assessment, Development and Evaluation
HEI	Health Effects Institute
HR	hazard ratio
ICD-10	International Statistical Classification of Diseases and Related Health Problems, 10th edition
IHD	ischaemic heart disease

ISA	(US EPA) Integrated Science Assessment
MCC	Multi-Country Multi-City
NCD	noncommunicable disease
NDPM	net dust particulate matter
NO₂	nitrogen dioxide
O₃	ozone
PECOS	population, exposure, comparator, outcome and study design
PM	particulate matter
PM_{2.5}	particulate matter, where particles have an aerodynamic diameter equal to or less than 2.5 µm
PM₁₀	particulate matter, where particles have an aerodynamic diameter equal to or less than 10 µm
PNC	particle number concentration
ppb	parts per billion
ppm	parts per million
RBPM	regional background particulate matter
REVIHAAP	Review of evidence on health aspects of air pollution (project)
RoB	risk of bias
RR	relative risk
SDG	Sustainable Development Goal
SDS	sand and dust storms
SDS-WAS	Sand and Dust Storm Warning Advisory and Assessment System
SO₂	sulfur dioxide
Swiss TPH	Swiss Tropical and Public Health Institute
UFP	ultrafine particles
UN	United Nations
UNECE	United Nations Economic Commission for Europe
UNEP	United Nations Environment Programme
US EPA	United States Environmental Protection Agency
VOC	volatile organic compound
WMO	World Meteorological Organization

Executive summary

The global burden of disease associated with air pollution exposure exacts a massive toll on human health worldwide: exposure to air pollution is estimated to cause millions of deaths and lost years of healthy life annually. The burden of disease attributable to air pollution is now estimated to be on a par with other major global health risks such as unhealthy diet and tobacco smoking, and air pollution is now recognized as the single biggest environmental threat to human health.

Despite some notable improvements in air quality, the global toll in deaths and lost years of healthy life has barely declined since the 1990s. While air quality has markedly improved in high-income countries over this period, it has generally deteriorated in most low- and middle-income countries, in step with large-scale urbanization and economic development. In addition, the global prevalence of noncommunicable diseases (NCDs) as a result of population ageing and lifestyle changes has grown rapidly, and NCDs are now the leading causes of death and disability worldwide. NCDs comprise a broad range of diseases affecting the cardiovascular, neurological, respiratory and other organ systems. Air pollution increases morbidity and mortality from cardiovascular and respiratory disease and from lung cancer, with increasing evidence of effects on other organ systems. The burden of disease resulting from air pollution also imposes a significant economic burden. As a result, governments worldwide are seeking to improve air quality and reduce the public health burden and costs associated with air pollution.

Since 1987, WHO has periodically issued health-based air quality guidelines to assist governments and civil society to reduce human exposure to air pollution and its adverse effects. The WHO air quality guidelines were last published in 2006. *Air quality guidelines – global update 2005. Particulate matter, ozone, nitrogen dioxide and sulfur dioxide* (WHO Regional Office for Europe, 2006) provided health-based guideline levels for the major health-damaging air pollutants, including particulate matter (PM),¹ ozone (O₃), nitrogen dioxide (NO₂) and sulfur dioxide (SO₂). *Global update 2005* has had a significant impact on pollution abatement policies all over the world. Its publication led to the first universal frame of reference.

In various ways, these guidelines have stimulated authorities and civil society alike to increase efforts to control and study harmful air pollution exposures.

¹ That is, PM_{2.5} (particles with an aerodynamic diameter of $\leq 2.5 \mu\text{m}$) and PM₁₀ (particles with an aerodynamic diameter of $\leq 10 \mu\text{m}$).

In response to this growing awareness, the Sixty-eighth World Health Assembly adopted resolution WHA68.8, *Health and the environment: addressing the health impact of air pollution*, which was endorsed by 194 Member States in 2015 (WHO,2015). This resolution stated the need to redouble efforts to protect populations from the health risks posed by air pollution. In addition, the United Nations (UN) Sustainable Development Goals (SDGs) were designed to address the public health threat posed by air pollution via specific targets to reduce air pollution exposure and the disease burden from household and ambient exposure.

More than 15 years have passed since the publication of *Global update 2005*. In that time there has been a marked increase in evidence on the adverse health effects of air pollution, built on advances in air pollution measurement and exposure assessment and an expanded global database of air pollution measurements (discussed in [Chapter 1](#)). New epidemiological studies have documented the adverse health effects of exposure to high levels of air pollution in low- and middle-income countries, and studies in high-income countries with relatively clean air have reported adverse effects at much lower levels than had previously been studied.

In view of the many scientific advances and the global role played by the WHO air quality guidelines, this update was begun in 2016.

Objectives

The overall objective of the updated global guidelines is to offer quantitative health-based recommendations for air quality management, expressed as long- or short-term concentrations for a number of key air pollutants. Exceedance of the air quality guideline (AQG) levels is associated with important risks to public health. These guidelines are not legally binding standards; however, they do provide WHO Member States with an evidence-informed tool that they can use to inform legislation and policy. Ultimately, the goal of these guidelines is to provide guidance to help reduce levels of air pollutants in order to decrease the enormous health burden resulting from exposure to air pollution worldwide.

Specific objectives are the following.

- Provide evidence-informed recommendations in the form of AQG levels, including an indication of the shape of the concentration–response function in relation to critical health outcomes, for PM_{2.5}, PM₁₀, ozone, nitrogen dioxide, sulfur dioxide and carbon monoxide for relevant averaging times.

These pollutants were chosen because of their worldwide importance. However, this choice does not imply that other air pollutants are irrelevant.

- Provide interim targets to guide reduction efforts towards the ultimate and timely achievement of the AQG levels for countries that substantially exceed these levels.
- Provide qualitative statements on good practices for the management of certain types of PM (i.e. black carbon or elemental carbon (BC/EC),² ultrafine particles (UFP), and particles originating from sand and dust storms (SDS)) for which the available information is insufficient to derive AQG levels but indicates risk.

Methods used to develop the guidelines

The guidelines were formulated by following a rigorous process involving several groups with defined roles and responsibilities ([Chapter 2](#)). In particular, the different steps in the development of the AQG levels included:

- a determination of the scope of the guidelines and formulation of systematic review questions;
- a systematic review of the evidence and meta-analyses of quantitative effect estimates to inform updating of the AQG levels;
- an assessment of the level of certainty of the bodies of evidence resulting from systematic reviews for the pollutants; and
- the identification of AQG levels, that is, the lowest levels of exposure for which there is evidence of adverse health effects.

In addition, the 2005 air quality interim targets were updated to guide the implementation of the new AQG levels, and good practice statements were formulated to support the management of the specific types of PM of concern. Interim targets are air pollutant levels that are higher than the AQG levels, but which authorities in highly polluted areas can use to develop pollution reduction policies that are achievable within realistic time frames. Therefore, the interim targets should be regarded as steps towards the ultimate achievement of AQG levels in the future, rather than as end targets. The number and numerical values of the interim targets are pollutant specific, and are justified in the relevant sections of [Chapter 3](#).

The process and methods for developing these guidelines are described in detail in [Chapter 2](#).

² An indicator of airborne soot-like carbon.

The systematic reviews that informed the formulation of AQG levels and other related evidence discussed during the process are available in a special issue of *Environment International*, entitled *Update of the WHO global air quality guidelines: systematic reviews* (Whaley et al., 2021).

Recommendations on classical air pollutants

In this guideline update, recommendations on AQG levels are formulated, together with interim targets, for the following pollutants: PM_{2.5}, PM₁₀, ozone, nitrogen dioxide, sulfur dioxide and carbon monoxide (Table 0.1). The evidence-informed derivation of each AQG level and an indication of the reduction in health risk associated with the achievement of consecutive interim targets can be found in Chapter 3. Only evidence assessed as having high or moderate certainty of an association between a pollutant and a specific health outcome was used to define the recommended AQG levels, and all recommendations are classified as strong according to the adapted Grading of Recommendations Assessment, Development and Evaluation (GRADE) approach (discussed in Chapter 2).

Table 0.1. Recommended AQG levels and interim targets

Pollutant	Averaging time	Interim target				AQG level
		1	2	3	4	
PM _{2.5} , µg/m ³	Annual	35	25	15	10	5
	24-hour ^a	75	50	37.5	25	15
PM ₁₀ , µg/m ³	Annual	70	50	30	20	15
	24-hour ^a	150	100	75	50	45
O ₃ , µg/m ³	Peak season ^b	100	70	–	–	60
	8-hour ^a	160	120	–	–	100
NO ₂ , µg/m ³	Annual	40	30	20	–	10
	24-hour ^a	120	50	–	–	25
SO ₂ , µg/m ³	24-hour ^a	125	50	–	–	40
CO, mg/m ³	24-hour ^a	7	–	–	–	4

^a 99th percentile (i.e. 3–4 exceedance days per year).

^b Average of daily maximum 8-hour mean O₃ concentration in the six consecutive months with the highest six-month running-average O₃ concentration.

It is important to note that the air quality guidelines recommended in previous WHO air quality guidelines for pollutants and those averaging times not covered in this update remain valid. This includes the short averaging times for nitrogen dioxide, sulfur dioxide and carbon monoxide that were included in *Global update 2005* and indoor air quality guidelines from 2010 (and not re-evaluated in this update). [Table 0.2](#) shows existing air quality guidelines for nitrogen dioxide, sulfur dioxide and carbon monoxide with short averaging times. The reader is referred to previous volumes of air quality guidelines – *Air quality guidelines for Europe* (WHO Regional Office for Europe, 1987), *Air quality guidelines for Europe, 2nd edition* (WHO Regional Office for Europe, 2000a); and *WHO guidelines for indoor air quality: selected pollutants* (WHO Regional Office for Europe, 2010) – for other pollutants that are not covered in this 2021 update.

Table 0.2. Air quality guidelines for nitrogen dioxide, sulfur dioxide and carbon monoxide (short averaging times) that were not re-evaluated and remain valid

Pollutant	Averaging time	Air quality guidelines that remain valid
NO ₂ , µg/m ³	1-hour	200
SO ₂ , µg/m ³	10-minute	500
CO, mg/m ³	8-hour	10
	1-hour	35
	15-minute	100

Good practice statements about other PM types

As yet, insufficient data are available to provide recommendations for AQG levels and interim targets for specific types of PM, notably BC/EC, UFP and SDS. However, due to health concerns related to these pollutants, actions to enhance further research on their risks and approaches for mitigation are warranted. Good practice statements for these pollutants are summarized in [Table 0.3](#). The full text of and rationales for the statements can be found in [Chapter 4](#).

Table 0.3. Summary of good practice statements

Good practice statements	
BC/EC	1. Make systematic measurements of black carbon and/or elemental carbon. Such measurements should not replace or reduce existing monitoring of those pollutants for which guidelines currently exist. 2. Undertake the production of emission inventories, exposure assessments and source apportionment for BC/EC. 3. Take measures to reduce BC/EC emissions from within the relevant jurisdiction and, where appropriate, develop standards (or targets) for ambient BC/EC concentrations.
UFP	1. Quantify ambient UFP in terms of PNC for a size range with a lower limit of ≤ 10 nm and no restriction on the upper limit. 2. Expand the common air quality monitoring strategy by integrating UFP monitoring into the existing air quality monitoring. Include size-segregated real-time PNC measurements at selected air monitoring stations in addition to and simultaneously with other airborne pollutants and characteristics of PM. 3. Distinguish between low and high PNC to guide decisions on the priorities of UFP source emission control. Low PNC can be considered $< 1\,000$ particles/cm³ (24-hour mean). High PNC can be considered $> 10\,000$ particles/cm³ (24-hour mean) or $20\,000$ particles/cm³ (1-hour mean). 4. Utilize emerging science and technology to advance approaches to the assessment of exposure to UFP for their application in epidemiological studies and UFP management.
SDS	1. Maintain suitable air quality management and dust forecasting programmes. These should include early warning systems and short-term air pollution action plans to alert the population to stay indoors and take personal measures to minimize exposure and subsequent short-term health effects during SDS incidents with high levels of PM. 2. Maintain suitable air quality monitoring programmes and reporting procedures, including source apportionment activities to quantify and characterize PM composition and the percentage contribution of SDS to the overall ambient concentration of PM. This will enable local authorities to target local PM emissions from anthropogenic and natural sources for reduction. 3. Conduct epidemiological studies, including those addressing the long-term effects of SDS, and research activities aimed at better understanding the toxicity of the different types of PM. Such studies are especially recommended for areas where there is a lack of sufficient knowledge and information about the health risk due to frequent exposure to SDS. 4. Implement wind erosion control through the carefully planned expansion of green spaces that considers and is adjusted to the contextual ecosystem conditions. This calls for regional collaboration among countries in the regions affected by SDS to combat desertification and carefully manage green areas. 5. Clean the streets in those urban areas characterized by a relatively high population density and low rainfall to prevent resuspension by road traffic as a short-term measure after intense SDS episodes with high dust deposition rates.

PNC: particle number concentration.

The settings to which these guidelines apply

The present guidelines are applicable to both outdoor and indoor environments globally. Thus, they cover all settings where people spend time. However, as in previous editions, these guidelines do not cover occupational settings, owing to the specific characteristics of the relevant exposures and risk reduction policies and to potential differences in population susceptibility of the adult workforce in comparison with the general population.

What these guidelines do not address

These guidelines do not include recommendations about pollutant mixtures or the combined effects of pollutant exposures. In everyday life, people are exposed to a mixture of air pollutants that varies in space and time. WHO acknowledges the need to develop comprehensive models to quantify the effects of multiple exposures on human health. However, as the main body of evidence on air quality and health still focuses on the impact of single markers of ambient air pollution on the risk of adverse health outcomes, the current guidelines provide recommendations for each air pollutant individually. Achievement of the AQG levels for all these pollutants is necessary to minimize the health risk of the exposure.

Furthermore, these guidelines do not address specific recommendations on policies and interventions because these are largely context specific: what might be effective in one setting might not work in another. Lastly, individual-level interventions, such as the use of personal respiratory protection (e.g. masks, respirators, air purifiers) or behavioural measures, are addressed in another document, *Personal interventions and risk communication on air pollution* (WHO, 2020a).

Target audience

The WHO global air quality guidelines aim to protect populations from the adverse effects of air pollution. They are designed to serve as a global reference for assessing whether, and how much, exposure of a population (including particularly vulnerable and/or susceptible subgroups) to various levels of the considered air pollutants results in health concerns. The guidelines are a critical tool for the following three main groups of users:

- policy-makers, lawmakers and technical experts operating at the local, national and international levels who are responsible for developing and implementing regulations and standards for air quality, air pollution control, urban planning and other policy areas;

- national and local authorities and nongovernmental organizations, civil society organizations and advocacy groups, such as patients, citizen groups, industrial stakeholders and environmental organizations; and
- academics, health and environmental impact assessment practitioners, and researchers in the broad field of air pollution.

These groups are the targets of the information, education and communication strategies outlined in [Chapter 5](#). The strategies, and the tools to implement them, will be essential to ensure that these global guidelines are widely disseminated and considered in policy and planning decisions. In addition, these groups are addressed in [Chapter 6](#), on implementation of the guidelines. This includes the aspects involved in developing air quality standards based on the recommendations and general risk management principles, which are built on decades of experience.

Implementation of the guidelines

While achievement of the AQG levels should be the ultimate goal of actions to implement the guidelines, this might be a difficult task for many countries and regions struggling with high air pollution levels. Therefore, gradual progress in improving air quality, marked by the achievement of interim targets, should be considered a critical indicator of improving health conditions for populations. Key institutional and technical tools supported by human capacity-building are necessary to achieve this goal. Implementation of the guidelines requires the existence and operation of air pollution monitoring systems; public access to air quality data; legally binding, globally harmonized air quality standards; and air quality management systems. Policy decisions to set priorities for action will profit from the health risk assessment of air pollution.

While actions to reduce air pollution require cooperation among various sectors and stakeholders, health sector involvement is crucial for raising awareness of the impacts of air pollution on health and, thus, the economy, and for ensuring that protecting health strongly figures in policy discussions. Monitoring and evaluation are equally crucial to ensure that guidelines are implemented; they are addressed in [Chapter 7](#).

Currently, the accumulated evidence is sufficient to justify actions to reduce population exposure to key air pollutants, not only in particular countries or regions but on a global scale. Nevertheless, uncertainties and knowledge gaps remain. Future research (discussed in [Chapter 8](#)) will further strengthen the scientific evidence base for making decisions on clean air policy worldwide.

1

Introduction

The WHO air quality guidelines were last published in 2006: *Air quality guidelines – global update 2005. Particulate matter, ozone, nitrogen dioxide and sulfur dioxide* (hereafter referred to as *Global update 2005*) (WHO Regional Office for Europe, 2006). Since they were issued, air pollution has become recognized as the single biggest environmental threat to human health based on its notable contribution to disease burden. This is particularly true for PM (both PM_{2.5}, i.e. particles with an aerodynamic diameter equal to or less than 2.5 µm, and PM₁₀, i.e. particles with an aerodynamic diameter of equal to or less than 10 µm). However, other commonly measured air pollutants such as ozone (O₃), nitrogen dioxide (NO₂), sulfur dioxide (SO₂) and carbon monoxide (CO) are also of concern, as are other components of air pollution.

The burden of disease associated with both ambient and household air pollution exposure is large and growing. The growth is partly due to increases in exposures in low- and middle-income countries,³ but is in part also due to the rapidly increasing prevalence of NCDs worldwide as a result of population ageing and lifestyle changes. Air pollution especially increases morbidity and mortality from the noncommunicable cardiovascular and respiratory diseases that are the major causes of global mortality; it also increases the disease burden from lower respiratory tract infections and preterm birth and other causes of death in children and infants, which remain a major cause of the disease burden in low- and middle-income countries. Although air quality has improved gradually in high-income countries in the past decades, pollutant concentrations still exceed the levels published in *Global update 2005* for several pollutants in many areas. Air quality has generally deteriorated in most low- and middle-income countries, in step with large-scale urbanization and economic development that has largely relied on the burning of fossil fuels. Disparities in air pollution exposure are, therefore, increasing worldwide.

Science advances and, since the 2005 air quality guidelines were established, many new studies have continued to document the adverse health effects of air pollution. During this time, enormous advances have also occurred in measuring levels and trends in ground-level air pollution concentrations. In particular, the use of satellite remote sensing instruments in combination with advanced chemical transport models and ground-based measurements has substantially improved the understanding of worldwide pollution levels and trends. Studies conducted in low- and middle-income countries where concentrations are high are of great importance; however, equally important are studies in very clean areas, which answer important questions on the effects of low-level exposures and the evaluation of thresholds.

³ Country income groupings of low, lower-middle, upper-middle and high are determined by the World Bank based on gross national income per capita (World Bank, 2021).

These studies provide critical information on the benefits that might be expected if air pollution levels were reduced worldwide. In view of these many advances, revision of *Global update 2005* was both timely and necessary. This revision benefited from thousands of new studies and from following the rigorous process for developing guidelines outlined in the *WHO handbook for guideline development, 2nd edition* (WHO, 2014a).

Global update 2005 has had a significant impact on abatement policies all over the world. Its publication led to the first universal frame of reference. In various ways, the air quality guidelines have stimulated authorities and civil society alike to increase efforts to control harmful air pollution exposures. Major challenges still exist, however, and it is hoped that this update of the WHO air quality guidelines will continue to inspire and guide pollution reduction policies all over the world.

1.1 Objectives of the guidelines

The overall objective of these guidelines is to offer quantitative health-based recommendations for air quality, expressed as long- or short-term concentrations of a number of key air pollutants. Exceedance of the air quality guideline levels (hereafter referred to as AQG levels) is associated with important risks to public health. These guidelines are not legally binding standards; however, they do provide countries with an evidence-informed tool, which they can use to inform legislation and policy. In addition, the air quality guidelines will be a key component to support air quality policies globally and the development of standards, clean air policies and other tools for air quality management. Ultimately, the goal of these guidelines is to provide guidance to help reduce levels of air pollutants in order to decrease the enormous worldwide health burden resulting from exposure to air pollution.

Specifically, the objectives of these guidelines are the following.

- Provide evidence-informed recommendations in the form of AQG levels, including an indication of the shape of the concentration–response function (CRF) in relation to critical health outcomes, for PM_{2.5}, PM₁₀, nitrogen dioxide, ozone, sulfur dioxide and carbon monoxide for relevant averaging time periods. These pollutants were chosen in the process described in [section 2.3](#) because of their worldwide importance. This choice does not imply that other air pollutants are irrelevant.
- Provide interim targets to guide reduction efforts towards the ultimate and timely achievement of the AQG levels for those countries that substantially exceed the AQG levels.

- Provide qualitative statements on good practices for the management of certain types of PM – that is, BC/EC, UFP and particles originating from SDS – for which the available information is insufficient to derive AQG levels but indicates risk.

1.2 Target audience

The WHO guidelines to protect populations from the adverse effects of air pollution are designed to serve as a global reference for an audience of different groups of end-users, including those involved in policy-making, research and advocacy. Broadly, three main groups can be identified:

- policy-makers, lawmakers and technical experts at the local, national and international levels who are responsible for developing and implementing regulations and standards for air quality, air pollution control, urban planning and other policy areas;
- national and local authorities and nongovernmental organizations, civil society organizations and advocacy groups, such as patients, citizen groups, industrial stakeholders and environmental organizations; and
- academics, health and environmental impact assessment practitioners and researchers in the broad field of air pollution.

1.3 Background and rationale for updated guidelines

An update of the global WHO air quality guidelines was required for several reasons. More than 15 years have passed since the publication of *Global update 2005* and in the intervening years knowledge about the exposure of human populations, the adverse health effects of this exposure and the public health threat that it poses has seen a marked increase. Insight into global concentrations of some pollutants such as PM, ozone and nitrogen dioxide has increased dramatically ([section 1.3.1](#)). This is also true for insights in sources of emissions ([section 1.3.2](#)) and in the contribution of air pollutants to the global burden of disease ([section 1.3.3](#)). Much has been learned about the importance of addressing health inequities related to air pollution and of protecting vulnerable groups in society ([section 1.3.4](#)). Enormous advances have occurred since the early 2000s in measuring levels and trends in ground-level air pollution concentrations, and [section 1.3.5](#) provides a summary of some major trends and achievements. Finally, there have been significant advances in the worldwide adoption of the air quality guidelines presented in *Global update 2005* ([section 1.3.6](#)), and mitigating air pollution has become more central in WHO and UN activities related to achieving the UN SDGs ([section 1.3.7](#)).

1.3.1 Global concentrations and trends

Measurement of air pollutant concentrations at fixed-location monitoring sites has been the traditional approach used for air quality management, for assessment of trends and to estimate exposure for epidemiological analyses. However, despite growth in the numbers of monitoring locations globally, even for the most commonly monitored pollutants, coverage is inadequate – that is, it is often restricted to major cities – to accurately estimate exposure in the many different places where people live. There are two major gaps.

The first is a lack of monitoring in many countries of the world and inadequate monitoring in rural areas or outside of major cities in many countries. Although there is increasing coverage of PM monitoring, coverage for other pollutants such as ozone, nitrogen dioxide and sulfur dioxide is less extensive. The second gap relates to inadequate monitoring to characterize the spatial variation in specific air pollutants within cities. In particular, this holds for concentrations of pollutants such as nitrogen dioxide and black carbon and UFP (diameter of $\leq 0.1 \mu\text{m}$; or broader quasi-UFP, as discussed in [section 4.3](#) on UFP), which may vary by an order of magnitude over just a few hundred metres (Karner, Eisinger & Niemeier, 2010). Since 2010, there has been a dramatic improvement in the combination of satellite data retrievals and chemical transport models with land-use information and ground measurements to estimate concentrations globally, which have been used to address the first gap (Shaddick et al., 2018; Brauer et al., 2012, 2016; Larkin et al., 2017; de Hoogh et al., 2016; Novotny et al., 2011; Hystad et al., 2011; Knibbs et al., 2014; Chang et al., 2019). To address the second gap, land-use regression models (Hoek et al., 2008) have been used increasingly – these models capture within-city variability, as discussed for example for UFP (Morawska et al., 2008), and have been scaled up to the global context for nitrogen dioxide (Larkin et al., 2017).

Although in many countries, regional and local authorities maintain accessible databases of air quality measurements, the only global databases are the WHO Global Ambient Air Quality Database and OpenAQ. The WHO Global Ambient Air Quality Database provides information on the annual average concentrations of PM₁₀ and PM_{2.5} for specific cities based on available measurements (including averages from multiple monitors within a single city, where these are available) (WHO, 2021a). OpenAQ is a non-profit-making effort to maintain an open-source database of aggregated current and archived air quality data gathered in real time from government agencies (OpenAQ, 2021). Despite the progress made in monitoring and in data access, many publicly funded agencies still do not provide easy access to data.

Exposure to air pollutants is heavily dependent on their ambient concentrations. Ambient PM_{2.5} concentrations vary substantially between and within regions of the world. Importantly, more than 90% of the global population in 2019 lived in areas where concentrations exceeded the 2005 WHO air quality guideline of 10 µg/m³. In 2019 annual population-weighted PM_{2.5} concentrations were highest in the WHO South-East Asia Region, followed by the WHO Eastern Mediterranean Region. Elevated concentrations were also observed in some western African countries, largely due to the impact of Saharan dust. Windblown desert dust sometimes contributes to very high exposures to coarse particles larger than 2.5 µm or 10 µm in diameter. This is a prominent issue in many arid areas in the Middle East, northern Africa, the Gobi desert and elsewhere.

Many of the countries with the lowest national PM_{2.5} exposure levels were either in the WHO Region of the Americas or parts of the WHO European Region. Population-weighted PM_{2.5} concentrations averaged 7 µg/m³ or less in these countries. Trends in PM_{2.5} indicate a relatively stable population-weighted global mean concentration, which reflects both decreases in exposure in the WHO European Region, the WHO Region of the Americas and the WHO Western Pacific Region but increases elsewhere.

Population-weighted ozone concentrations vary less dramatically than is the case for PM_{2.5}, for example ranging from 30–50 µg/m³, mostly in small island nations, to 120–140 µg/m³ in Asia and the Middle East. Among the world's most populous countries in southern Asia, population-weighted seasonal ozone concentrations range up to approximately 130 µg/m³. Concentrations in African mega-cities are also likely to be high but there is still comparatively little documentation.

Trends in ozone at a regional scale show little change over time, although decreases within North America and Europe and increases in the Middle East and much of Asia are apparent.

The patterns of ambient nitrogen dioxide concentrations are quite different from those of PM_{2.5} and ozone, with the highest population-weighted concentrations in eastern Asia, the Middle East, North America and much of Europe, reflecting mobile sources (Larkin et al., 2017; Achakulwisut et al., 2019). In addition, nitrogen dioxide displays a distinct urban–rural gradient, with higher concentrations in more densely populated urban areas. This pattern contrasts distinctly from that of ozone, which displays higher concentrations downwind of urban areas, and PM_{2.5}, which is more homogeneous regionally due to its longer atmospheric lifetime and diversity of (urban, rural and regional) sources. Trends in population-weighted nitrogen dioxide concentrations (for 1992–2012) indicated

sharp decreases (-4.7%/year) in high-income North American countries and somewhat lesser decreases in western Europe (-2.5%/year) and high-income Asia-Pacific countries (-2.1%/year). In contrast, population-weighted nitrogen dioxide concentrations increased dramatically during this period in eastern Asia at a rate of 6.7%/year. Judging from satellite observations, concentrations in Africa seem to be generally low, with some evidence of increases in northern Africa and stable or slightly decreasing levels elsewhere (Geddes et al., 2016). However, there are few actual monitoring data on small-scale spatial variability within mega-cities in Africa.

1.3.2 Sources of emissions and exposure

Air pollution originates from numerous sources of emission, both natural and anthropogenic, with the latter becoming globally dominant since the beginning of industrialization. The process of combustion is the greatest contributor to air pollution, in particular, the combustion of fossil fuels and biomass to generate energy. In indoor environments, the use of polluting fuels in unvented heating and cooking stoves, tobacco combustion and combustion for other purposes, such as cultural or religious practices are also important. Fossil and biomass fuel burning for domestic heating is also an important source of outdoor air pollution in many parts of the world.

Outdoor combustion sources include land, air and water transportation; industry and power generation; and biomass burning, which includes controlled and uncontrolled forest and savannah fires and agricultural waste burning, as well as waste burning in urban areas. Other sources and processes contributing to outdoor pollution are the resuspension of surface dust and construction activities. Long-range atmospheric transport of pollutants from distant sources contributes to local pollution, particularly urban air pollution. Some of the pollutants are emitted directly by combustion sources as primary pollutants (with elemental carbon as the main constituent of PM), and some are formed in the air as secondary pollutants (such as nitrates, sulfates and organic carbon) through complex physicochemical processes involving gaseous precursors originating from combustion sources, agriculture (ammonia), other anthropogenic processes and natural processes such as biogenic emissions.

Comprehensive reviews of sources and concentrations of major outdoor air pollutants have been published by the United States Environmental Protection Agency (US EPA) (2010, 2016, 2017, 2019a, 2020). The European Environment Agency (EEA) every year produces a comprehensive report on air quality in Europe; the latest one from 2020 (EEA, 2020).

In indoor environments, pollution is also generated by combustion sources, mainly cooking and heating with polluting fuels such as coal, wood or dung; and using candles, incense and kerosene lamps (e.g. for light or religious practices). Tobacco smoking is also a significant source of indoor pollution. Non-combustion sources and processes also have a significant impact on indoor air pollution, particularly those that generate volatile and semi-volatile organic compounds (VOCs) and/or ozone. These include the renovation of houses, usage of consumer products (e.g. cleaning products and insecticides) and operation of electric devices such as laser printers. Dust resuspension due to human movement is another significant source in some indoor environments, particularly in schools. However, indoor air pollution is generated not only from indoor sources but also from outdoor air pollutants that are brought indoors in the processes of ventilation and penetration through the building envelope. In indoor environments without indoor sources of pollution, pollutants from outdoors are the main cause of indoor air pollution. Exposure is then further influenced by indoor decay, which is very fast for substances such as ozone (which is very reactive) and very slow for substances such as carbon monoxide (which is fairly inert).

Airborne pollutants originating from the sources and processes listed above include PM (measured as PM_{2.5}, PM₁₀ and UFP), gaseous pollutants (including ammonia (NH₃), carbon monoxide, nitrogen dioxide, sulfur dioxide and ozone) and organic air pollutants. PM is partly formed in the atmosphere through chemical reactions that produce inorganic nitrates and sulfates, as well as organic compounds summarized as organic carbon. Other airborne pollutants not discussed in this document include radon and its decay products, and biological agents. WHO has developed dedicated air quality guidelines for these and for other selected pollutants, dampness and mould, and household fuel combustion (WHO, 2014b; WHO Regional Office for Europe, 2009, 2010).

The spatial and temporal concentration of pollutants in outdoor air varies according to the spatial distribution of the sources and their pattern of operation (e.g. daily or seasonal), the characteristics of the pollutants and their dynamics (dispersion, deposition, interaction with other pollutants), and meteorological conditions. In urban environments, some pollutants are distributed more homogeneously than others; for example, PM_{2.5} concentration has much less spatial variation compared with the concentration of UFP or gases directly emitted by local combustion sources. Importantly, spatial variation determines to what extent ambient concentrations measured at a single fixed site reflect the outdoor concentrations at other sites in the area. Temporal variation is a very important feature of ambient air pollution.

Emissions often have specific and predictable temporal patterns (e.g. weekdays versus weekends). Most importantly, however, meteorological conditions are very strong determinants of temporal variations, and can have far larger effects than the temporal variation in emission alone. Epidemiological research of short-term health effects capitalizes on these short-term temporal variations in ambient concentrations. It offers opportunities to investigate whether temporally varying markers of health, including the number of adverse health events, correlate with the temporal variation in ambient concentrations of pollutants.

In indoor environments, concentrations of pollutants originating from outdoor air are influenced by their outdoor spatiotemporal patterns of concentration and, in particular, by the proximity of the building to outdoor sources (e.g. busy roads). Furthermore, indoor pollution concentrations depend on the amount of air pollution penetrating from outdoors; this is dependent on the penetration fraction, the ventilation rate and the decay rate. The penetration coefficient varies for different particle size fractions and is highest for PM_{2.5}. Finally, indoor pollution concentrations depend on the temporal pattern of operation of outdoor sources (e.g. traffic) but also on indoor sources (e.g. the daily cycle of cooking) and the decay process (in the case of highly reactive gases such as ozone).

People are exposed to air pollution in all the microenvironments in which they spend time, and the exposure puts them at risk. A microenvironment is defined as a three-dimensional space in which the pollutant level is uniform at some specified time. Exposure is a product of the pollutant concentration and the time over which a person is in contact with that pollutant. Assessment of exposure constitutes an element of risk assessment that is schematically represented as a chain of events from emissions through air pollution concentrations, population exposure, and body burden and pollutant dose at the organ or cellular level, to health risk.

In some locations, pollutant concentrations are low but the overall contribution to the exposure is high because of the longer time spent there (e.g. at home); in other locations, pollutant concentrations are very high (e.g. at traffic hot spots), and even short periods of time spent at such locations result in high exposures. When concentration varies with time, the time-averaged concentration is used for exposure calculation. For health risk assessment, exposures are defined on different time domains as (i) lifetime exposure, which is the sum of exposures that occurred in different environments – this is particularly important for carcinogenic pollutants; (ii) long-term exposure, measured as a mean of one or several years; and (iii) short-term exposure, measured over minutes to days.

Considering indoor exposures is important because people spend most of their time in various indoor environments, including home, workplace, school and commuting (where the microenvironment is a bus, car or train). Indoors is also where exposure predominantly occurs for vulnerable population groups, as sick and older people may not venture outside much. Although the exposures occur indoors, they are caused by both outdoor and indoor sources of emissions, since outdoor pollutants penetrate indoors, as discussed above.

The most accurate assessment of the risk caused by total air pollution would be based on the assessment of each individual's personal exposure, which would require pollution measurements in each microenvironment in which the individual spends time and an accurate account of the time spent there (time–activity diary). Yet, the most accurate assessment of exposure to ambient outdoor pollution – which is subject to clean air policy-making – may not necessarily be the measurement of personal exposure, unless the measured indicator of pollution is clearly and solely of outdoor origin. Presently it is not possible to measure all of the relevant pollutants in all microenvironments for each individual; therefore, the approach to exposure assessment is pragmatically based on the purpose of the assessment. For example, for studies on the long-term impact of outdoor air pollution (chronic effects), data are typically sourced from a limited number of monitors operating in some central outdoor locations. This has been shown to effectively represent population exposure to outdoor pollutants that are distributed more homogeneously, such as PM_{2.5} or ozone. More complicated is exposure assessment for studies on the acute effects of air pollution (such as mortality or hospital admissions), where spatiotemporal variations in pollution need to be taken into account. However, for many pollutants, daily concentrations are often very highly correlated temporally across rather large regions and, thus, temporal variation may be well captured by single monitors.

Advanced methods of exposure assessment are available, including not only ground base monitoring of pollution but also the use of satellite observations and various modelling tools such as chemical transport models and land-use regression models. Those modelling approaches have overcome some of the former limitations of reliance on only a few monitoring stations to describe population exposure in space and time.

1.3.3 Disease and economic burden

Air pollution is the leading environmental risk factor globally. WHO estimates show that around 7 million deaths, mainly from noncommunicable diseases, are attributable to the joint effects of ambient and household air pollution (WHO, 2018).

Similar global assessments of ambient air pollution alone suggest between 4 million and 9 million deaths annually and hundreds of millions of lost years of healthy life, with the greatest attributable disease burden seen in low- and middle-income countries (Burnett et al., 2018; GBD 2019 Risk Factors Collaborators, 2020; Vohra et al., 2021; WHO, 2018). To date, strong evidence shows causal relationships between PM_{2.5} air pollution exposure and all-cause mortality, as well as acute lower respiratory infections, chronic obstructive pulmonary disease (COPD), ischaemic heart disease (IHD), lung cancer and stroke (Cohen et al., 2017; WHO, 2018). A growing body of evidence also suggests causal relationships for type II diabetes and impacts on neonatal mortality from low birth weight and short gestation (GBD 2019 Risk Factors Collaborators, 2020). Air pollution exposure may increase the incidence of and mortality from a larger number of diseases than those currently considered, such as Alzheimer's and other neurological diseases (Peters et al., 2019). The burden of disease attributable to air pollution is now estimated to be competing with other major global health risks such as unhealthy diet and tobacco smoking, and was in the top five out of 87 risk factors in the global assessment (GBD 2019 Risk Factors Collaborators, 2020).

At the time of publishing these guidelines, global burden estimates are limited to PM_{2.5} and ozone. Other common pollutants such as nitrogen dioxide and sulfur dioxide are not yet included and, therefore, these figures based on exposure to PM_{2.5} and ozone are likely to underestimate the full health toll from ambient air pollution. For example, an analysis of the disease burden attributable to nitrogen dioxide on one outcome, incident paediatric asthma, indicated that nitrogen dioxide pollution was responsible for 13% of the burden (Achakulwisut et al., 2019). With a spatial pattern quite different than that for PM_{2.5}, exposure to nitrogen dioxide resulted in a comparatively high burden in many high-income countries.

Air pollution also leads to health-related economic impacts. Such impacts arise via two major pathways. The first, human health costs, are those related to the incidence of disease and mortality and are estimated by a willingness-to-pay approach. The second is due to lost labour productivity. In 2013 the World Bank estimated a global economic impact of US\$ 143 billion in lost labour income and of US\$ 3.55 trillion in welfare losses from exposure to PM_{2.5} (World Bank, 2016). The welfare losses ranged from an equivalent of 1% of gross domestic product in low-income countries to 5% in high-income countries not within the Organisation of Economic Co-operation and Development. Apart from the health-related burden, air pollution causes additional economic costs such as through its impact on agricultural crops or through damage to buildings and infrastructure. In addition, there are costs associated with air pollution-related climate change and environmental degradation.

Although some uncertainty surrounding the exact disease burden remains (discussed in [Chapter 8](#)), it is clear that the global burden of disease associated with air pollution takes a massive toll on human health and the economy worldwide: exposure to air pollution is estimated to cause millions of deaths and lost years of healthy life, as well as a loss of trillions of dollars annually. Air pollution is now recognized as the single largest environmental threat to human health and well-being.

1.3.4 Inequities and vulnerable and susceptible groups

As already discussed, air pollution from both ambient sources and household use of polluting fuels is a recognized threat to human health, even at low exposures, and causes increased mortality and morbidity worldwide.

This burden of disease is unevenly distributed, often disproportionately affecting the most vulnerable and susceptible populations. The impact of air pollution can be seen on vulnerable individuals with greater exposure levels and susceptible individuals with chronic conditions (such as asthma, COPD, diabetes, heart failure and IHD), as well as children and pregnant women.

According to WHO, health equity is the “the absence of unfair and avoidable or remediable differences in health among population groups defined socially, economically, demographically or geographically” (WHO, 2020b). Health inequities, therefore, involve more than inequality with respect to health determinants, access to the resources needed to improve and maintain health, and health outcomes. They also entail a failure to avoid or overcome inequalities that infringe on fairness and human rights norms.

The fact that this burden of disease and mortality is unevenly distributed also impedes reduction of inequities and progress towards achieving full human rights and the UN SDGs. Global efforts to reduce pollution levels will have a positive impact on lowering inequity (Universal Declaration of Human Rights, Art. 1 and Art. 2) and will promote the right of life and security by ensuring safe and healthy environments (as stated in Art. 3) (UN, 1948).

Successful interventions are feasible, effective and compatible with economic growth. However, only a few studies have looked at equity in health when evaluating intervention delivery. In general, interventions that aim to reduce air pollution in urban areas have a positive impact on air quality and mortality rates, but the documented effect on equity is less straightforward. There is no evidence on whether applied air pollution reduction interventions have reduced health inequalities, since results from studies published to date have been mixed and

not all interventions have had a positive distribution of health benefits. Indeed, depending on the health outcome(s) under study and intervention type/study design (simulations of air pollution concentrations or real interventions), more vulnerable groups such as older persons and deprived households were found to benefit more, equally or less than their socially better-off counterparts. For an in-depth review of published studies until the early 2010s, see Benmarhnia et al. (2014).

The largest inequities in air pollution exposure occur on the global rather than the local scale. Indeed, countries with policy-driven improvements in air quality have often seen particularly steep declines in pollution at hot spots since the 1990s, whereas declines have been gradual in regions with already good air quality. However, on a global scale, the steep decline in pollution in the vast majority of high-income countries is paralleled by an unprecedented increase in low- and middle-income countries. As documented by Zhang et al. (2017), the model of globalized movements of goods with inequities in emission and air quality standards contributes to inequity in air quality (UNEP, 2020). Weak policies in low- and middle-income countries allow pollution from the production of goods that are ultimately consumed in part in high-income countries.

1.3.5 Progress on scientific evidence

There has been tremendous progress in the scientific understanding of the health effects of air pollution since the early 2000s.

First of all, health effects of air pollution have now been studied in most WHO regions; in contrast, almost all evidence underpinning *Global update 2005* came from studies in Europe and North America. This is especially true for studies of short-term effects on mortality and morbidity (Chen et al., 2017; Yang J et al., 2020). However, quite a few studies of long-term effects have now also been reported, especially from Asia and Oceania. These studies have generally found relationships between air pollutants and ill-health that are qualitatively similar to those in high-income countries, although the CRFs are sometimes quantitatively different, with less steep relationships at high than at low concentrations (Yang X et al., 2020; Hanigan et al., 2019).

Secondly, air pollution has now been implicated in the development or worsening of several health conditions not considered in previous research. These include, among others, asthma, diabetes, reproductive outcomes and several neurocognitive end-points (Yang B-Y et al., 2020; Paul et al., 2019) (Thurston et al., 2017).

Thirdly, many studies have tried to identify which sources and/or physicochemical characteristics of airborne PM contribute most greatly to toxicity. This is a challenging area of research, given the great heterogeneity of airborne particles, and a definitive set of particle characteristics has yet to be identified. However, in its 2013 review of the evidence (WHO Regional Office for Europe, 2013a), WHO did point out that a focus on primary combustion particles, secondary inorganic aerosols and secondary organic aerosols was warranted (Thurston et al., 2016b; US EPA, 2019a; Lippmann et al., 2013; Vedal et al., 2013).

Lastly, investigators have learned to collaborate on an unprecedented scale. Prior to 2005, there were few examples of multicentre studies in the domain of time-series studies investigating the short-term effects of air pollution; two notable examples are the Air Pollution and Health, a European Approach (APHEA) studies in Europe and the National Morbidity and Mortality Air Pollution Study (NMMAPS) in the United States of America. These were followed after 2005 by the Air Pollution and Health: A European And North American Approach (APHENA) study across Europe, Canada and United States (Samoli et al., 2008); the ESCALA (Estudio de Salud y Contaminación del Aire en Latinoamérica) study in Latin America (Romieu et al., 2012); and the Public Health and Air Pollution in Asia (PAPA) study in Asia (Wong et al., 2008) – all studies of short-term effects. A remarkable culmination is the Multi-Country Multi-City (MCC) Collaborative Research Network (Chen et al., 2021; Liu et al., 2019; Meng et al., 2021; Vicedo-Cabrera et al., 2020), which combines multiyear data from 652 cities across the world in a single joint analysis of the short-term effects of PM_{2.5}, ozone, nitrogen dioxide and carbon monoxide, among other studies. Large collaborations have also emerged in studies of long-term effects such as the European Study of Cohorts for Air Pollution Effects (ESCAPE), which includes data from 36 different cohorts (Beelen et al., 2014). Another example is the Global Exposure Mortality Model (GEMM), which includes data from 41 cohorts from 16 countries across the globe (Burnett et al., 2018). Finally, an ongoing collaboration is studying the long-term health effects of low levels of air pollution in Europe (HEI, 2021), Canada and the United States (Brauer et al., 2019; Dominici et al., 2019).

Collectively, these studies have considerably strengthened the evidence for health effects of air pollution by increasing study power and using highly standardized preplanned methods of data collection, analyses and reporting (Brauer et al., 2019; Di et al., 2017a).

Methods of assessing exposure to air pollution have become much more refined. In 2005 the annual air quality guideline for PM_{2.5} was largely based on results from two studies, the Harvard Six Cities study (Dockery et al., 1993) and the

American Cancer Society Cancer Prevention Study II (Pope et al., 2002). In these studies, exposure to PM_{2.5} was assessed from one or a few monitoring sites per city. In addition, advanced chemical transport models, land-use regression models, satellite observations and much more detailed ground-level monitoring have formed the basis for very detailed assessment of exposure to PM_{2.5} (as well as other pollutants) at very fine temporal and spatial scales. This has been useful not only for population studies of health effects but also for estimating the worldwide health impact of air pollution (Hammer et al., 2020; de Hoogh et al., 2018).

These new methods of exposure assessment have facilitated studies of nationwide populations, not only those living in cities but also those living in rural areas where air pollution monitoring is sparse or even absent. Often, these nationwide studies make use of administrative databases, which have increasingly become automated. These include death registers, disease registers, census data and population statistics. Such studies have the advantage of often including large populations of millions or even tens of millions of subjects. In addition, the data included are often more representative of underlying populations than regular cohort studies. A disadvantage of such databases is that they usually do not contain much information on potential confounding and modifying factors such as smoking and diet. However, innovative solutions have been developed to deal with this (e.g. survey results in Medicare and indirect adjustment for covariates in Canadian census studies) (Crouse et al., 2015; Cesaroni et al., 2013). Such databases usually also lack information from biological markers and specimens and, thus, cannot shed light on biological pathways to explain the observed associations.

Advances in statistical analyses techniques and conceptualization of causal modelling in epidemiology have produced new insights into the robustness of epidemiological associations between air pollutants and health effects. Machine learning techniques are increasingly being applied to explain patterns in complex exposure patterns. Most recently, large collaborative studies of the so-called exposome (defined as the totality of exposure individuals experience over their lives and how these exposures affect health) have started in an attempt to understand the effects of lifelong exposures to complex environmental factors on the development of health and disease throughout the life course. In such studies, air pollution is regularly included as one of several sets of complex environmental exposures and is combined with individual data, ranging from the molecular, genetic or cellular level up to the level of social, cultural and lifestyle data (Vrijheid et al., 2020).

Decision-makers have increasingly asked for reliable estimates of the burden of disease caused by air pollution as input for cost–benefit analyses of policy alternatives and as a basis for risk communication. Since 2005, major steps forward have been taken, especially by WHO and the Global Burden of Disease (GBD) project. An innovative, integrated exposure–response function was developed, integrating insights from studies on outdoor air pollution, on the health effects of indoor exposure to household air pollution from solid fuel combustion and environmental tobacco smoke, and on active smoking (Burnett et al., 2014). The integrated exposure–response function formed the basis for the first-ever truly global burden of disease estimate from exposure to PM_{2.5}, ozone and household air pollution from solid fuel burning, published in 2012 (Lim et al., 2012). These estimates used the global exposure estimates mentioned in [section 1.3.1](#) and worldwide data on mortality and morbidity. They have been updated several times as new exposure estimates became available, and the integrated exposure–response function was updated based on new study findings (Cohen et al., 2017). The latest version no longer includes studies on active smoking, for instance (GBD 2019 Risk Factors Collaborators, 2020). Widely available software tools, such as WHO AirQ+ (WHO Regional Office for Europe, 2021a) or the US EPA's Environmental Benefits Mapping and Analysis Program – Community Edition (BenMAP-CE) (US EPA, 2021) facilitate similar analysis on a local (city, region, country) level.

Decision-makers have also sought evidence that measures to reduce air pollution actually produce health benefits. So-called accountability research (i.e. assessment of the effectiveness of interventions) addresses the consequences of policy interventions. An early example is a study from Dublin suggesting that a ban on coal burning led to reduced mortality (Clancy et al., 2002; Dockery et al., 2013). A nationwide study from the United States found that life expectancy increased most in areas where fine particle concentrations decreased the most (Pope, Ezzati & Dockery, 2009). A research programme on this subject, developed by the United States-based Health Effects Institute (HEI), showed promise, as well as pitfalls (Boogaard et al., 2017), while a Cochrane review on interventions to reduce ambient air pollution and their effects on health concluded that more research is needed in this area to reduce uncertainty (Burns et al., 2019).

Another issue of great interest to decision-makers is the that the co-benefits of policies aimed at reducing greenhouse gases may also have adverse direct or indirect health effects (e.g. methane, a powerful greenhouse gas and an ozone precursor) or, conversely, that policies aimed at reducing health-relevant air pollutants (such as black carbon) may also have climate forcing capabilities.

1.3.6 Adoption of the 2005 air quality guidelines worldwide

The first two editions of the air quality guidelines in 1987 and 2000 were successful in providing guidance, mostly to European countries, and provided the basis for the European Union (EU) legislation on air quality. *Global update 2005* was intended to be relevant to the diverse conditions within all WHO regions.

Evidence-informed guidance on air quality and associated health effects is necessary so that countries can use this information in standard setting and in providing information to the public. In 2012 a review of the processes followed to establish national ambient air quality standards (AAQS) for PM₁₀ and sulfur dioxide (24-hour average) in the period 2007–2008 concluded that WHO air quality guidelines were the resource used most often to establish or revise national standards by the relevant authorities (Vahlsing & Smith, 2012). At that time, 91% of the countries that responded to a survey planned on using *Global update 2005* for future revision of their AAQS; however, this information was only available for 96 countries. In collaboration with WHO, the Swiss Tropical and Public Health Institute (Swiss TPH) has compiled information on the existence of legally binding AAQS for all UN Member States for PM (PM_{2.5}, PM₁₀ and other relevant types), ozone, nitrogen dioxide, sulfur dioxide and carbon monoxide for different averaging times (both long and short term) (Kutlar Joss et al., 2017; WHO, 2021b). This unique update of the current state of AAQS worldwide provides a useful insight into the degree to which the 2005 air quality guidelines and interim targets are used as a basis for legally binding and non-binding AAQS. Information was identified for over 170 countries in the different WHO regions, of which 53 did not define any standards (see Table 1.1). In general, standards for short-term exposure were set more often than annual limit values. Levels varied greatly by country and by air pollutant.

Daily mean standards for PM₁₀ and sulfur dioxide (averaging time ≤ 24 hours) and 1-hour maximum values for nitrogen dioxide were most often defined. Although compliance with WHO air quality guidelines was rather low, it was generally higher for short-term than for long-term standards. Among all countries with standards for 24-hour averaging times for PM_{2.5} and PM₁₀, 21% and 46% met the air quality guidelines, respectively. In contrast, only seven countries (2%) adopted the WHO annual mean air quality guidelines for PM₁₀ and PM_{2.5}. In case of sulfur dioxide (24 hours), only 7% of countries were in line with the air quality guidelines and 16% aligned their standard with the 1-hour guidelines for ozone. Adoption rates were higher for nitrogen dioxide, sulfur dioxide (10-minute averaging time) and carbon monoxide.

In addition, in the EU, WHO guidelines are referenced in the Ambient Air Quality Directive (European Parliament & Council of the European Union, 2008),

and several countries use/will use WHO air quality guidelines and/or interim targets within existing and forthcoming legislation.

Analysis of the level of adoption of WHO air quality guidelines (see [Table 1.1](#)) shows that many countries have guidelines or standards for at least one air pollutant; however, there are many countries without standards or where information is lacking. The gap between the WHO air quality guidelines and the levels adopted in national regulations reflects the policy-making process. Whereas the WHO guidelines are evidence-informed, health-oriented recommendations, the process of developing legally binding regulations is driven by national policy-makers and the willingness to set environmental standards. This process involves different actors and may be influenced by a range of considerations.

Table 1.1. Adoption of WHO air quality guidelines in different regions

WHO region	Countries in the region (n)	Countries with standards for at least one pollutant and averaging time		Countries without standards		Countries with no information	
		n	%	n	%	n	%
African Region	47	17	36	21	45	9	19
Region of the Americas	35	20	57	13	37	2	6
South-East Asian Region	11	7	64	3	27	1	9
European Region	53	50	94	2	4	1	2
Eastern Mediterranean Region	21	11	52	1	5	9	43
Western Pacific Region	27	12	44	13	48	2	7
Total	194	117	60	53	27	24	12

Source: Kutlar Joss et al. (2017).

The difficulty of attaining the air quality guidelines for PM and other pollutants was recognized in *Global update 2005*, and a series of interim targets were set to provide milestones for countries on the way to achieving the air quality guidelines. Interim targets were defined as air pollutant levels that are higher than the air quality guidelines, but which authorities in highly polluted areas can

use to develop pollution reduction policies that are achievable within realistic time frames. The interim targets should be regarded as steps towards ultimately achieving air quality guidelines in the future, rather than as end targets. The number and numerical values of the interim targets are pollutant specific and they are justified in the relevant sections of [Chapter 3](#).

1.3.7 Air pollution and health in the global agenda

World Health Assembly resolution and road map

In May 2015 the Sixty-eighth World Health Assembly adopted resolution WHA68.8, *Health and the environment: addressing the health impact of air pollution*, which was endorsed by 194 WHO Member States (WHO, 2015). This resolution stated the need to redouble the efforts of Member States and WHO to protect populations from the health risks posed by air pollution. Member States were urged to raise public and stakeholder awareness on the impacts of air pollution on health; provide measures to reduce or avoid exposure; facilitate relevant research; develop policy dialogue, strengthen multisectoral cooperation at national, regional and international levels; and take effective steps to reduce health inequities related to air pollution.

Specifically, the resolution recognized the role of the WHO air quality guidelines, for both ambient and indoor air quality, in providing guidance and recommendations for clean air that protect human health. It requested the Director-General to strengthen WHO capacities in the field of air pollution and health through further development and regular updating of the WHO air quality guidelines to facilitate effective and efficient decision-making, and to provide support and guidance to Member States in their efficient implementation. A road map for implementation of this resolution on air pollution and health was presented at the Sixty-ninth World Health Assembly and approved by Member States (WHO, 2016a).

UN Sustainable Development Agenda and other UN processes

The WHO air quality guidelines support the strategic priorities for NCDs (UN, 2018a), as well as those established in the 2030 Agenda for Sustainable Development, which was adopted at the United Nations Sustainable Development Summit in 2015 (UN, 2015). These priorities emphasize the need to strengthen national capacities to reduce modifiable risk factors, including air pollution, for NCDs and to accelerate countries' responses for their prevention and control. The 17 SDGs contained in the Agenda present an indicator framework for global monitoring and include 169 specific associated targets (UN Statistics Division, 2020).

These, in turn, are divided into indicators, thereby providing a tool for quantitative assessment of achievement towards meeting the goals. This update of the WHO air quality guidelines provides evidence-informed benchmarks on the health impacts of air pollution, and will help assess the following air pollution-related SDG indicators to inform the health trends associated with exposure to air pollution:

- Indicator 3.9.1: Mortality rate attributed to household and ambient air pollution
- Indicator 7.1.2: Percentage of population with primary reliance on clean fuels and technology
- Indicator 11.6.2: Annual mean levels of fine PM (population-weighted).

The health impacts of air pollution are a main driver for action by the environment sector. The UN Environment Assembly adopted the following three resolutions on the topic.

- Resolution 1/7 from the United Nations Environment Programme (UNEP), adopted at its first session in 2014 on Strengthening the role of the United Nations Environment Programme in promoting air quality, highlights the effects of air pollution, especially from a perspective of sustainable development. In particular, it encourages governments to take cross-sectoral action to improve air quality and formulate action plans while establishing (and implementing) nationally determined air quality and emissions standards, taking into account relevant information (e.g. WHO guidelines) (UNEP, 2014).
- Additionally, the UN Environment Assembly presented a resolution, at its second session in 2016, requesting the Executive Director to engage with all relevant UN entities to promote a coordinated approach to combating the challenges of SDS globally by supporting Member States in the identification of relevant data and information gaps, best policy measures, and actions to address the problem and by inviting them to intensify monitoring data collection and knowledge sharing on all relevant aspects of SDS, including their impact on ecosystems and on human health and well-being (UNEP, 2016a).
- Finally, the resolution on Preventing and reducing air pollution to improve air quality globally calls for Member States to take action across sectors to reduce all forms of air pollution. Among its recommendations, the resolution urges Member States to:
 - consider joining or cooperating with, as appropriate, relevant global initiatives such as the Climate and Clean Air Coalition and the Global Methane Initiative; [and] facilitate action to reduce air pollution in urban and rural areas including by encouraging cities and local governments to consider participating in, as appropriate, the BreatheLife campaign (UNEP, 2018).

Lastly, a report from 2019 from the UN Special Rapporteur on the Issue of Human Rights Obligations Relating to the Enjoyment of a Safe, Clean, Healthy and Sustainable Environment highlighted the different state obligations in relation to the right to breathe clean air, as well as the specific obligation to protect people and groups in vulnerable situations (UN, 2019a). The Special Rapporteur focused on the right to breathe clean air as one of its components and describes the negative impact of air pollution on the enjoyment of many human rights, in particular the right to life and the right to health, especially by vulnerable groups. The Special Rapporteur identified several good practices implemented worldwide that have helped to improve air quality; offered a number of recommendations to Member States for actions they should consider as part of a national air quality action plan; and urged businesses, in order to fulfil their responsibility in this regard, to contribute to and support efforts to reduce air pollution.

1.4 WHO guidelines relating to air quality

WHO air quality guidelines have been widely used as a reference tool to help decision-makers across the world in setting standards and goals for air quality management. Since the mid-1980s, WHO has coordinated the development of several editions of air quality guidelines for both ambient and indoor air quality. Although the methodologies used and the requirements needed to produce them have evolved over time, these guidelines remain in essence manuals aiming to provide evidence-informed recommendations in the form of air quality guidelines for different averaging times (WHO Regional Office for Europe, 2017).

Since 2009, WHO has issued a separate series of guidelines for indoor air quality, which provide recommendations on biological contaminants of indoor air, selected air pollutants typically measured in indoor settings, and household fuel combustion.

Air quality guidelines for Europe (1987)

The first volume of the air quality guidelines created the initial framework for the scientific rationale for the series. The expert panel formulated guidelines for 28 air pollutants on exposure in both outdoor and indoor environments. In specific cases (e.g. mercury), a guideline level was formulated for indoor settings only. For 19 noncarcinogenic pollutants, recommendations were provided as guideline values using the toxicological concepts of the lowest/no observed adverse effect level and protection factors. In contrast, ranges were provided for cadmium, lead, PM (expressed as black smoke), ozone and sulfur dioxide, with a recommendation to use gravimetric methods for measuring particles.

Because of the impossibility of identifying no-effect levels of exposure, the panel recommended unit risk factors for carcinogenic (genotoxic) pollutants (WHO Regional Office for Europe, 1987).

Air quality guidelines for Europe. second edition (2000)

In response to strengthening of the evidence during the 1990s, the revised air quality guidelines were published in the year 2000. A total of 35 air pollutants were evaluated, including the pollutants covered in the first edition, the additional organic pollutants butadiene, polychlorinated biphenyls and polychlorinated dibenzodioxins/dibenzofurans, and three indoor air pollutants (anthropogenic vitreous fibres, radon and second-hand tobacco smoke). As in the first edition, guidelines were presented in the form of levels/ranges for noncarcinogenic pollutants and as unit risk factors for carcinogenic substances. In contrast, the ozone guideline was formulated as a level and a CRF, whereas the PM guidelines were presented as a CRF alone, this time separately from sulfur dioxide. To aid implementation, a specific chapter was devoted to air quality management and translation of the guidelines into binding standards (WHO Regional Office for Europe, 2000a).

Air quality guidelines – global update 2005. Particulate matter, ozone, nitrogen dioxide and sulfur dioxide (2006)

Global update 2005 provided numerical guideline values for the classical pollutants – PM, ozone, nitrogen dioxide and sulfur dioxide – based on a comprehensive review of all available evidence at the time. Air quality guidelines for PM were presented for the first time, while the nitrogen dioxide levels from previous editions were retained. In addition, the concept of interim targets as “incremental steps in progressive reduction of air pollution” was introduced and used for PM, ozone and sulfur dioxide. Acknowledging that exposure to these pollutants occurs in both outdoor and indoor settings, the guideline levels were meant to apply in all environments, including indoors in households, schools and vehicles. However, the guideline panel recognized the significance of indoor air pollution as a stand-alone risk factor that needed different management approaches to those employed for outdoor air pollution. Therefore, a specific chapter was dedicated to indoor air quality, including a framework for the future development of WHO indoor air quality guidelines (WHO Regional Office for Europe, 2006).

WHO guidelines for indoor air quality: dampness and mould (2009)

This first volume of the series aimed to raise awareness and assist users in identifying and mitigating the health hazards related to biological contamination in all indoor settings. The guidelines included a comprehensive assessment of the evidence on the adverse health effects associated with

dampness and biological agents such as bacteria, mould and fungi. The guideline panel concluded that the most relevant health outcomes of concern were respiratory and immunological, including asthma and allergies. Given a lack of exposure–response relationships, recommendations were formulated as indicators of health risk, such as the persistence of dampness or presence of mould, rather than numerical levels. The guideline panel recommended the prevention/reduction of such indicators on interior surfaces and building structures as an overarching principle that users could follow to manage risks through specific measures (WHO Regional Office for Europe, 2009).

WHO guidelines for indoor air quality: selected pollutants (2010)

The second volume of the series provided recommendations for nine air pollutants either as numerical levels or unit risks, prioritized according to their presence in potentially harmful concentrations indoors and the availability of data for risk assessment. Thus, comprehensive monographs were prepared for benzene, carbon monoxide, formaldehyde, naphthalene, nitrogen dioxide, polycyclic aromatic hydrocarbons, radon, trichloroethylene and tetrachloroethylene. Guidelines for indoor PM were not formulated, since PM had been covered in *Global update 2005*, which was intended for all environments. Although the evidence on indoor nitrogen dioxide was re-evaluated, guideline levels remained the same as before due to a lack of new evidence suggesting a threshold of effect. In addition, some general measures to reduce exposure indoors were proposed, such as controlling sources of emission, ensuring proper ventilation, using low-emission materials and, switching to cleaner fuels and technologies for indoor combustion (WHO Regional Office for Europe, 2010).

WHO guidelines for indoor air quality: household fuel combustion (2014)

Building on previous guidelines for PM and carbon monoxide, modelling and extensive reviews, the latest volume in the series offered recommendations related to household fuel combustion. Using a new WHO guideline development approach, as outlined in the *WHO handbook for guideline development, 2nd edition* (WHO, 2014a), the guideline panel set emission rate targets for PM_{2.5} and carbon monoxide from household fuels combustion, discouraged the use of kerosene and unprocessed coal, and provided guidance for transition to the sustained adoption of clean fuels (e.g. liquefied petroleum gas) and technologies. In addition, risks related to the use of conventional fuels were highlighted, including burns, poisoning, house fires and those related to fuel-wood collection. As overarching principles, the guidelines highlight the importance of reducing outdoor air pollution to achieve indoor air quality guidelines, and of addressing all main household energy end uses to maximize health (WHO, 2014b).

2

Guideline development process

2.1 Introduction

WHO guideline development follows a rigorous process and involves several groups of individuals with well-defined roles, responsibilities and tasks (WHO, 2014a). The process involves the following main steps:

1. formulation of the scope and key questions of the guidelines ([section 2.3](#));
2. systematic review of the relevant evidence ([section 2.4](#));
3. assessment of the certainty level of the body of evidence resulting from systematic reviews ([section 2.4.4](#));
4. formulation of the air quality guideline (AQG) levels ([section 2.5](#)); and
5. formulation of other supporting guidance ([section 2.5.3](#)).

Throughout the whole process, the principles of the GRADE approach were followed (Schünemann et al., 2013).

The WHO steering group was primarily involved in initiating, structuring and executing the guideline development process; the guideline development group (GDG), composed of leading experts and stakeholders, was mainly responsible for determining the scope of the guidelines and formulating AQG levels and other guidance; the systematic review team conducted the systematic reviews of evidence; and the external review group (ERG) provided input and peer review, as needed. The WHO Guidelines Review Committee reviewed and approved the guideline document prior to publication.

The process of developing this update of the air quality guidelines started in 2016. Following WHO procedures, the WHO Regional Office for Europe's European Centre for Environment and Health in Bonn, Germany obtained planning approval and established the WHO steering group, the GDG, the systematic review team and the ERG.

Several meetings of the GDG were held in Bonn throughout the guideline development process. During the first meeting of the GDG in September 2016, GDG members helped define the scope of the guidelines, prioritized air pollutants and critical health outcomes, formulated the key questions to be addressed and set a timeline for completion of the work.

In March 2018 and June 2019, the GDG and the systematic review team met to discuss the preliminary results of the methods adaptation work and systematic reviews of evidence. Revision and publication of the systematic reviews of evidence was completed in mid-2020. In February and June 2020, the GDG finalized the AQG levels and other elements of guidance.

The external consultation of the draft guideline document took place in November and December 2020 through an online survey. In January 2021, the GDG met to address the comments from the external consultation of the draft guideline document.

Throughout the guideline development process, several ad hoc working groups were established to address specific (methodological) issues. Composed of subject matter and methodological experts, these groups worked through remote meetings and contributed within the adapted approaches for systematic review and guideline development to the air quality and health domain.

The following sections describe the groups of experts involved in, and the different steps of, the guideline development process.

2.2 Groups involved in and general procedures of guideline development

The development of WHO guidelines is carried out by several groups of people with defined roles and responsibilities. These are the WHO steering group, the GDG, the systematic review team and the ERG comprising WHO staff members, external experts and stakeholders. In addition, the process was supported by an external guideline methodologist with expertise in systematic review and certainty assessment methods, and other external consultants, including experts in risk of bias (RoB) assessment and environmental epidemiology (shown in [Annex 1](#), Tables A1.1–A1.7).

2.2.1 WHO steering group

The WHO steering group is composed of a limited number of WHO staff with extensive work experience at technical level in the area of air quality and health, who were recruited from all relevant departments, centres and WHO regional offices. Members of this group provided input during the different stages of planning, selection of members of the other groups, reviewing evidence, formulating draft recommendations and guidance, and overseeing peer review. The complete list of members of the WHO steering group can be found in Annex 1, [Table A1.1](#).

2.2.2 Guideline development group

The GDG included subject matter experts who were convened to appraise the evidence and formulate recommendations and related guidance. The group was selected by the WHO steering group, as informed by the results of a survey of WHO expert networks, with the aim to cover the technical skills, perspectives and geographical representation needed in a global guideline development process.

The GDG assisted in determining the scope of the guidelines, chose the critical health outcomes and defined the key review questions. Members of the GDG contributed to drafting the guideline document and responded to peer reviewers. Details of the members of the GDG and their specific roles, affiliations and areas of expertise are listed in Annex 1, [Table A1.2](#).

2.2.3 Methodological working groups

Members of this GDG also worked with the guideline methodologist, a RoB methodologist and other experts in ad hoc working groups to adapt the methods of systematic review and guideline development to the specific field of air quality and health. In particular, the following working groups were formed:

- Working Group on Risk of Bias Assessment;
- Working Group on Certainty of Evidence Assessment;
- Working Group on Derivation of Air Quality Guideline Levels and Interim Targets; and
- Working Groups on Good Practice Statements.

The external methodologists are listed in Annex 1, [Table A1.4](#), and members of the working groups are listed in Annex 1, [Table A1.7](#).

2.2.4 Systematic review team

The systematic review team consisted of experts in environmental and clinical epidemiology, who were commissioned by WHO to conduct the systematic reviews informing the recommendations. The team also provided input into the adaptation of systematic review methods and tools. The GDG and WHO steering group identified the members of the systematic review team based on their publications in the field and their expertise. Members of the systematic review team are listed in Annex 1, [Table A1.3](#).

2.2.5 External review group

The ERG included technical experts and representatives from stakeholders such as patient organizations, environmental advocacy groups, industry associations and scientific societies. Members were identified among networks of excellence, WHO collaborating centres and partner groups such as Cochrane, with support from the GDG and online searches.

Based on several considerations (expertise, sex, geographical representation), about 100 individual experts from 38 countries and territories across all WHO regions were identified and invited to participate in the ERG. Of these, 65 experts provided input at different stages of the guideline development process, as needed.

In particular, they provided information on specific topics, assessed and translated scientific papers, peer-reviewed the evidence base, and/or commented on the draft guideline document. Likewise, an inclusive mapping exercise took place of stakeholder organizations from all WHO regions, working at either regional or global level. Of the 100 identified organizations, 72 were invited to be members of the ERG. Ultimately, 14 organizations participated in the external consultation of the draft guideline document and provided comments that were all addressed by the GDG and WHO steering group. The individual experts and stakeholder organizations are listed in Annex 1, [Table A1.5](#) and [Table A1.6](#), respectively.

2.2.6 Management of conflicts of interest

Conflicts of interest – with or without bias – can undermine the credibility of a guideline; hence, their appropriate management is crucial in WHO guideline development. The members of the GDG as well as the other experts involved in the guideline development process were asked to complete declaration of interest forms. In addition, all experts received briefings about the types of conflicts of interest (financial, intellectual/academic and non-academic). Declarations from all experts were collected and managed according to the relevant WHO procedures. No experts had to be excluded from their respective roles. Further information about the process for identifying, managing and reporting conflicts of interest can be found in the *WHO handbook for guideline development, 2nd edition* (WHO, 2014a). A summary of declared conflicts of interest is presented in [Annex 2](#).

2.2.7 Decision-making during the process

The members of the GDG agreed to make decisions by consensus, through discussions moderated by the appointed GDG co-chairs. In (very rare) cases where consensus was not possible, informal voting was employed. The view of the majority (90% or more of the GDG members, as a result of the discussions to reach agreement in the group) was implemented in developing the guidelines.

Decisions in the ad hoc working groups were made in the same way among the participating GDG members, the external guideline methodologist and/or other external experts. Consensus could not be reached among the GDG members and the methodologist on one aspect of the certainty of evidence assessment. This was about whether upgrades of the evidence certainty should be allowed in case of downgrades: the GDG members thought so, but the methodologist did not. The view of the majority (in this case, the complete GDG) was taken. The methodologist made the first proposals on derivation of AQG levels and interim targets but did not participate in the phase of final formulation of recommendations. The GDG, supported by a technical consultant (Annex 1, [Table A1.7](#)), concluded this work.

The members of the systematic review team conducted the systematic reviews independently, with regular interaction with the working groups and the GDG to ensure that the most important needs of the GDG were addressed appropriately. One member of the systematic review team served as liaison with the GDG and supported the methodological work on AQG levels and interim targets.

2.2.8 Document preparation and external review

The guideline document was drafted in a stepwise manner following the guideline development process. The GDG identified the background and other relevant supporting information early in the preliminary phase. In their second meeting, the WHO steering group and GDG decided on the table of contents, and several of their members started drafting specific sections. At a later stage, a designated technical editor worked towards ensuring consistency and logical flow.

The guideline document went through several rounds of extensive internal and external review. In particular, the external consultation of the draft document was managed through an online survey targeting 71 members of the ERG (48 provided comments and were acknowledged). As prescribed, the procedure focused on the identification of missing data, unclear information, factual errors and issues related to implementation, but not on changing the recommendations. The GDG and WHO steering group considered all comments provided during this external consultation and revised the guideline document where appropriate.

2.3 Determining the scope of the guidelines and formulation of review questions

Determining the scope of the guidelines involved the selection of air pollutants to be considered, as well as the critical health outcomes for each in relation to durations and scale of exposure. This was a multistep procedure in which experts evaluated the strength of the evidence for the pollutants; the causality of pollutant–outcome pairs; and other considerations such as the severity of health outcomes, burden of disease, expected increases in exposure and policy considerations.

The present guidelines are applicable to both outdoor and indoor environments. Thus, they cover all settings in which people spend a significant portion of their time. This has been the case since the publication of the first edition of the guidelines in 1987 and was reinforced in *Global update 2005* and the 2010 guidelines for indoor air quality.

It is important to note that AQG levels recommended in previous WHO air quality guidelines for pollutants or averaging times not re-evaluated in this update

remain valid, including those for the short averaging times for nitrogen dioxide, sulfur dioxide and carbon monoxide included in the 2005 *Global update* and indoor air quality guidelines from 2010. The reader is referred to previous volumes of air quality guidelines (WHO Regional Office for Europe, 1987, 2000a, 2010) for the other pollutants not covered in this update. As in previous volumes, the guidelines do not cover occupational settings, due to the specific characteristics of the relevant exposures and the potential differences in population susceptibility of the adult workforce in comparison with the general population.

Furthermore, the guidelines do not include recommendations about any kind of multiple exposures. In everyday life, people are often exposed to a mixture of air pollutants at the same time. WHO acknowledges the need to develop comprehensive models to quantify the effects of multiple exposures on human health. However, as the main body of evidence on air quality and health still focuses on the impact of single air pollutants on health outcomes, the current guidelines provide recommendations for each air pollutant individually.

The GDG also decided not to formulate specific recommendations on population-wide interventions because these are largely context specific: what might be effective in one setting might not work in another. Instead, general risk management principles, based on decades of experience, are summarized in [Chapter 6](#), on implementation of the guidelines. In addition, individual-level interventions, such as the use of personal respiratory protection (e.g. masks, respirators), air purifiers and behavioural measures, are not addressed here but in a report from a separate WHO consultation (WHO, 2020a).

2.3.1 Preliminary consultation

Following the conclusions from the Review of evidence on health aspects of air pollution (REVIHAAP) project (WHO Regional Office for Europe, 2013a), WHO organized an expert consultation in Bonn in September–October 2015 as a first step for this update of the air quality guidelines. The objective was to gather expert opinion and guidance in order to identify and discuss the latest available evidence on health effects of air pollutants and interventions to reduce exposure to air pollution for the purpose of informing this update of the air quality guidelines.

Twenty-eight participants – representing a wide array of expertise and geographical locations – attended the consultation, which included not only a review of the available scientific evidence on a number of ambient air pollutants but also methodological issues and the implications of exposure and intervention studies. Experts recommended that a focus of these guidelines on pollutant-

specific risk assessment was still appropriate and prioritized 32 air pollutants according to four categories to reflect their relative importance in the context of updating the air quality guidelines. Since reviewing the evidence systematically for all air pollutants was infeasible considering the available resources, experts suggested prioritizing the pollutants PM_{2.5}, PM₁₀, ozone, nitrogen dioxide, sulfur dioxide and carbon monoxide for this update. This advice was based on the large body of new health-related evidence that had been published since *Global update 2005* (WHO Regional Office for Europe, 2016a).

2.3.2 Selection of priority pollutants

The final selection of air pollutants took place in September 2016, during the first meeting of the GDG. Prior to the meeting, WHO surveyed GDG members on the final list of air pollutants to be included in this update of the air quality guidelines. The air pollutants identified in the global expert consultation, together with a number of different health outcomes, were included in the survey. In the ensuing discussion, the GDG decided to develop AQG levels for PM₁₀, PM_{2.5}, nitrogen dioxide, ozone, sulfur dioxide and carbon monoxide in relation to health outcomes critical for decision-making and for relevant averaging times.

For PM types such as BC/EC and UFP, the GDG agreed that AQG levels for these pollutants could not be formulated owing to the absence of clear quantitative evidence on independent health effects. However, the results of reviews of evidence conducted by other groups would be examined in order to reach a better-informed decision about whether recommendations should be formulated and in which form. Likewise, for SDS, the GDG agreed that any recommendation would likely be qualitative in nature and geared towards guiding countries in moving towards mitigation and adaptation measures.

Following presentations by invited experts, the GDG, at its third meeting, decided to include guidance on BC/EC, UFP and SDS in the form of good practice statements.

2.3.3 Prioritization of health outcomes

In order to define the pollutant–outcome pairs that would be systematically reviewed to inform the formulation of AQG levels, the GDG developed a prioritization framework based on the considerations outlined in [Box 2.1](#).

Box 2.1. Health outcome prioritization framework

- Evidence on causality for a health outcome would be considered first, according to the latest determination (causal or likely causal) from the Health Canada, the International Agency for Research on Cancer, US EPA or other available integrated science assessments. As mandated by the Clean Air Act, the US EPA periodically reviews all scientific evidence about the health effects of so-called criteria pollutants, including all five pollutants considered in this report. These Integrated Science Assessments (ISAs) include a structured analysis of all evidence – including from toxicology – that supports a classification of a specific effect being causal, likely causal, suggestive of a causal relationship, inadequate to infer a causal relationship or not likely to be a causal relationship. For details, see Owens et al. (2017). These classifications have been used in support of identification of the relevant pollutant–outcome associations addressed in this document.
- Where causality is not determined to be proven or likely (e.g. suggestive causality), the precautionary principle would be used when determining which additional most-severe health outcomes could be included. These outcomes would be based on other considerations such as contribution to burden of disease (e.g. prevalence of disease, disability weight), policy implications and expected increase in exposure to a pollutant in the future.
- Causality determination would supersede the severity of a health outcome but, in some cases, two (or more) different health outcomes might be systematically evaluated for the same pollutant (e.g. one with a causal or likely causal link to the pollutant, and another health outcome for which the evidence is suggestive only but which is very severe or prevalent in the population). Severity of disease would be informed by considerations proposed by the latest update of the joint European Respiratory Society and American Thoracic Society policy statement on health effects from air pollution (fatality, persistence of effect, susceptible groups and medical/functional significance, including loss of autonomy and reduced quality of life) (Thurston et al., 2017).
- Lastly, as health outcomes can be assessed in various ways in studies, the specific health outcome measure(s) would be identified, based on evidence and the expert judgement of the GDG, to be used for quantitative health risk assessment in the guidelines.

By applying the prioritization framework, the GDG identified the following critical health outcomes associated with the selected air pollutants:

- all-cause (non-accidental) mortality⁴ (hereafter referred to as all-cause mortality);
- cause-specific mortality, as per the International Statistical Classification of Diseases and Related Health Problems, 10th edition (ICD-10), 2016 version (WHO, 2016b): cardiovascular (ICD-10 codes I00–I99), lung cancer (ICD-10 codes C30–C39) and respiratory (ICD-10 codes J00–J99);
- hospital admissions and emergency room visits related to asthma (ICD-10 code J45); and
- hospital admissions and emergency room visits related to IHD (ICD-10 codes I20–I25; ultimately restricted to myocardial infarction, ICD-10 codes I21–I22).⁵

The pollutant–outcome pairs that were included comprise those for which there is broad scientific consensus regarding the causal nature of the reported relationships; others were chosen based on the strength of the epidemiological evidence.

Table 2.1 gives an overview of the different considerations included in the prioritization process. This table was adapted from the document resulting from the population, exposure, comparator, outcome and study design (PECOS) process (see section 2.3.4) finalized in November 2016.

Since then, it was decided to include one more pollutant–outcome pair: the association between exposure to short-term sulfur dioxide and all-cause and respiratory mortality. This was done to provide continuity with *Global update 2005*, as explained below.

The GDG recognizes that associations for many more pollutant–outcome pairs have been reported and reviewed in the literature. It would be practically impossible, given the resources available for the current guideline update, to include all of these for consideration and review.

⁴ In an epidemiological study of air quality and health, all-cause mortality (ICD-10 code A00–Z99) refers to all deaths, and non-accidental mortality (ICD-10 code A00–R99) includes all deaths with the exception of deaths due to accidents, murder, suicide, etc. Although all-cause mortality includes accidental deaths, the proportion of deaths caused by accidents, etc. is typically small (< 10%) in comparison with the other causes of death.

⁵ The systematic review by Lee et al. (2020) focused on myocardial infarction (ICD-10 codes I21–I22) as the only IHD outcome because it is not possible to establish the precise time of onset of other IHD outcomes. Further, other conditions within the spectrum of IHD are routinely managed in outpatient settings rather than in the emergency room/department or hospital wards.

The GDG, however, sees no grounds for assuming that the AQG levels, as derived in [Chapter 3](#), would be very different if more outcomes would have been considered for the pollutants that were included. Obviously, this does not apply to those pollutants that were not considered at all in the current update. A draft version of this document has been reviewed by a large number of experts and stakeholder organizations, and no examples have been provided that would change this assessment.

The GDG also emphasizes that there has been no separate, independent assessment of the mechanistic, toxicological and human clinical studies relating air pollution to human health. However, comprehensive evaluations by authoritative bodies such the Committee on the Medical Effects of Air Pollutants (COMEAP) in the United Kingdom, Health Canada and the US EPA were taken into account in the development of the AQG levels.

Information about all the specific pollutant–outcome pairs reviewed can be found in the systematic reviews of evidence available in a virtual special issue of *Environment International* entitled *Update of the WHO global air quality guidelines: systematic reviews* (Whaley et al., 2021).

Table 2.1. Air pollutants and health outcomes proposed for systematic review in the guideline development process^a

LONG-TERM EXPOSURE			
Pollutant	Health outcomes used in <i>Global update 2005</i>	Health outcomes selected for updating in the 2021 air quality guidelines	Justification for health outcome selection
PM _{2.5} and PM ₁₀	Total, cardiopulmonary and lung cancer mortality	• All-cause mortality • Cardiovascular mortality (all, cerebrovascular, IHD) • Respiratory mortality (any, COPD, acute lower respiratory infections) • Lung cancer mortality	CAUSALITY DETERMINATION (REFERENCE)
			PM_{2.5}
			• Causal for cardiovascular and respiratory mortality (US EPA, 2009) • Causal for total and cardiovascular mortality (Health Canada, 2013)
			PM
			• Causal for total mortality in relation to PM (Health Canada, 2013) • Group 1^b lung cancer for PM (Straif et al., 2013) • Likely causal for lung cancer mortality in relation to PM (Health Canada, 2013)
	SUPPORTING CONSIDERATIONS		
	PM₁₀		• Health outcome supported by evidence from PM₁₀ and PM_{2.5}
	OTHER RELEVANT CAUSAL DETERMINATIONS (REFERENCE)		
	PM_{2.5}		• Likely causal for respiratory effects (US EPA, 2009) • Likely causal for respiratory effects (Health Canada, 2013)

Table 2.1 contd

LONG-TERM EXPOSURE			
Pollutant	Health outcomes used in <i>Global update 2005</i>	Health outcomes selected for updating in the 2021 air quality guidelines	Justification for health outcome selection
O ₃	No long-term guideline provided	• All-cause mortality • Respiratory mortality	CAUSALITY DETERMINATION (REFERENCE) • Suggestive causality for mortality (US EPA, 2013) • Suggestive causality for respiratory mortality (Health Canada, 2013) SUPPORTING CONSIDERATIONS • Severity of health outcome, burden of disease • Precautionary principle from expected increase of this pollutant due to climate change (policy implications and end-user perspectives) OTHER RELEVANT CAUSAL DETERMINATIONS (REFERENCE) • Likely causal for respiratory effects (US EPA, 2013) • Suggestive causality for respiratory effects (Health Canada, 2013)

Table 2.1 contd

LONG-TERM EXPOSURE			
Pollutant	Health outcomes used in <i>Global update 2005</i>	Health outcomes selected for updating in the 2021 air quality guidelines	Justification for health outcome selection
NO ₂	Respiratory effects in children	• All-cause mortality • Respiratory mortality	CAUSALITY DETERMINATION (REFERENCE) • Suggestive causality for total mortality (US EPA, 2016) • Suggestive causality for total mortality (Health Canada, 2016a))
			SUPPORTING CONSIDERATIONS • Severity of health outcome, burden of disease • Recent studies show associations with respiratory mortality, consistent with likely causality for respiratory effects (see other causal determinations below) • The causal determination of US EPA for mortality is suggestive, in the light of the limited number of studies properly addressing confounding by other transport-related air pollutants • The causal determination of US EPA of likely causal for respiratory effects (see other causal determinations below) takes into account respiratory mortality
			OTHER RELEVANT CAUSAL DETERMINATIONS (REFERENCE) • Likely causal for respiratory effects (US EPA, 2016) • Likely causal for respiratory effects (Health Canada, 2016a)

Table 2.1 contd

SHORT-TERM EXPOSURE			
Pollutant	Health outcomes used in <i>Global update 2005</i>	Health outcomes selected for updating in the 2021 air quality guidelines	Justification for health outcome selection
CO	COHb levels of below 2% in nonsmokers' blood (also protective for long-term exposure) (WHO Regional Office for Europe, 2000a, 2010)	• Hospital admissions and emergency room visits related to IHD ^c	CAUSALITY DETERMINATION (REFERENCE)
			• Likely causal for cardiovascular effects (US EPA, 2010)
			• Likely causal for cardiovascular effects (Health Canada, 2010)
			SUPPORTING CONSIDERATIONS
			• Consistent with WHO indoor air quality guidelines (WHO Regional Office for Europe, 2010) using COHb levels linked to IHD ^c -related symptoms (e.g. ST-segment changes, reduced time to exercise-induced angina)
			• IHD ^c appears to be the most consistent outcome from cardiovascular effects associated with short-term CO exposure in epidemiological studies (US EPA, 2010)
PM _{2.5} and PM ₁₀	COHb levels of below 2% in nonsmokers' blood (also protective for long-term exposure) (WHO Regional Office for Europe, 2000a, 2010)	• All-cause mortality • Cardiovascular mortality • Respiratory mortality	CAUSALITY DETERMINATION (REFERENCE)
			PM_{2.5}
			• Causal for all-cause, cardiovascular and respiratory mortality (US EPA, 2009)
			• Causal for all-cause, respiratory and cardiovascular mortality (Health Canada, 2013)

Table 2.1 contd

SHORT-TERM EXPOSURE		
Pollutant	Health outcomes used in <i>Global update 2005</i>	Health outcomes selected for updating in the 2021 air quality guidelines
PM _{2.5} and PM ₁₀ (contd)		Justification for health outcome selection
		PM (any size fraction) • Causal for all-cause mortality (Health Canada, 2013)
		SUPPORTING CONSIDERATIONS • Cardiovascular and respiratory mortality also considered in causal determination of respiratory/cardiovascular effects (US EPA, 2009) (see other relevant causal determinations) • PM₁₀, supported by evidence from PM_{2.5}
		OTHER RELEVANT CAUSAL DETERMINATIONS (PM_{2.5}) (REFERENCE) • Likely causal for respiratory effects (US EPA, 2009) • Causal for cardiovascular effects (US EPA, 2009) • Causal for respiratory effects (Health Canada, 2013) • Causal for cardiovascular effects (Health Canada, 2013)

Table 2.1 contd

SHORT-TERM EXPOSURE			
Pollutant	Health outcomes used in <i>Global update 2005</i>	Health outcomes selected for updating in the 2021 air quality guidelines	Justification for health outcome selection
O ₃	Daily mortality	• Hospital admissions and emergency room visits related to asthma • All-cause mortality	CAUSALITY DETERMINATION (REFERENCE) • Causal for respiratory effects (US EPA, 2013) • Causal for respiratory effects (Health Canada, 2013) • Likely causal for total mortality (US EPA, 2013) • Likely causal for total mortality (Health Canada, 2013) SUPPORTING CONSIDERATIONS • Stronger causal determination for respiratory effects than for mortality outcomes (see other relevant causal determinations below) • Experimental studies demonstrate decreases in lung function at exposures as low as 60–70 ppb O₃ in young healthy adults. Equally strong evidence demonstrates associations of ambient O₃ with asthma hospital admissions and emergency room visits, including for at-risk subpopulations (e.g. children, people with asthma or other pre-existing diseases), who cannot ethically be included in experimental studies

Table 2.1 contd

SHORT-TERM EXPOSURE			
Pollutant	Health outcomes used in <i>Global update 2005</i>	Health outcomes selected for updating in the 2021 air quality guidelines	Justification for health outcome selection
O ₃ (contd)			• Mortality also included because of the severity of health outcome, number of exposed individuals and precautionary principle (expected future increase of this pollutant due to climate change, with policy and end-user implications). On the other hand, studies on mortality might target other subgroups of the population such as older people
OTHER RELEVANT CAUSAL DETERMINATIONS (REFERENCE)			
• Likely causal for cardiovascular effects (US EPA, 2013) • Suggestive causality for cardiovascular effects (Health Canada, 2013) • Likely causal for cardiopulmonary mortality (Health Canada, 2013)			

Table 2.1 contd

SHORT-TERM EXPOSURE			Justification for health outcome selection
Pollutant	Health outcomes used in <i>Global update 2005</i>	Health outcomes selected for updating in the 2021 air quality guidelines	
NO ₂	Bronchial responsiveness in asthmatics	• Hospital admissions and emergency room visits related to asthma • All-cause mortality	CAUSALITY DETERMINATION (REFERENCE) • Causal for respiratory effects (US EPA, 2016) • Causal for respiratory effects (Health Canada, 2016a) • Likely causal for total mortality (Health Canada, 2016a) • Suggestive causality for total mortality (US EPA, 2016) SUPPORTING CONSIDERATIONS • Stronger causal determination for respiratory effects than for mortality outcomes (see other relevant causal determinations below) • Both Health Canada and US EPA causality assessments for respiratory effects are very recent (2016); limited new evidence might have accumulated since

Table 2.1 contd

SHORT-TERM EXPOSURE			
Pollutant	Health outcomes used in <i>Global update 2005</i>	Health outcomes selected for updating in the 2021 air quality guidelines	Justification for health outcome selection
NO ₂ (contd)			• Strongest evidence for relationships of short-term NO₂ exposure with respiratory effects is for asthma exacerbations. More uncertainty exists with independent effect of short-term NO₂ exposure on non-asthma respiratory effects due to less consistent evidence across scientific disciplines and limited evidence to support biological plausibility. Additionally, studies of short-term NO₂ exposure with asthma hospital admissions and emergency room visits include at-risk subpopulations (e.g. children, people with asthma or other pre-existing diseases) who cannot ethically be included in experimental studies • Mortality is also included because of severity of health outcome and number of exposed individuals. On the other hand, studies on mortality might target other subgroups of the population such as older people

Table 2.1 contd

SHORT-TERM EXPOSURE			
Pollutant	Health outcomes used in <i>Global update 2005</i>	Health outcomes selected for updating in the 2021 air quality guidelines	Justification for health outcome selection
SO ₂	All-age mortality and childhood respiratory disease	• Hospital admissions and emergency room visits related to asthma • All-cause mortality^d • Respiratory mortality^d	CAUSALITY DETERMINATION (REFERENCE) • Causal for respiratory effects (US EPA, 2015) • Causal in adults for respiratory effects (Health Canada, 2016b) SUPPORTING CONSIDERATIONS • Experimental studies demonstrate lung function decrements and respiratory symptoms at very short-term exposures (i.e. 5–10 minutes) to SO₂ in adults with asthma. Studies of hospital admissions and emergency room visits related to asthma, including evidence from at-risk subpopulations (e.g. children, people with asthma or other pre-existing diseases) who cannot ethically be included in experimental studies, report positive associations with short-term SO₂ exposures, particularly for children

COHb: carboxyhaemoglobin; ppb: parts per billion.

^a This table was adapted from the document resulting from the PECOS process (see [section 2.3.4](#)) finalized in November 2016.

^b Group 1 means carcinogenic to humans.

^c Ultimately restricted to myocardial infarction (ICD-10 codes I21 and I22).

^d In the second GDG meeting held in 2018, the GDG agreed to include the pollutant-outcome pairs SO₂-all-cause mortality and SO₂-respiratory mortality.

In the second GDG meeting in March 2018, the question was raised of why a systematic review of short-term associations between sulfur dioxide and all-cause mortality would be needed in addition to the reviews on PM, ozone and nitrogen dioxide. It was suggested to also review SO₂ and mortality to ensure continuity with the previous *Global update 2005*. If a new review is not feasible, the GDG suggested formulating clear justification as to why mortality attributed to SO₂ is not considered. It was subsequently decided to commission a separate review on short-term associations between sulfur dioxide and all-cause as well as respiratory mortality, to be conducted by the team that had already conducted the review on short-term associations of PM, ozone and nitrogen dioxide with mortality. Therefore, the latest US EPA ISA of causality for this association is mentioned below.

In 2017 the latest US EPA ISA on sulfur oxides was published (US EPA, 2017). This did not change the assessment noted in [Table 2.1](#) of a causal relationship between short-term sulfur dioxide concentrations and respiratory effects. The association between short-term sulfur dioxide concentrations and total mortality was deemed to be suggestive of a causal relationship. This association was not considered by the GDG in 2016, but was added at a later stage, as previously mentioned. Therefore, the causality assessment is not reported in [Table 2.1](#).

The US EPA published an updated ISA of PM in 2019 (US EPA, 2019a). The causality determinations for long- and short-term PM effects on mortality and on respiratory and cardiovascular morbidity were the same as those in the 2009 ISA (US EPA, 2009), which was quoted in [Table 2.1](#).

The US EPA published an updated ISA of ozone in 2020 (US EPA, 2020). The causality determinations for ozone effects on respiratory morbidity were the same as those in the 2009 ISA, which was quoted in [Table 2.1](#). For short-term mortality and for cardiovascular morbidity, the evidence was changed from likely causal to suggestive of a causal relationship, in part because new human exposure studies such as the Multicenter Ozone Study in older Subjects (MOSES) study did not clearly demonstrate the cardiovascular effects of ozone (Frampton et al., 2017; Rich et al., 2020). Other reasons quoted were the lack of control for co-pollutants in epidemiological studies and uncertainty about the short-term effects of ozone on cardiovascular emergency room and hospital admissions.

Although the US EPA ISA was published in 2020, the literature on long- and short-term effects of ozone has grown since then. [Chapter 3](#) includes five new studies on the long-term effects of ozone on mortality and one very large

worldwide multicity study on the short-term effects of ozone on mortality, which have provided further evidence of the short- and long-term effects on total and respiratory mortality. The reader is referred to [section 3.4](#) for further details.

The nitrogen dioxide causality assessments shown in Table 2.1 are based on reviews published in 2016. Since then, COMEAP published a report in 2018 entitled *Associations of long-term average concentrations of nitrogen dioxide with mortality*, which is somewhat more supportive of a causal role of long-term nitrogen dioxide in increasing all non-accidental mortality, especially respiratory mortality (PHE, 2018). The 2016 EPA ISA classified the relationship between short-term nitrogen dioxide and respiratory effects as causal and the relationship between long-term nitrogen dioxide and respiratory effects as likely causal. A footnote to the causality determination defined the health outcome as “[a]n array of outcomes is evaluated as part of a broad health effects category: physiological measures (e.g. airway responsiveness), clinical outcomes (e.g. hospital admissions), cause-specific mortality”. This suggests the causality determinations also extend to respiratory mortality, although the further detailed assessments in the ISA provide some qualifications for the separate health effects that were evaluated (US EPA, 2016). A 2018 review by the German Environment Agency (in German, with a summary in English) also supports a role for long-term nitrogen dioxide in causing cardiovascular mortality (Schneider et al., 2018).

The GDG notes that one review has specifically investigated how sensitive the associations between long-term nitrogen dioxide concentrations and mortality were to adjustment for different PM metrics (Faustini, Rapp & Forastiere, 2014). Associations with nitrogen dioxide were found to be generally robust.

Since 2016, no authoritative reviews have been published on short-term associations between carbon monoxide and hospital admissions for myocardial infarction.

2.3.4 Formulation of review questions

As per the WHO procedure of developing guidelines (WHO, 2014a), key questions to guide the review of evidence are best developed using the population, intervention, comparator and outcome format.

However, in environmental health risk guidelines such as the WHO air quality guidelines, recommendations are typically given in the form of numerical concentration values to prevent adverse health effects from exposure to environmental pollutants (so-called AQG levels). Typically, the best available evidence from human studies in this field consists mostly, if not exclusively,

of observational studies, as opposed to (randomized) controlled trials. Therefore – and as raised by several expert guideline development methodologists dealing with environmental risk guidelines (Collaboration for Environmental Evidence, 2013) – the use of a slightly adapted formulation of the traditional population, intervention, comparator and outcome question was used: a PECOS question. The intervention (I) term was replaced by an exposure (E) term, reflecting the concentration in ambient air of the particular air pollutant under consideration; also, an S was added to define study designs to be considered in evaluating the evidence, resulting in a PECOS question: population, exposure, comparator, outcome and study design (Table 2.2).

The GDG proposed the following PECOS questions (Box 2.2), which were later adapted to the health outcome and specific type of studies relevant for each pollutant and time average (short- or long-term exposure) considered in the updated guidelines.

Box 2.2. Generic PECOS question for long- and short-term exposures

Long-term exposures

In any population, including subgroups of susceptible adults and children (P), what is the increase in risk of health outcome x (O) per unit increase (C) in $\mu\text{g}/\text{m}^3$ of long-term exposure (in the order of months to years) to ambient^a concentration of air pollutant y (E), observed in studies relevant for the health outcome and exposure duration of interest (S)? In these studies, what is the lowest concentration that produces a measurable increase in risk?

Short-term exposures

In any population, including subgroups of susceptible adults and children (P), what is the increase in risk of health outcome x (O) per unit increase (C) in $\mu\text{g}/\text{m}^3$ of short-term exposure (in the order of hours to days) to ambient concentration of air pollutant y (E), observed in studies relevant for the health outcome and exposure duration of interest (S)? In these studies, what is the lowest concentration that produces a measurable increase in risk?

^a Ambient refers to both outdoor and indoor environments.

These PECOS questions were designed to retrieve the epidemiological evidence necessary to develop updated AQG levels and inform the shape of the CRF for the different pollutant–outcome pairs.

For the specific purpose of updating the WHO air quality guidelines, the PECOS terms were defined as follows ([Table 2.2](#)).

Table 2.2. Elements of a PECOS question

Element	Explanation
Population	The general population, all age groups, from developed and developing countries, living both in urban and in rural areas exposed on a daily basis to the pollutant of interest through ambient air (understood as encompassing exposure in both outdoor and indoor environments), and not exclusively in occupational settings or as a result of indoor exposure alone. Population subgroups that are vulnerable to the effects of air pollution would be included, such as those with specific pre-existing health conditions (e.g. respiratory or cardiovascular diseases), pregnant women, newborns, children or older people. Whenever applicable, the considered health effect of exposure to the pollutant of interest in these vulnerable subgroups of the population would be assessed separately
Exposure	Exposure to air pollutants from any source, measured as long term (months to years) or short term (hours to days)
Comparator	Exposure to the lowest levels of air pollutants from any source, measured as long- (months to years) or short-term (hours to days)
Outcome	Health outcome(s) upon which the AQG levels are developed for each air pollutant considered in the guidelines
Study design	Type of studies evaluated, such as cohort and case–control studies (long term) and time-series, case-crossover and panel studies (short term)

2.4 Systematic review of the evidence

To address the PECOS questions posed by the GDG, a preliminary search of the relevant literature was conducted to identify available systematic reviews and meta-analyses on air quality and health. Based on an overview that assessed the quality of reviews in the field (Sheehan et al., 2016), it was decided that

included peer-reviewed articles that were of sufficient quality and addressed the formulated PECOS would serve as a starting point for most systematic reviews. Missing elements (e.g. specific assessments or syntheses) would be extracted anew and searches updated to the latest possible date within the process.

Selected members of the systematic review team, who were mostly identified through the above procedure, reviewed and synthesized all the relevant epidemiological literature in the area of air quality and health, following the principles outlined in the *WHO handbook for guideline development, 2nd edition* (WHO, 2014a) and guidance provided from methodologists and experts in the discipline.

The instruments needed to assess the RoB for individual studies included in the reviews and the overall certainty of evidence across studies were adapted to better reflect the particularities of the air quality and health field.

PECOS questions were formulated for each of the major pollutant–outcome pairs and relevant averaging times. When the same health outcomes and averaging times were assessed, various air pollutants were grouped under the same systematic review, resulting in six systematic reviews.

All systematic reviews followed a common protocol prepared according to the provisions set out by the *WHO handbook for guideline development, 2nd edition* (WHO, 2014a) and later fine-tuned in relation to the specific exposure–outcome averaging time combinations that each review aimed to address.

The protocols for each systematic review are registered on PROSPERO, an international register of systematic reviews, maintained by the University of York (NIHR, 2021).

Furthermore, all systematic reviews used in the derivation of AQG levels are publicly available in a special issue of the journal *Environment International* (Whaley et al., 2021), which are also summarized in [Annex 3](#).

- Long-term exposure to PM and all-cause and cause-specific mortality: a systematic review and meta-analysis (Chen & Hoek, 2020).
- Long-term exposure to NO₂ and O₃ and all-cause and respiratory mortality: a systematic review and meta-analysis (Huangfu & Atkinson, 2020).
- Short-term exposure to particulate matter (PM₁₀ and PM_{2.5}), nitrogen dioxide (NO₂), and ozone (O₃) and all-cause and cause-specific mortality: systematic review and meta-analysis (Orellano et al., 2020).

- Short-term exposure to sulfur dioxide (SO₂) and all-cause and respiratory mortality: a systematic review and meta-analysis (Orellano, Reynoso & Quaranta, 2021).
- Short-term exposure to ozone, nitrogen dioxide, and sulfur dioxide and emergency room visits and hospital admissions due to asthma: a systematic review and meta-analysis (Zheng et al., 2021).
- Short-term exposure to carbon monoxide and myocardial infarction: a systematic review and meta-analysis (Lee et al., 2020).

The core systematic reviews of adverse health effects were commissioned to address the PECOS questions. To ensure and confirm that no relevant studies in indoor settings had been missed in these reviews (none had been identified in the searches, based on the selected pollutant–outcome pairs), complementary searches were also conducted. In addition, several reviews and analyses were conducted in the context of this update of the guidelines. These included work on the health effects of exposure to particles originating from SDS, the burden of disease attributable to air pollution, the effectiveness of individual-level interventions and the cost–effectiveness of air quality interventions. Relevant review work conducted by other groups was closely monitored (e.g. on the health effects of BC/EC and UFP).⁶

2.4.1 Identification and retrieval of evidence

Based on the PECOS questions, a list of inclusion and exclusion criteria were defined for each systematic review and later fine-tuned by the systematic review team (Table 2.3 and Table 2.4).

Because most of the systematic reviews were based on peer-reviewed papers, the original search strategies were revised to reflect any additional eligibility criteria to ensure that all papers addressing the PECOS questions were identified.

Specific search strategies using both free text and controlled vocabulary terms were run for each database. More details can be found in the systematic reviews published in *Environment International* (Whaley et al., 2021).

All efforts were made to include all relevant papers published, which entailed searching a considerable number of literature sources, the inclusion of papers in languages other than English and the use of time frames spanning from database inception to late 2018.

⁶ With the exception of the review and analysis performed by Evangelopoulos et al. (2020) and the review by Fussell & Kelly (2021), none of the reviews conducted in the context of the update of the guidelines have yet been published.

Table 2.3. Generic eligibility criteria for systematic reviews of long-term exposures

PECOS	Inclusion	Exclusion
Population	- General human population (including subgroups at risk: children, pregnant women, older people and patients with particular conditions) of all ages, living in developed and developing areas, both urban and rural. No geographical restrictions - Exposure to the pollutant of interest predominantly via inhalation through ambient air (this covers exposures in both outdoor and indoor environments)	Exposure to the pollutant of interest in occupational settings or as a result of indoor exposure exclusively
Exposure	- Long-term exposure (in the order of months to years) to ambient air PM_{2.5}, PM₁₀, O₃ and NO₂ expressed in a concentration unit (µg/m³, ppb) - For the NO₂ systematic review, NO_x studies may be included	- Less than 1 year of data available - No exclusion criteria applied based on adjustment for co-pollutants
Comparator	Exposure to lowest levels of the air pollutant of interest in the same or a control population	–
Outcome	Health outcomes selected in relation to long-term exposure include (ICD-10 codes (version 2016)): all-cause mortality and cause-specific mortality, including cardiovascular (I00–I99), lung cancer (C30–C39) and respiratory (J00–J99)	–
Study design	- Human epidemiological studies such as: - prospective and retrospective studies - cohort studies - case-control and nested case-control studies - Published (or accepted for publication, i.e. in press) studies in peer-reviewed indexed journals in any language (abstract in English language) and grey literature, where relevant.	- Qualitative studies - Studies without individual-level data, that is, fully group-level (ecological) covariates - Studies where no original data were analysed - Reviews and methodological papers - Non-human studies (in vivo, in vitro, other)

ppb: parts per billion.

Table 2.4. Generic eligibility criteria for systematic reviews of short-term exposures

PECOS	Inclusion	Exclusion
Population	- General human population (including subgroups at risk: children, pregnant women, older people, and patients with particular conditions) of all ages, living in developed and developing areas, both urban and rural. No geographical restrictions - Exposure to the pollutant of interest predominantly via inhalation through ambient air (this covers exposures in both outdoor and indoor environments)	Exposure to the pollutant of interest in occupational settings or as a result of indoor exposure exclusively
Exposure	- Short-term exposure (in the order of hours to 7 days) to ambient air PM_{2.5}, PM₁₀, O₃, NO₂, SO₂ and CO, from any source expressed in a concentration unit (µg/m³, ppb) - For NO₂ systematic review, NO_x studies may be included	No exclusion criteria were applied based on adjustment for co-pollutants
Comparator	Exposure to lowest levels of the air pollutant of interest in the same or a control population	–
Outcome	Health outcomes selected for short-term exposure include (ICD-10 codes (version 2016)): all-cause mortality and cause-specific mortality, including cardiovascular (I00–I99) and respiratory (J00–J99), and hospital admissions and emergency room visits related to asthma (J45–J46) and myocardial infarction (I21–I22)	–
Study design	- Human epidemiological studies such as: - time-series studies - case-crossover studies - panel studies - Published (or accepted for publication, i.e. in press) studies in peer-reviewed indexed journals in any language (abstract in English language) and grey literature, where relevant	- Qualitative studies - Studies without individual-level data, that is, fully group-level (ecological) covariates - Reviews and methodological papers - Non-human studies (in vivo, in vitro, other) - Studies with geographical and temporal overlap during meta-analysis

ppb: parts per billion.

For each of the systematic reviews, two reviewers independently screened titles and abstracts of papers identified with the systematic search and identified those that could be excluded based on the eligibility criteria. The full texts of the remaining articles were independently reassessed by two reviewers to ensure that all eligibility criteria were met. Disagreements among reviewers were resolved by discussion or through consultation with a third reviewer. The reasons for excluding articles were recorded. In addition, references of identified relevant articles (and reviews/guidelines, where relevant) were scanned to identify additional papers matching the PECOS question. The resulting list of papers was circulated with the systematic review team and the GDG to identify any potentially relevant missing studies (published or in press). Lastly, papers identified through the peer review process were incorporated as appropriate, either quantitatively or qualitatively, as feasible.

Two reviewers extracted all relevant data needed for the process using pre-defined forms. Key data included the elements defined by PECOS and declared conflicts of interest, as well as the data necessary to conduct RoB assessments (e.g. confounding factors) and to derive the AQG levels (i.e. for onset of the CRF: 5th–95th percentiles of population exposure, mean/median, and minimum and maximum pollutant concentrations; for shape of the CRF: methods and results of authors' assessments). Where necessary data were missing, the systematic review team obtained them from the authors of the primary studies or calculated them.

2.4.2 RoB assessment of individual studies

To assess RoB for individual studies, a specific instrument was developed by a working group composed of GDG members and methodologists. Based on a review of existing tools, the group agreed to take into account six key domains (confounding, selection bias, exposure assessment, outcome measurement, missing data, selective reporting), each including several subdomains or signalling questions. Judgement options included high, moderate and low RoB. The group also prepared guidance notes to assist the systematic review team in performing the task, including a list of critical and additional potential confounders to consider when making judgements about confounding and key information on the particularities of exposure assessment in the field. To avoid carrying forward the ratings from one domain to the others, the working group considered that an overall judgement of bias at the study level was not appropriate: instead, subgroup analyses were suggested per RoB domain across studies. This approach was considered more suited to identify which particular type of bias had an impact on the pooled effect size, as well as its direction and magnitude (Morgan et al., 2019).

A detailed description of the instrument is available in a dedicated publication (WHO Regional Office for Europe, 2020).

2.4.3 Synthesis of evidence

Meta-analyses were conducted to obtain summary pooled estimates of the risk for an adverse health outcome per unit increase in exposure to a given air pollutant. When three or more studies were identified for the same pollutant and health outcome, a quantitative synthesis was performed. Otherwise, the effect estimates were described qualitatively. Overall, statistical analyses were performed according to the *Guidelines for application of meta-analysis in environmental epidemiology* (Blair et al., 1995), the *Cochrane handbook for systematic reviews of interventions* (Higgins & Green, 2011) or other authoritative guidance. The approach used was the inverse variance method, assuming a linear concentration–response relationship.

When exposure metrics differed among studies, the data were transformed to the same metric, generally the relative risk (RR).

Although no dose–response meta-analytic techniques were employed to assess the shape of the CRF, potential deviations from linearity were assessed by other means, for example, by stratifying by mean pollutant concentrations or qualitatively evaluating the determinations and judgements made by study authors.

Because of differences in populations and pollution composition across populations, it was decided that estimates were to be pooled by means of a random-effects meta-analysis (maximum likelihood approach). Several measures of statistical heterogeneity were calculated, including *I-squared* and *tau-squared*. If considerable heterogeneity was present, attempts were made to explain the source of heterogeneity by subgroup analysis, meta-regression or sensitivity analysis (only possible if enough studies were available).

Other sensitivity analyses included those needed to inform the judgements on RoB, large magnitude of effect size and publication bias within the certainty of evidence approach. Lastly, additional sensitivity analyses were conducted to explore the impact of multipollutant models, conflicts of interest of study authors, population characteristics or lag patterns, where appropriate.

2.4.4 Grading of the certainty of the overall body of evidence

Evaluation of the certainty of evidence is foundational for systematic review, with a focus on the validity and precision of effect estimates. In the clinical realm, evidence-informed review has become the starting point for establishing

guidelines for clinical practice, including guidance for therapeutics and diagnostics. Much of the evidence considered in the clinical context comes from randomized controlled trials, where exposures are assigned at random by the investigator to provide some degree of assurance that potential confounders and effect modifiers, both known and unknown, are balanced across treatment groups. In the clinical context, evidence may also come from observational studies, including cohort and case-control studies, case series and other data resources. Given the strength of the randomized controlled trial design for ensuring comparability of treatment and control groups, a hierarchy of evidence sources has been established in which randomized controlled trials (providing the strongest evidence) have the highest ranking and lower rankings are given to other sources.

The GRADE approach has been adopted as the basis for evidence review in support of WHO guidelines (Schünemann et al., 2013; WHO, 2014a). GRADE was implemented for the purpose of evaluating evidence in support of formulation of clinical guidelines and, as such, it divided studies into randomized and non-randomized designs and ranked randomized studies as higher-quality evidence.

The initial certainty level of evidence was determined by the type of study, with randomized controlled trials starting at high certainty and non-randomized studies starting at low certainty. Thereafter, five domains were assessed for downgrading the certainty of the evidence resulting from randomized and non-randomized studies, and three domains were assessed for upgrading the certainty of evidence from non-randomized studies alone (Box 2.3).

Consistent with the standard approach, the certainty of the effect estimate was graded as high, moderate, low or very low. The ratings were subsequently used to select the risk functions used to derive AQG levels.

With the extension of GRADE to topics for which evidence derives largely or totally from observational studies, there are areas for which evidence from randomized designs is not available and decision-making, of necessity, draws on other evidence. For environmental agents, the evidence foundation is diverse and with very few exceptions does not involve a randomized exposure (e.g. air cleaner with filter versus air cleaner without filter). The human evidence is observational, coming from population-level studies (time-series studies, geospatial analyses, cohort studies, case-control studies and cross-sectional studies). A further issue that arises with environmental agents is identifying and summarizing the evidence derived from toxicological studies, in vivo animal bioassays and in vitro work addressing mechanisms.

Box 2.3. GRADE domains

Domains assessed for downgrading the certainty of evidence by one or two levels

- Limitations or RoB in all studies that constitute the body of evidence
- Indirectness of evidence in the studies
- Inconsistency of results between studies
- Imprecision of the pooled effect estimate
- Publication bias detected in the body of evidence.

Domains assessed for upgrading the certainty of evidence by one level

- Large magnitude of the pooled effect estimate
- All plausible confounding shifting the pooled effect estimate towards the null
- Existence of a concentration–response gradient.

Source: adapted from WHO (2014a).

Recognizing these complexities, different groups have made efforts to adapt GRADE for the assessment of evidence on exposures, but limitations were still under discussion at the time of developing these guidelines (National Research Council, 2014; Morgan et al., 2016; Saracci, 2017; Steenland et al., 2020). In this context, a working group was convened to adjust the standard GRADE approach to the field of air quality and health. The current adaptation was not aimed to assess causality through an examination of all the relevant streams of research (Woodruff & Sutton, 2011), but instead aimed to rate how certain one can be that the “true” estimate of the association between an air pollutant and an adverse health effect lies within a particular range (Hultcrantz et al., 2017).

The working group decided to start the rating for air pollution observational studies at moderate rather than high certainty evidence because of the risk of unmeasured confounding in observational research. From this level, the certainty of the evidence was then downgraded or upgraded based on several criteria per GRADE domain.

In addition, the working group recognized the need for taking a more nuanced view of the evidence, as well as for incorporating the following additional criteria to complement or replace existing guidance:

- calculation of an 80% prediction interval, to help assess heterogeneity in conjunction with the 95% confidence interval (CI) (IntHout et al., 2016);
- calculation of the sample size needed for a study based on a specific RR and CI, to help guide judgements about imprecision (Rothman & Greenland, 2018);
- estimation of the extent to which confounding may influence a pooled effect size using the *E-value*, to facilitate judgements for large magnitude of effect size (Mathur & VanderWeele, 2020); and
- additional approaches to help assess publication bias, such as a subgroup analysis of multicentre studies compared with single-city studies in case of evidence based on time-series studies, an analysis of differences in effect estimates from earlier versus later studies, and a comparison with published results of attempts to quantify the magnitude of bias.

A detailed description of the adaptation of GRADE is provided in the supplementary materials of the articles published in the special issue of *Environment International* (WHO Global Air Quality Guidelines Working Group on Certainty of Evidence Assessment, 2020).

2.5 From evidence to recommendations

The GDG decided that the recommendations (AQG levels) would be primarily based on epidemiological evidence. The GDG discussed how to account for contextual factors in formulating the AQG levels. Given the very large variability in exposures, socioeconomic conditions and other policy considerations across the world, the GDG concluded that retaining and enhancing the widely adopted interim targets from the previous guidelines would be a more useful instrument to assist end-users in implementing the recommendations. Contextual factors should instead be considered during the policy-making process at national, regional or local level, as discussed in [Chapters 1 and 6](#) of this document. The recommendations were based on the certainty of evidence judgements alone, whereby low/very low certainty would prevent the GDG from formulating a recommendation for an AQG level. See, however, the caveats about this in [sections 2.5.1 and 2.5.2](#).

Furthermore, two additional elements of guidance are offered in these guidelines. These elements differ from the recommendations in that they are not derived from systematic reviews of evidence of adverse health effects from air pollution.

Instead, they are based on an expert assessment of several types of evidence that included their utility to support end-users in their efforts to improve air quality. These elements of guidance are interim targets and good practice statements. Interim targets are air pollutant levels that are higher than the AQG levels, but which authorities in highly polluted areas can use to develop pollution reduction policies that are achievable within realistic time frames. The interim targets should be regarded as steps towards ultimately achieving AQG levels in the future, rather than as end targets. The number and numerical values of the interim targets are pollutant specific, and they are justified in the relevant sections of [Chapter 3](#).

Contextual factors also did not play a direct role in the formulation of this guidance, although some considerations were described in a qualitative manner where relevant (e.g. burden of disease in relation to interim targets, resource considerations in relation to some good practice statements).

The following sections provide a detailed description of the approaches used by the GDG to formulate the recommendations and the additional guidance.

2.5.1 Formulation of long-term AQG levels

2.5.1.1 Definition

Long-term AQG levels are developed to provide advice to end-users to reduce the adverse effects of long-term exposure to air pollutants and, thereby, reducing associated disease and mortality.

Health outcomes in the current process are restricted to all-cause and respiratory mortality (PM_{2.5}, PM₁₀, ozone, nitrogen dioxide). In addition, cardiovascular and lung cancer mortality are considered for some pollutants (PM_{2.5}, PM₁₀).

A long-term AQG level is defined as the lowest exposure level of an air pollutant above which the GDG is confident that there is an increase in adverse health effects. Confidence refers primarily to the adapted GRADE qualification confirming that there is high or moderate certainty evidence for an association between a specific pollutant and a specific health outcome. The GRADE certainty rating is based on eight criteria (discussed later in this section). The GDG also took into account additional considerations, including causality determinations.

In principle, AQG levels were developed only for pollutant–outcome pairs with at least moderate certainty data. The GDG recognizes that, following the precautionary principle, conditional recommendations could be considered where the certainty of the evidence is less than moderate.

This would be the case, for instance, when exposure is widespread and the effect on population health is severe. However, as will become evident in [Chapter 3](#), there was at least moderate certainty evidence to support long- and short-term AQG levels for all pollutants considered.

This approach avoids consideration about what level of exposure should be considered safe, given that the available evidence cannot currently identify levels of exposure that are risk free for any of the pollutant–outcome pairs considered in this document. Moreover, the approach also avoids defining a so-called accepted level of risk, which would violate clean air acts or directives in countries where adverse health effects of air pollution are not accepted.⁷ It also avoids making inferences for exposure levels below those for which there is solid evidence. The challenge is then to find the lowest level of exposure for which the GDG is still confident that there is at least moderate certainty evidence for adverse health effects. Note that this also requires some consideration of what is an adverse health effect. As a reference for this, the GDG used the latest update of the joint European Respiratory Society and American Thoracic Society policy statement on “what constitutes an adverse health effect of air pollution” (Thurston et al., 2017).

The systematic reviews commissioned by WHO formed the starting point for the body of evidence on which an AQG level is based and, therefore, underpin these guidelines. The systematic reviews each provide a summary estimate of the RR derived from the included meta-analyses for each pair of air pollutant and adverse health effect, a 95% CI for this estimate and a GRADE qualification for the certainty of the evidence. In principle, the GDG used this estimate for guideline derivation only if the 95% CI from a random-effects meta-analysis did not include an RR of 1. However, as air pollution has no known health benefits, the GDG decided in specific, well-argued cases to deviate from this principle.

2.5.1.2 Procedure

To find the lowest level of long-term exposure for which the GDG would be confident of an adverse health effect, a dedicated working group developed a procedure for each pollutant–outcome pair, based on the following eight steps ([Table 2.5](#)).

7 For example, the US National Ambient Air Quality Standards are based on the Clean Air Act, which stipulates: “National primary ambient air quality standards, prescribed under subsection (a) shall be ambient air quality standards the attainment and maintenance of which in the judgment of the Administrator, based on such criteria and allowing an adequate margin of safety, are requisite to protect the public health” (42 U.S.C. 7409(a)).

Table 2.5. Eight steps in formulation of long-term AQG levels

Step	Description
Step 1	Assess RR estimates and, when available, CRF for each critical health outcome per pollutant as provided by the systematic review. In its first meeting in 2016, based on an initial survey, the GDG decided that the following health outcomes are critical (depending on air pollutant): (i) all-cause mortality (or all, natural-cause mortality, excluding accidental deaths); (ii) respiratory mortality; (iii) cardiovascular mortality, associated with both long- and short-term exposures; (iv) short-term, day-to-day variations in hospital admissions and emergency room visits related to asthma; and (v) myocardial infarction. The GDG recommends AQG levels for all pollutant–outcome pairs identified in 2016 except for those associations not meeting at least moderate levels of certainty. This includes pairs with different likelihoods of causality, according to authoritative reviews by COMEAP, Health Canada, International Agency for Research on Cancer, US EPA and others
Step 2	Determine the lowest level of exposure measured in the studies included in the systematic review or in the subset of studies in the systematic review that estimate risk at this lowest level. For individual studies that used statistical models to evaluate the shape of the CRF, ensure that the lowest level of exposure is associated with a monotonic increase of the CRF curve
Step 3	Determine the minimal relevant increase in health outcomes
Step 4	Determine the starting point for AQG level determination as the long-term concentration of pollutant from which the minimal relevant amount of the health outcome will result
Step 5	Compare the AQG levels for a specific pollutant across critical health outcomes. Take as the final AQG level the lowest AQG level found for any of the critical health outcomes
Step 6	Assess the certainty of the evidence at low levels of exposure. The adapted GRADE assessment is for the entire body of evidence, not the subset of studies conducted at the lowest exposure levels. The evidence provided by these latter studies needs to be discussed, starting from the RoB assessment that was conducted at individual study level
Step 7	Consider new relevant evidence not included in the systematic reviews in a qualitative or, where possible, quantitative manner
Step 8	Reconsider causality of associations between pollutants and outcomes, taking into account whether or not associations have been classified as causal or likely causal in recent reviews by COMEAP, Health Canada, US EPA, WHO or other authoritative bodies

Each of the eight steps is discussed below.

Step 1. The GDG used the meta-analytic effect estimate that results from the systematic review and the assessment of the certainty of the evidence that underpins this effect estimate according to GRADE. In principle, effect estimates are only used if the 95% CI does not include an RR of 1 in the random-effects meta-analysis of the relevant body of evidence for a specific exposure–outcome pair. In addition, they are only used when underpinned by moderate to high certainty evidence. This is because there would be little confidence in an AQG level based on a non-significant meta-analytic effect estimate or on an effect estimate for which there is only low-certainty evidence.

The GDG recognized that the probability value (*P* value) generated by a test of statistical significance is a continuous measure and that even a statistically non-significant result may be more consistent with a real increased risk than with the null. Therefore, in cases of borderline significance or where significance is restricted to major subgroups, the GDG decided whether or not to proceed with guideline development, regardless of the overall statistical significance. It was noted that in the meta-analyses of the systematic reviews, statistical significance was based on two-sided tests. As the air pollutants under consideration have no known health benefits, this indicated that careful consideration was necessary for any meta-analytic random-effects effect estimate with a two-sided *P* value of less than 0.10.

It is important to realize that the adapted GRADE assessments apply to the whole body of evidence or to some part thereof based on, for example, a selection of studies at low or moderate RoB. No separate GRADE assessments were carried out for the – necessarily smaller number of – studies providing information at the lowest levels of exposure. GRADE assessments for a small number of studies are less robust. Key elements of GRADE such as RoB can be assessed for a smaller number of studies, and this was done where applicable.

Step 2. Since the effect estimates examined in the systematic reviews were generally evaluated using linear models and existing evidence generally supports a linear or supralinear, no-threshold relationship for the pollutant–outcome pairs, there must be a procedure to determine the lowest level of observed (measured or modelled) exposure that is sufficiently underpinned with evidence and can, therefore, be used.

Pragmatically, the GDG used as a starting point the 5th percentile of the exposure distributions from at least a few studies with the lowest levels of exposure (see below).

The rationale was that below the 5th percentile of an exposure distribution, where data density tends to be sparse, there is typically little confidence in the shape of the CRF. This is evident, for example, from the CIs of splines reported in a number of relevant papers. Confidence depends on the study size. When there are no studies with narrow CIs for effect estimates down to the 5th percentile of the exposure distribution, a higher percentile can be chosen as a starting point.

One would hesitate to use the 5th percentile of just one study, but the bodies of evidence considered for AQG level derivation varied considerably in terms of the numbers of studies included in the meta-analyses. In each case, the GDG made a pragmatic choice of studies to include.

Step 3. Next, the GDG determined what amount or increase in mortality or other outcome above the lowest level would be considered a relevant increase. This is an a priori decision based on a judgement by the GDG. The GDG decided to use zero as a baseline when reviewing studies (i.e. any increase of the adverse health risk from the lowest long-term concentration – as defined in step 2 – would be considered relevant). A zero increase represents a figure that comes closest to the ideal of having an AQG level that is based on health arguments only. With a positive slope of the CRF at this lowest exposure level, any increase in exposure will result in a non-zero risk increase. See below, however, for a discussion of what zero means in practical terms, and how that differs for the derivation of long- and short-term AQG levels.

Step 4. The lowest level – the mean of a number of observed 5th percentile concentrations, as defined in step 2 – of measurement is the point above which the GDG assumed (with some confidence) that an increase in risk occurs. Since the GDG decided not to allow any increase in the adverse health risk from the lowest level measured, this is then the starting point for derivation of the AQG level.

Step 5. The GDG established an AQG level for all critical health outcomes associated with a specific pollutant following steps 1–4. Of these, the lowest AQG level is recommended as the WHO AQG level for that pollutant. This will prevent the possibility that, for example, using an AQG level based on all-cause mortality would still allow a substantial amount of asthma to occur. For example, if the AQG level for asthma were lower than the all-cause mortality level, the AQG level based on the asthma outcome would be taken as the WHO AQG level.

Step 6. No separate GRADE assessments were carried out for the relatively few studies reporting the lowest levels of exposure since GRADE was applied to the whole body of evidence and not to single studies.

Nevertheless, a critical discussion was warranted on the merits of studies reporting the lowest exposure levels. This discussion started from an assessment of the RoB, which was conducted at the individual study level. If a study that found a low exposure level was deemed to be at high RoB, then it was excluded from consideration unless the GDG had sound reasons to disagree with this assessment in the relevant systematic review. The GDG also considered whether studies conducted at the lowest exposure levels continued to show increased RRs.

Step 7. The systematic reviews concluded their literature searches in early autumn of 2018. Since then, several relevant studies have been published. The GDG considered new evidence up to the meeting in June 2020 – after verifying that it met the same standards for inclusion as the studies already included.

Step 8. The GDG reconsidered causality of associations for all pollutant–outcome pairs. However, as all pollutant–outcome pairs were considered worthy of further consideration at the start of the process in 2016, such considerations generally did not prevent recommendations of an AQG level whenever the epidemiological evidence was considered to be of moderate or high certainty.

Specifically, the GDG referred to the causality assessments shown in [Table 2.1](#), which formed the basis of the current AQG level development process. The assessment was updated, when necessary, to include newer evaluations published since 2016. These updates are all discussed at the end of [section 2.3.3](#).

The steps outlined above produce a rounded integer value as a starting point for AQG level development. This starting point is not equivalent to a threshold of no effect: it is merely a level below which there is less certainty about the existence of an effect. Where there was no threshold, the starting point level was associated with some effect on health. The magnitude of this effect could be estimated from the meta-analytic effect estimate from the systematic review by assuming that, in the absence of a threshold, any level of exposure increases risk. It was useful to do this as a benchmark for comparing the starting points for long-term AQG levels between the pollutants PM_{2.5}, PM₁₀, ozone and nitrogen dioxide. It also provided a benchmark for comparing estimated health effects between long- and short-term AQG levels for the same pollutant.

2.5.2 Formulation of short-term AQG levels

There are fundamental differences between AQG levels for short-term and long-term exposures. For long-term exposures, AQG levels are derived based on the lowest long-term exposures that are, with at least moderate certainty, associated with adverse health effects.

Such guidelines are typically expressed as annual averages. Daily and hourly concentrations vary around the annual average, often in a lognormal distribution. If short-term AQG levels are derived based on lowest short-term exposures that are – with at least moderate certainty – associated with adverse health effects, then much lower values are obtained than those determined for long-term AQG levels. (The caveat about evidence of less than moderate certainty expressed in [section 2.5.1](#) also applies here.) Importantly, the short-term variation in air pollution concentrations is largely driven by meteorology, which cannot be controlled. Short-term guidelines are typically defined as a high percentile of the distribution of daily values, for example the 98th or 99th percentiles equivalent to seven or three days a year exceeding this value (i.e. exceedance days). The rationale for choosing a high percentile and not the maximum is that the maximum of daily values for a given year is a less stable statistic than the high percentiles.

For locations in which concentrations are below the annual mean AQG level, days with such high daily mean concentrations will be rare and a large proportion of days will have concentrations below the annual mean AQG level. Thus, the health burden related to a few days with higher concentrations corresponds to a very small fraction of the total air pollution-related burden.

In contrast, the long-term variation in air pollution concentrations is largely driven by spatial variation in air pollution sources and emissions, which can be controlled, although control for some sources such as desert dust, pose unique and much more considerable challenges. Typically, the magnitude of the health effects associated with variations in long-term exposure is larger, per mass unit, than the magnitude of the health effects associated with short-term variations. As a consequence, long-term AQG levels for most health outcomes are more health protective than short-term AQG levels. In such instances, the long-term AQG level is used to derive a short-term AQG level whenever the same health effect is considered (e.g. mortality) for both long- and short-term exposures.

According to this line of reasoning, all eight steps outlined for long-term AQG level development remain valid for short-term AQG level development, except for step 3: defining the minimal relevant increase in health outcomes.

2.5.2.1 Procedure

In keeping with established practice, as a starting point, short-term AQG levels were considered by the GDG as the 99th percentiles of daily concentrations empirically observed in distributions with a mean equal to the long-term AQG level, for pollutant–outcome pairs for which a long-term AQG level is also being recommended. This is the case for all-cause mortality and PM_{2.5}, PM₁₀, ozone and

nitrogen dioxide. It is also the case for cause-specific mortality and PM_{2.5} and PM₁₀. The GDG evaluated the percentage of excess daily deaths expected from the meta-analytic linear short-term effect estimate to occur at a day at the 99th percentile of the distribution of daily, 24-hour average concentrations, compared with a day at the annual mean guideline concentration.

In the cases of sulfur dioxide and all-cause mortality and hospital admissions and emergency room visits related to asthma and of carbon monoxide and myocardial infarction, no long-term AQG levels were recommended and there are no long-term AQG levels from 2005. The same approach as described at the beginning of step 2 was followed, by evaluating the percentage of excess daily deaths expected from the meta-analytic linear effect estimate to occur at the 99th percentile, relative to a specified and justified low concentration. The rationale for the long-term reference concentrations of sulfur dioxide and carbon monoxide is discussed in [Chapter 3](#).

Once the starting point for the short-term AQG level was calculated, it was rounded to the nearest integer value.

The rationale for having short-term AQG levels next to long-term AQG levels for the same pollutant is based on the need to provide air quality managers, health-care providers, vulnerable patients and the general population with tools to communicate health risks and short-term emission controls. The GDG notes that there is substantial evidence that some susceptible groups may be harmed by short-term elevations of some pollutants: those with asthma, coronary heart disease, COPD and other chronic conditions and diseases. Overall, these susceptible groups represent a substantial proportion of the population in many countries.

The rationale for having short-term AQG levels in the absence of long-term AQG levels is typically based on documented acute elevation of risk over timescales of minutes to one or a few days.

More detailed advice to policy-makers and air quality managers is provided in [Chapter 6](#) of these guidelines.

In this protocol a distinction is made between three different scenarios ([Table 2.6](#)).

Scenario 1. In the first scenario, internally consistent long- and short-term AQG levels is desired, and the argument is that meeting the long-term AQG level protects against serious short-term effects on mortality. This can be shown for PM, nitrogen dioxide and ozone.

Table 2.6. Scenarios in formulation of short-term AQG levels

Scenario	Description
Scenario 1	Development of a short-term AQG level for a pollutant for which a long-term AQG level for the same outcome was developed (e.g. all-cause mortality)
Scenario 2	Development of a short-term AQG level for a pollutant for which a long-term AQG level was developed for another outcome (e.g. hospital admissions and emergency room visits related to asthma versus all-cause mortality)
Scenario 3	Development of a short-term AQG level for a pollutant for which no long-term AQG level was developed

First, for PM_{2.5} and all-cause mortality, in *Global update 2005* the annual mean air quality guideline is 10 µg/m³ and the short-term 99th percentile 24-hour average air quality guideline is 25 µg/m³ so the ratio between short-term and long-term guideline values was 2.5. At the time, this ratio was not empirically underpinned; the ratio was simply said to be 2.5, with some recognition that it may vary from place to place and from time to time. There is now a very large database – including the 652 cities from the Liu et al. (2019) paper – to document the ratios between higher percentiles of the distributions of 24-hour average concentrations and the annual means.

The GDG recommends using the same ratio everywhere for the purpose of deriving a 24-hour average AQG level. The primary motivation is that short-term effect estimates for PM_{2.5} and all-cause mortality do not significantly vary between different regions of the world. (Note that there are differences in effect estimates depending on PM_{2.5} level, but that is not important when deriving AQG levels for relatively low short-term concentrations; it is important when quantifying the health burdens associated with the higher interim target levels.)

The database from the MCC Collaborative Research Network (A. Gasparri, London School of Hygiene and Tropical Medicine, unpublished data, 23 June 2020) has descriptive data on long time series of daily average PM_{2.5} and PM₁₀ concentrations from 384 and 480 cities, respectively. The ratio of the 99th percentile of the daily average concentrations to the multiyear mean is 3.05 for PM_{2.5}, 2.85 for PM₁₀, 2.34 for nitrogen dioxide (398 cities), 2.05 for ozone (244 cities), 3.90 for sulfur dioxide (396 cities) and 2.97 for carbon monoxide (349 cities).

Based on this database, the Network has published a series of articles, which are published as open access (Chen et al., 2021; Liu et al., 2019; Meng et al., 2021; Vicedo-Cabrera et al., 2020). When considering long- and short-term AQG levels for ozone, the GDG realized that long-term AQG levels could be based on the mean peak-season ozone levels, which have a different relationship to the 99th percentile of the daily distributions than the annual means.

As an example, the GDG recommended setting the long-term AQG level for PM_{2.5} at 5 µg/m³, and if a ratio of 3 were used to calculate the corresponding 99th percentile of daily means, a 24-hour AQG level of 15 µg/m³ would be derived. All the recommendations can be found in [Chapter 3](#).

Scenario 2. In the second scenario, there may be long-term AQG levels for nitrogen dioxide and ozone based on effect estimates for respiratory mortality, and short-term AQG levels based on effect estimates for all-cause mortality only. For PM_{2.5} and PM₁₀, there are long- and short-term effect estimates for all-cause mortality as well as a number of cause-specific mortalities. In most cases, these are from the same studies, so there are no serious differences between the 5th percentiles of PM in the lowest-level studies for natural-cause and cause-specific mortality. If there are differences, the expectation is that in the smaller numbers of cause-specific mortality studies the 5th percentiles of the concentration distributions are more likely to be higher than lower, compared with the all-cause mortality studies. The general pattern is that effect estimates for both long- and short-term cause-specific mortality are somewhat bigger than those for all-cause mortality. This is always true for PM_{2.5}; the picture for PM₁₀ is a bit more mixed. Nevertheless, these patterns do not lead to different conclusions for AQG level derivation based on all-cause mortality as compared with AQG levels based on cause-specific mortality. This again assumes that the 5th percentiles from the lowest-level studies are not lower for cause-specific mortality studies than for all-cause mortality studies.

There will be short-term AQG levels for ozone, nitrogen dioxide and sulfur dioxide and hospital admissions and emergency room visits related to asthma, and for carbon monoxide and hospital admissions and emergency room visits related to myocardial infarction. The GDG recommends (for ozone and nitrogen dioxide) to start from the long-term AQG level based on mortality, look at the internally consistent short-term AQG level for mortality and then quantify the effect on hospital admissions and emergency room visits related to asthma at that level. Here, too, data from the Liu et al. (2019) collaboration provide insight into the ratios between 99th percentiles and annual means for ozone as well as for nitrogen dioxide.

A judgement on whether one or the other effect should drive the short-term AQG level is then needed, with potential consequences for consistency between long- and short-term AQG levels. As mentioned before, for ozone and nitrogen dioxide, only short-term CRFs for all-cause mortality are available. Therefore, comparisons (as outlined for PM above) are not always possible.

Scenario 3. Lastly, in the third scenario, a short-term AQG level needs to be derived for a pollutant for which no long-term AQG level is being developed.

A case in point is sulfur dioxide, for which there are systematic reviews of short-term 24-hour associations with all-cause and respiratory mortality and asthma hospital admissions and emergency room visits.

For carbon monoxide, there is a 24-hour AQG level based on the systematic review of associations between 24-hour mean carbon monoxide concentrations and hospital admissions and emergency room visits due to myocardial infarction, which can then be compared with the air quality guideline of 7 mg/m³ for carbon monoxide indoors (WHO, 2014b) and the shorter-term AQG levels as well.

To develop an AQG level for a particular pollutant–outcome pair, the GDG examined external evidence for causality of the pollutant–outcome association. Causality judgements were part of the process that produced the PECOS questions for the current process ([Table 2.1](#)).

In the case of hospital admissions and emergency room visits related to asthma and myocardial infarction, further adaptations were needed to compare visits/admissions to deaths. The GDG specified short-term AQG levels for hospital admissions and emergency room visits related to asthma or myocardial infarction based on quantification of the expected increase in such visits/admissions at the proposed short-term AQG level. This recognizes that the health burden related to a few days (three to four per year when using 99th percentiles) with higher concentrations corresponds to a very small fraction of the total air pollution-related burden.

2.5.3 Formulation of interim targets and good practice statements

2.5.3.1 Interim targets

Interim targets were introduced in *Global update 2005* as additional integral elements of guidance, designed to complement the WHO air quality guidelines.

Interim targets may be defined as air pollutant concentrations associated with a

specific decrease in health risk that serve as “incremental steps in progressive reduction of air pollution [...] intended for use in areas where pollution is high” (WHO Regional Office for Europe, 2006). As stated in *Global update 2005*, “countries may find these interim targets helpful in gauging progress over time in the difficult process of steadily reducing population exposures [to air pollution]”.

Moreover, interim targets “aim to promote a shift from high air pollutant concentrations, with acute and serious health consequences, to lower concentrations” (WHO Regional Office for Europe, 2006), in line with the AQG levels. Further:

[i]f these [interim] targets were to be achieved, one could expect significant reductions in risks for acute and chronic human health effects from air pollution. Progress towards the guideline values should, however, be the ultimate objective of air quality management and health risk reduction in all areas (WHO Regional Office for Europe, 2006).

The GDG decided that interim targets, and specifically the 2005 interim targets, should be retained in the updated air quality guidelines for two reasons.

- The first is to promote continuous air quality improvement in places with high levels of ambient air pollution with the goal of achieving AQG levels as expeditiously as possible. Interim targets for reduction of air pollution have been shown to be achievable with abatement measures and have practical value in that several countries have standards equal to some of the interim targets (Kutlar Joss et al., 2017). Importantly, interim targets also have been helpful in achieving AQG levels.
- The second is to maintain continuity. Policy-makers, nongovernmental organizations and the scientific community in low- and middle-income countries are already familiar with the 2005 interim targets and have employed them since their introduction 15 years ago. Changing the interim targets at this point would be confusing and unnecessary because the interim target levels are still globally relevant, although the 2005 air quality guideline would be added as an interim target in the event that the AQG level is lowered.

Descriptors for each interim target have been provided to inform decision-makers of the implications of achieving the corresponding air pollutant concentrations. These are the risk descriptors calculated using updated CRFs.

Lastly, the results of simulating a reduction of the 2016 burden of disease attributable to PM_{2.5} to the interim target and the new AQG level are provided in

section 3.9, in order to illustrate the mortality and disability adjusted life-year benefits that could be achieved by expeditiously reducing air pollutant levels (Evangelopoulos et al., 2020).

2.5.3.2 Good practice statements

The *WHO handbook for guideline development, 2nd edition* (WHO, 2014a), provides for the development of good practice statements in certain cases. This occurs when a GDG is confident that a large body of diverse evidence that is difficult to synthesize indicates that the desirable effects of a particular course of action far outweigh its undesirable effects. In other words, when a GDG is confident that implementing a measure would be beneficial with high certainty but when conducting numerous systematic reviews and detailed assessments of evidence would be a poor use of resources (WHO, 2014c).

The evidence considered may be of a diverse nature, including linked or indirect evidence, physical and biochemical properties, ethical principles and human rights conventions (WHO, 2019a). Along these lines, the types of evidence that the GDG may consider in the context of air quality guidelines would also include air quality management principles and good practices implemented by reputable institutions.

The option of developing good practice statements was used to provide much-needed guidance in relation to some specific types of PM identified as critical in the preliminary phase. The GDG chose to closely follow-up major external reviews on BC/EC, UFP and SDS throughout the process. The decision was made to develop good practice statements for these, rather than numerical AQG levels, in the absence of clear quantitative evidence on independent health effects from these pollutants.

3

Recommendations on classical air pollutants

3.1 Introduction

This chapter presents specific recommendations on air quality guideline (AQG) levels for the pollutants PM_{2.5}, PM₁₀, ozone, nitrogen dioxide, sulfur dioxide and carbon monoxide, together with the corresponding interim targets.

In [Chapter 2](#), a detailed protocol was described that was followed to derive AQG levels for the pollutants and averaging times. [Chapter 2](#) also provide the rationales for including the specific pollutant–outcome associations that formed the basis for the recommendations given in this chapter. The averaging times considered were long term (annual mean or, for ozone, highest six-month average) and short term (24 hours). Long-term effects were considered only for all-cause and cause-specific mortality (PM_{2.5}, PM₁₀, ozone and nitrogen dioxide). For those, any pollutant-attributed increase in long-term mortality was considered harmful. Short-term effects were considered for all non-accidental and cause-specific mortality (PM_{2.5}, PM₁₀, ozone, nitrogen dioxide and sulfur dioxide), for asthma hospital admissions and emergency room visits (ozone, nitrogen dioxide and sulfur dioxide), and for myocardial infarction hospital admissions and emergency room visits (carbon monoxide only). When both long- and short-term AQG levels were considered for a pollutant–outcome pair, preference was given to the long-term AQG level and the short-term AQG level was aligned using empirical data on frequency distributions of 24-hour concentrations. When only short-term AQG levels were considered, analogy with other pollutant–outcome pairs was used.

Information about all the specific pollutant–outcome pairs reviewed can be found in the systematic reviews of evidence available in a special issue of *Environment International* (Whaley et al., 2021).

3.2 PM_{2.5}

3.2.1 General description

The general description comes from *Global update 2005*.

PM in urban and non-urban environments is a complex mixture with components having diverse chemical and physical characteristics. Research on PM and the interpretation of research findings on exposure and risk are complicated by this heterogeneity, and the possibility that the potential of particles to cause injury varies with size and other physical characteristics, chemical composition and source(s). Different characteristics of PM may be relevant to different health effects. Newer research findings continue to highlight this complexity and the dynamic nature of airborne PM, as it is formed either primarily or secondarily

and then continues to undergo chemical and physical transformation in the atmosphere.

Nonetheless, particles are still generally classified by their aerodynamic properties, because these determine transport and removal processes in the air and deposition sites and clearance pathways within the respiratory tract. The aerodynamic diameter is used as the summary indicator of particle size; the aerodynamic diameter corresponds to the size of a unit-density sphere with the same aerodynamic characteristics as the particle of interest. The differences in aerodynamic properties among particles are exploited by many particle sampling techniques (WHO Regional Office for Europe, 2006).

The focus in recent decades has been on particles with aerodynamic diameters of less than or equal to 2.5 μm ($\text{PM}_{2.5}$) or 10 μm (PM_{10}).

3.2.2 Recommended AQG level for long-term exposure to $\text{PM}_{2.5}$

Based on the methods for deriving an AQG level outlined in the guideline development protocol in [Chapter 2](#), this section provides a recommendation for an annual AQG level for $\text{PM}_{2.5}$ that is based on all non-accidental mortality and cause-specific mortality ([Table 3.1](#)).

The epidemiological evidence underpinning the AQG level is discussed in a systematic review commissioned by WHO, which is referred to in [section 2.4](#). The review of this pollutant (Chen & Hoek, 2020) was published in *Environment International* (Whaley et al., 2021) as open access.

As discussed in [section 2.3](#), there has been no separate, independent assessment of the mechanistic, toxicological and human clinical studies relating ambient particles to human health.

The recommendations in this chapter follow the eight steps outlined in the protocol for AQG level development in [Chapter 2](#) ([section 2.5](#)). The tables and figures mentioned during the eight steps are listed at the end of the discussion of each recommendation.

Step 1. Assess RR estimates and, when available, CRFs

The systematic review on $\text{PM}_{2.5}$ and all non-accidental mortality (Chen & Hoek, 2020) reported a meta-analytic effect estimate of RR of 1.08 (95% CI: 1.06–1.09) per 10 $\mu\text{g}/\text{m}^3$ $\text{PM}_{2.5}$, assuming a linear relationship. The authors found an indication of a supralinear relationship, suggesting a steeper risk increase at lower exposure levels.

The certainty of the evidence was considered high according to GRADE. CRFs were provided by several studies. These are shown in [Fig. 3.1](#), [Fig. 3.2](#), [Fig. 3.3](#) and [Fig. 3.4](#) (which follow a discussion of the eight steps) for the studies with information at low to very low levels of measured exposure (step 2) (Pinault et al., 2016, 2017; Villeneuve et al., 2015; Di et al., 2017a). CRFs were published from four of the six studies with the lowest exposure levels. Two studies did not provide a CRF (Weichenthal et al., 2014; Cakmak et al., 2018). For obvious reasons, the uncertainty in the shape of the CRFs is higher in single than in pooled studies, and higher in small than in large studies. Very large studies such as the study by Di et al. (2017a) provide the best evidence for the shape of the CRF at the low end of the exposure range. These shapes generally show linear relationships down to very low concentrations or somewhat steeper curves at low than at higher concentrations.

Step 2. Determine the lowest level of exposure measured

In 18 of the 25 studies included in the meta-analysis, a 5th percentile of the exposure distribution was reported or could be calculated from the reported mean and standard deviation ([Table 3.2](#)). As the concentration distributions are often lognormal, this calculation is not straightforward. Therefore, preference was given to actual reports of the relevant numbers obtained from the published papers or upon request from the study authors. This is indicated in [Table 3.2](#), [Table 3.3](#), [Table 3.4](#) and [Table 3.5](#). The five lowest levels reported or estimated in these studies were 3.0 $\mu\text{g}/\text{m}^3$ (Pinault et al., 2016), 3.2 $\mu\text{g}/\text{m}^3$ (Cakmak et al., 2018), 3.5 $\mu\text{g}/\text{m}^3$ (Pinault et al., 2017), 4.8 $\mu\text{g}/\text{m}^3$ (Villeneuve et al., 2015) and 6.7 $\mu\text{g}/\text{m}^3$ (Weichenthal et al., 2014). Weichenthal et al. (2014) found no effect. The Villeneuve et al. (2015) study provided no evidence of an effect of $\text{PM}_{2.5}$ on all non-accidental mortality below 8 $\mu\text{g}/\text{m}^3$. The study by Di et al. (2017a) has the next lowest 5th percentile (7.1 $\mu\text{g}/\text{m}^3$) and the study by Hart et al. (2015) the next lowest (7.8 $\mu\text{g}/\text{m}^3$). The average $\text{PM}_{2.5}$ level across these five studies with the lowest exposure measurements in the systematic review is 4.2 $\mu\text{g}/\text{m}^3$. A sensitivity analysis disregarding the Villeneuve et al. (2015) and Weichenthal et al. (2014) studies produced a mean of 4.9 $\mu\text{g}/\text{m}^3$ $\text{PM}_{2.5}$. The sum of weights in the meta-analysis was > 25%, indicating that these studies were influential in the meta-analysis.

Step 3. Determine the minimal relevant increase in health outcomes

The GDG decided to consider as relevant any increase in risk for an adverse health outcome related to long-term exposure to a pollutant.

Step 4. Determine the starting point for AQG level determination as the long-term concentration of the pollutant from which the minimal relevant amount of the health outcome will result

The average of the five lowest 5th percentile levels measured in these five studies was the starting point for deriving an AQG level (4.2–4.9 $\mu\text{g}/\text{m}^3$ $\text{PM}_{2.5}$). The data obtained support a long-term AQG level of no more than 5 $\mu\text{g}/\text{m}^3$, based on the association between long-term $\text{PM}_{2.5}$ and all non-accidental mortality.

Step 5. Compare the AQG level across critical health outcomes: cause-specific mortality

The cause-specific mortality outcomes that were investigated all yielded bigger hazard ratios (HRs) for $\text{PM}_{2.5}$ compared with the HR for all non-accidental mortality, with an HR of 1.11 (95% CI: 1.09–1.14) for circulatory mortality, 1.10 (95% CI: 1.03–1.18) for non-malignant respiratory mortality and 1.12 (95% CI: 1.07–1.16) for lung cancer mortality. The certainty of the evidence on $\text{PM}_{2.5}$ was rated as high for circulatory and lung cancer mortality and moderate for non-malignant respiratory mortality. Starting points for AQG level determination for these other outcomes would be 4.0–4.3 $\mu\text{g}/\text{m}^3$ based on the five studies with the lowest 5th percentiles and 4.1–6.2 $\mu\text{g}/\text{m}^3$ based on the five studies documenting positive associations (HR > 1) for these three cause-specific mortality end-points (Table 3.3, Table 3.4 and Table 3.5). The data obtained for cause-specific mortality also support a long-term $\text{PM}_{2.5}$ AQG level of no more than 5 $\mu\text{g}/\text{m}^3$.

Step 6. Assess certainty of the evidence

None of the studies that make up the lowest levels measured in the all-cause mortality studies were considered to have a high RoB; thus, there is no reason to change the AQG level because of low certainty of the evidence in the lowest-level studies.

Step 7. Consider new evidence

Several new studies were published between autumn 2018 and the summer of 2020. They are discussed in the systematic review by Chen & Hoek (2020). When adding the new studies to the meta-analysis, the joint effect estimate for all-cause mortality and $\text{PM}_{2.5}$ was exactly the same as for the studies already included (Fig. A7.43 in Chen & Hoek (2020)). therefore, there is no reason to change the assessment based on the newly published studies. Chen & Hoek (2020) also referred to an analysis of a large number of cohort studies from many different areas of the world, showing a near linear association between annual $\text{PM}_{2.5}$ and all-cause mortality, defined as mortality from NCD plus lower respiratory illness, over a range of 2.4–80 $\mu\text{g}/\text{m}^3$ (Fig. 3.5; published as Fig. 1 in Burnett et al. (2018)).

Step 8. Reconsider causality

All PM–outcome associations were deemed to be causal or likely causal in the 2016 outcome prioritization framework (see [section 2.3.3](#)). These judgements have not changed in more recent authoritative assessments. For more details, see [Table 2.1](#) and additional text in [section 2.3.3](#).

The 5th percentile and mean or median of exposure distributions in studies of PM_{2.5} and the all-cause mortality meta-analysis results are indicated in [Table 3.2](#) based on data from the systematic review by Chen & Hoek (2020). [Table 3.3](#), [Table 3.4](#) and [Table 3.5](#) have the same information for studies on circulatory, non-malignant respiratory and lung cancer mortality, respectively.

3.2.2.1 Interim targets

Interim targets are proposed as incremental steps in a progressive reduction of air pollution and are intended for use in areas where pollution is high. For a more detailed rationale for establishing and using interim targets, see [section 2.5.3](#).

**The recommendation is an annual PM_{2.5} AQG level of 5 µg/m³.
The GDG recommends maintaining the 2005 interim targets and
introducing an interim target 4 at the level of the 2005 air quality
guideline, as shown in [Table 3.1](#).**

Table 3.1. Recommended annual AQG level and interim targets for PM_{2.5}

Recommendation	PM _{2.5} (µg/m ³)
Interim target 1	35
Interim target 2	25
Interim target 3	15
Interim target 4	10
AQG level	5

If mortality in a population exposed to PM_{2.5} at the AQG level is arbitrarily set to 100, then it will be 124, 116, 108 and 104, respectively, in populations exposed to PM_{2.5} at interim target 1, 2, 3 and 4 levels. These projections are based on the linear HR of 1.08 per 10-µg/m³ increase in PM_{2.5} for all non-accidental mortality reported in the systematic review. At higher concentrations, the CRF may no longer be linear, which would change the numbers in this example.

Table 3.2. Studies on long-term PM_{2.5} exposure and all non-accidental mortality included in the systematic review by Chen & Hoek (2020), ordered by me(di)an concentration

Study	Me(di)an (µg/m ³)	SD	P5	P25	HR (95% CI) ^a
Pinault et al. (2016)	5.9	–	3.0 ^b	4.2	1.26 (1.19–1.34)
Cakmak et al. (2018)	6.5	2.0	3.2 ^c	–	1.16 (1.08–1.25)
Pinault et al. (2017)	7.1	–	3.5 ^b	5.4	1.18 (1.15–1.21)
Weichenthal et al. (2014)	9.5	1.7	6.7 ^c	–	0.95 (0.76–1.19)
Villeneuve et al. (2015)	9.5	3.5	4.8 ^b	–	1.12 (1.05–1.20)
Di et al. (2017a)	11.5	2.9	7.1 ^b	9.5	1.08 (1.08–1.09)
Parker, Kravets & Vaidyanathan (2018)	11.8	–	–	10.1	1.03 (0.99–1.08)
Bowe et al. (2018)	11.8	–	7.9 ^b	10.2	1.08 (1.03–1.13)
Hart et al. (2015)	12.0	2.8	7.8 ^b	10.2	1.13 (1.05–1.22)
Turner et al. (2016)	12.6	2.9	7.8 ^c	–	1.07 (1.06–1.09)
Carey et al. (2013)	12.9	1.4	10.6 ^c	–	1.11 (0.98–1.26)
Beelen et al. (2014)	13.4	–	7.9 ^b	11.3	1.14 (1.03–1.27)
Thurston et al. (2016a)	13.6	3.6	8.9 ^b	11.1	1.03 (1.01–1.06)
Hart et al. (2011)	14.1	4.0	7.8 ^b	11.8	1.10 (1.02–1.18)
Lepeule et al. (2012)	15.9	–	–	–	1.14 (1.07–1.22)
Bentayeb et al. (2015)	17.0	–	–	–	1.16 (0.98–1.36)
Puett et al. (2011)	17.8	3.4	12.2 ^c	–	0.86 (0.72–1.02)
Ostro et al. (2015)	17.9	–	–	13.1	1.01 (0.97–1.05)
Badaloni et al. (2017)	19.6	1.9	16.5 ^c	–	1.05 (1.02–1.08)
Enstrom (2005)	23.4	–	–	–	1.01 (0.99–1.03)
Beelen et al. (2008)	28.3	2.1	24.8 ^c	–	1.06 (0.97–1.16)
Tseng et al. (2015)	29.6	–	–	–	0.92 (0.72–1.17)
Yin et al. (2017)	40.7	18.6	10.1 ^c	–	1.09 (1.08–1.10)
Yang et al. (2018)	42.2	–	–	–	1.06 (1.01–1.10)
McDonnell et al. (2000)	59.2	16.8	31.6 ^c	–	1.09 (0.98–1.21)

–, data unavailable; P5: 5th percentile (of the distribution of concentrations assigned to study participants);

P25: 25th percentile; HR: hazard ratio; SD: standard deviation.

^a Per 10 µg/m³.

^b Reported in paper or by authors on request.

^c Calculated from mean and standard deviation using the following formula: Me(di)an – 1.645 × SD.

Table 3.3. Studies on long-term PM_{2.5} exposure and circulatory mortality included in the systematic review by Chen & Hoek (2020), ordered by me(di)an concentration

Study	Me(di)an (µg/m ³)	SD	P5	P25	HR (95% CI) ^a
Pinault et al. (2016)	5.9	–	3.0 ^b	4.2	1.19 (1.07–1.31)
Pinault et al. (2017)	7.1	–	3.5 ^b	5.4	1.25 (1.19–1.30)
Crouse et al. (2015)	8.9	–	3.5 ^b	6.0	1.06 (1.04–1.08)
Weichenthal et al. (2014)	9.5	1.7	6.7 ^c	–	1.15 (0.76–1.73)
Villeneuve et al. (2015)	9.5	3.5	3.7 ^c	–	1.32 (1.14–1.52)
Dehbi et al. (2017)	9.9	–	–	9.4	1.30 (0.39–4.34)
Parker, Kravets & Vaidyanathan (2018)	11.8	–	–	10.1	1.16 (1.08–1.25)
Turner et al. (2016)	12.6	2.9	7.8 ^c	–	1.12 (1.09–1.15)
Carey et al. (2013)	12.9	1.4	10.6 ^c	–	1.00 (0.85–1.17)
Vedal et al. (2013)	12.9	2.8	8.3 ^c	–	1.31 (0.94–1.83)
Beelen et al. (2014)	13.4	–	7.9 ^b	11.3	0.98 (0.83–1.16)
Thurston et al. (2016a)	13.6	3.6	8.9 ^b	11.1	1.05 (0.98–1.13)
Hart et al. (2011)	14.1	4.0	7.8 ^b	11.8	1.05 (0.93–1.19)
Laden et al. (2006)	–	–	–	–	1.08 (0.79–1.48)
Bentayeb et al. (2015)	17.0	–	–	–	1.21 (0.72–2.04)
Ostro et al. (2015)	17.9	–	–	13.1	1.05 (0.99–1.12)
Badaloni et al. (2017)	19.6	1.9	16.5 ^c	–	1.08 (1.03–1.12)
Beelen et al. (2008)	28.3	2.1	24.8 ^c	–	1.07 (0.75–1.52)
Tseng et al. (2015)	29.6	–	–	–	0.80 (0.43–1.49)
Yin et al. (2017)	40.7	18.6	10.1 ^c	–	1.09 (1.08–1.10)
Yang et al. (2018)	42.2	–	–	–	1.02 (0.93–1.11)

–, data unavailable; P5: 5th percentile (of the distribution of concentrations assigned to study participants); P25: 25th percentile; SD: standard deviation.

^a Per 10 µg/m³.

^b Reported in paper or by authors on request.

^c Calculated from mean and standard deviation using the following formula: Me(di)an – 1.645 × SD.

Table 3.4. Studies on long-term PM_{2.5} exposure and non-malignant respiratory mortality included in the systematic review by Chen & Hoek (2020), ordered by me(di)an concentration

Study	Me(di)an (µg/m³)	SD	P5	P25	HR (95% CI) ^a
Pinault et al. (2016)	5.9	–	3.0 ^b	4.2	1.52 (1.26–1.84)
Pinault et al. (2017)	7.1	–	3.5 ^b	5.4	1.22 (1.12–1.32)
Crouse et al. (2015)	8.9	–	3.5 ^b	6.0	0.95 (0.91–0.98)
Villeneuve et al. (2015)	9.5	3.5	3.7 ^c	–	0.82 (0.61–1.11)
Turner et al. (2016)	12.6	2.9	7.8 ^c	–	1.16 (1.10–1.22)
Carey et al. (2013)	12.9	1.4	10.6 ^c	–	1.57 (1.30–1.91)
Dimakopoulou et al. (2014)	13.4	–	7.9 ^b	11.3	0.79 (0.47–1.34)
Thurston et al. (2016a)	13.6	3.6	8.9 ^b	11.1	1.10 (1.05–1.15)
Hart et al. (2011)	14.1	4.0	7.8 ^b	11.8	1.18 (0.91–1.53)
Laden et al. (2006)	14.8	–	–	–	1.08 (0.79–1.48)
Bentayeb et al. (2015)	17.0	–	–	–	0.88 (0.57–1.36)
Ostro et al. (2015)	17.9	–	–	13.1	0.99 (0.90–1.09)
Cesaroni et al. (2013)	23.0	4.4	15.8 ^c	20.3	1.03 (0.98–1.08)
Beelen et al. (2008)	28.3	2.1	24.8 ^c	–	1.04 (0.90–1.21)
Katanoda et al. (2011)	30.5	–	–	–	1.16 (1.04–1.30)
Yang et al. (2018)	42.2	–	–	–	1.11 (1.04–1.19)
McDonnell et al. (2000)	59.2	16.8	31.6 ^c	–	1.23 (0.97–1.55)

–, data unavailable; P5: 5th percentile (of the distribution of concentrations assigned to study participants); P25: 25th percentile; SD: standard deviation.

^a Per 10 µg/m³.

^b Reported in paper or by authors on request.

^c Calculated from mean and standard deviation using the following formula: Me(di)an – 1.645 × SD.

Table 3.5. Studies on long-term PM_{2.5} exposure and lung cancer mortality included in the systematic review by Chen & Hoek (2020), ordered by me(di)an concentration

Study	Me(di)an (µg/m ³)	SD	P5	P25	HR (95% CI) ^a
Pinault et al. (2016)	5.9	–	3.0 ^b	4.2	1.17 (0.98–1.40)
Cakmak et al. (2018)	6.5	2.0	3.2 ^c	–	1.29 (1.06–1.59)
Pinault et al. (2017)	7.1	–	3.5 ^b	5.4	1.16 (1.07–1.25)
Weichenthal et al. (2014)	9.5	1.7	6.7 ^c	–	0.75 (0.34–1.65)
Villeneuve et al. (2015)	9.5	3.5	3.7 ^c	–	0.97 (0.80–1.18)
Turner et al. (2016)	12.6	2.9	7.8 ^c	–	1.09 (1.03–1.16)
Carey et al. (2013)	12.9	1.4	10.6 ^c	–	1.11 (0.86–1.44)
Hart et al. (2011)	14.1	4	7.8 ^b	11.8	1.05 (0.88–1.26)
Lepeule et al. (2012)	15.9	–	–	–	1.37 (1.07–1.75)
Cesaroni et al. (2013)	23.0	4.4	15.8 ^c	20.3	1.05 (1.01–1.10)
Beelen et al. (2008)	28.3	2.1	24.8 ^c	–	1.06 (0.82–1.38)
Katanoda et al. (2011)	30.5	–	–	–	1.24 (1.12–1.37)
Yin et al. (2017)	40.7	18.6	10.1 ^c	–	1.12 (1.09–1.16)
McDonnell et al. (2000)	59.2	16.8	31.6 ^c	–	1.39 (0.79–2.46)
Lipsett et al. (2011)	–	–	–	–	0.95 (0.70–1.28)

–, data unavailable; P5: 5th percentile (of the distribution of concentrations assigned to study participants);

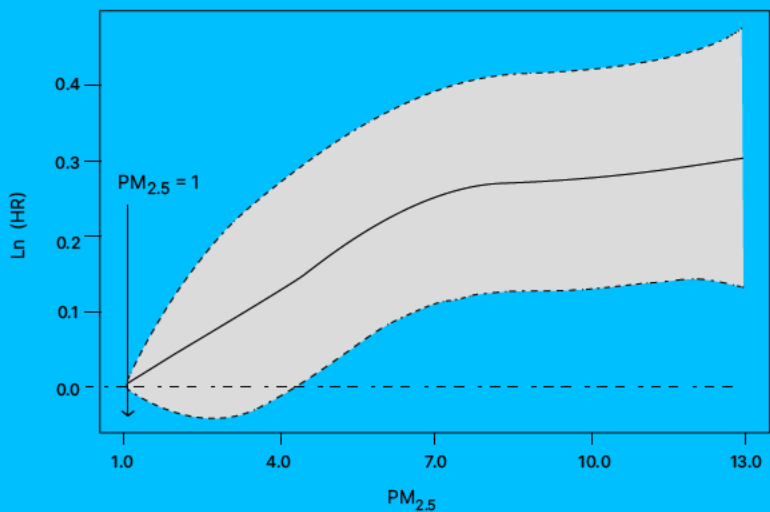
P25: 25th percentile; SD: standard deviation.

^a Per 10 µg/m³.

^b Reported in paper or by authors on request.

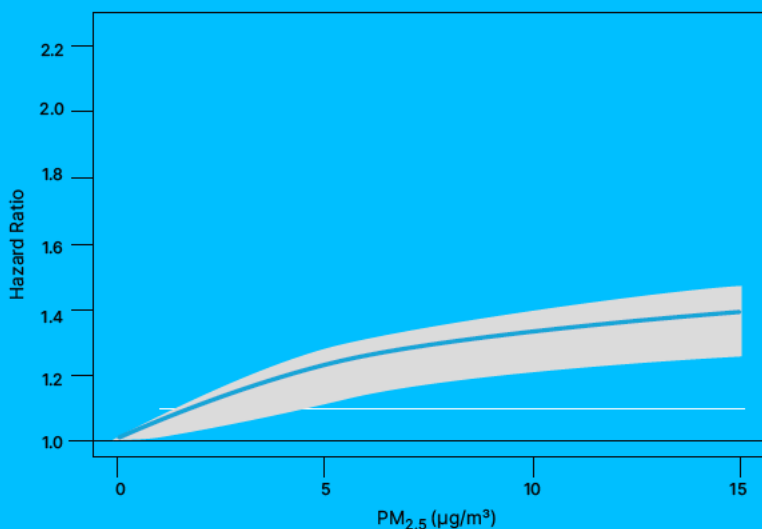
^c Calculated from mean and standard deviation using the following formula: Me(di)an – 1.645 × SD.

Fig. 3.1. CRF for long-term PM_{2.5} exposure (µg/m³) and all non-accidental mortality



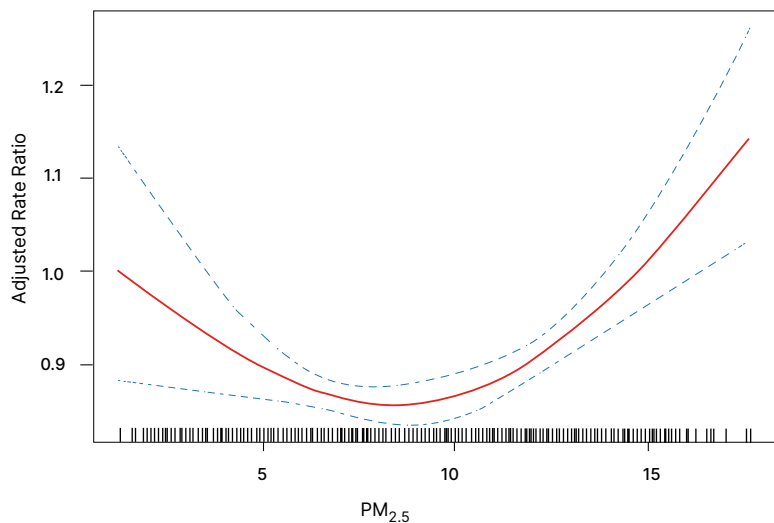
Ln (HR): log HR, with an HR of 1 at a PM_{2.5} concentration of 1 µg/m³.
Source: Pinault et al. (2016).

Fig. 3.2. CRF for long-term PM_{2.5} exposure (µg/m³) and all non-accidental mortality



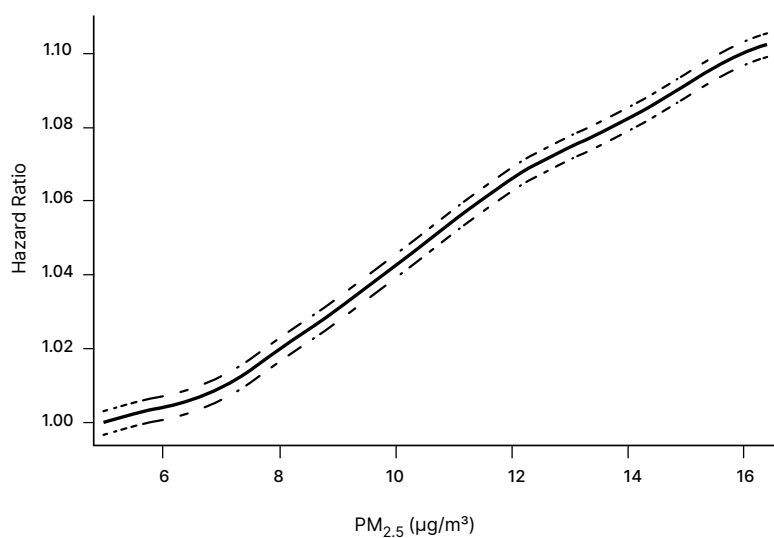
Source: reprinted from Pinault et al. (2017) with permission from Elsevier.

Fig. 3.3. CRF for long-term PM_{2.5} exposure (µg/m³) and all non-accidental mortality



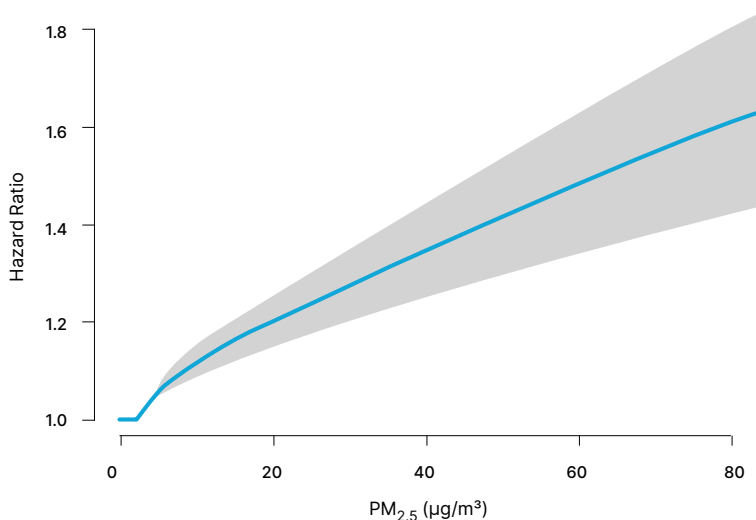
Source: reprinted from Villeneuve et al. (2015) with permission from the publisher. Copyright © 2015 Wolters Kluwer Health, Inc. All rights reserved.

Fig. 3.4. CRF for long-term PM_{2.5} exposure (µg/m³) and all non-accidental mortality



Source: reprinted from Di et al. (2017a) with permission from the Massachusetts Medical Society. Copyright © 2017 Massachusetts Medical Society.

Fig. 3.5. Association between long-term PM_{2.5} exposure (µg/m³) and mortality from NCDs and lower respiratory illness, as observed in an analysis of data from 41 different cohort studies



Notes: The lowest observed PM_{2.5} concentration was 2.4 µg/m³.

Source: Burnett et al. (2018), Fig. 1.

3.2.3 Recommended AQG level for short-term exposure to PM_{2.5}

Based on the methods for deriving an AQG level outlined in the guideline development protocol, this section provides a recommended AQG level for short-term, 24-hour average PM_{2.5} that is based on all-cause non-accidental mortality and cause-specific mortality (Table 3.6).

The epidemiological evidence underpinning the AQG level is discussed in a systematic review commissioned by WHO, as explained in more detail in section 2.4. The review (Orellano et al., 2020) was published in *Environment International* (Whaley et al., 2021) as open access.

As discussed in section 2.3, there has been no separate, independent assessment of the mechanistic, toxicological and human clinical studies relating ambient particles to human health.

This section follows the eight steps outlined in the protocol for AQG level development according to scenario 1 for short-term AQG levels (section 2.5.2). Tables and figures mentioned during the eight steps are listed at the end of the discussion of each recommendation.

Step 1. Assess RR estimates and, when available, CRFs

The systematic review by Orellano et al. (2020) on PM_{2.5} and all-cause non-accidental mortality reported a meta-analytic effect estimate of RR of 1.0065 (95% CI: 1.0044–1.0086) per 10 µg/m³ PM_{2.5}, assuming a linear relationship. The certainty of the evidence was considered high according to GRADE. The authors found an indication of a supralinear relationship, suggesting a steeper risk increase at lower exposure levels. CRFs were provided by several studies. Examples show that the associations persist to very low levels of exposure (see Fig. 5A of the original study (Di et al., 2017b) and Fig. 3.6 of this document (taken from Liu et al. (2019))).

Step 2. Determine the lowest level of exposure measured

As discussed in the protocol for deriving AQG levels in section 2.5, the lowest concentrations in time-series studies of the effects of daily variations in air pollution concentrations are often very low. Therefore, the 5th percentiles of these daily distributions cannot be used as starting points for AQG level development. In such cases, the protocol suggests identifying the 99th percentile of common distributions of daily air pollution concentrations corresponding to an average long-term concentration equivalent to the annual AQG level. Thus, it is expected that daily means will be higher than the short-term AQG level not more than three to four times per year once air quality complies with the proposed annual mean AQG level. The proposed annual mean AQG level is 5 µg/m³ for PM_{2.5}. Common distributions observed in large numbers of cities around the world (data from Liu et al. (2019)) suggest that the 99th percentiles of daily concentrations are about three times higher than the annual mean PM_{2.5} concentration.

Step 3. Determine the minimal relevant increase in health outcomes

The GDG decided to consider as relevant any increase in risk for an adverse health outcome related to long-term exposure to a pollutant. For short-term exposures, the linear CRFs from the systematic review by Orellano et al. (2020) were used to calculate the increase in mortality expected on a day with a PM_{2.5} concentration of 15 µg/m³, compared with a day with a PM_{2.5} concentration of 5 µg/m³. With an RR for all non-accidental mortality of 1.0065 per 10 µg/m³, the estimated excess mortality on such a day would be 0.65%. For locations in which concentrations are below the annual mean AQG level, days with such high daily mean concentrations will be rare and most days will have concentrations below the annual mean AQG level. Thus, the health burden related to a few days with higher concentrations corresponds to a very small fraction of the total air pollution-related burden. The GDG notes that at higher concentrations, the CRFs may no longer be linear but sublinear (e.g. see Liu et al. (2019)) so that the excess mortality will be overestimated by using a linear function.

Step 4. Determine the starting point for AQG level determination as the 99th percentile, as mentioned in step 3

The data presented in the previous three steps support a short-term AQG level of no more than 15 $\mu\text{g}/\text{m}^3$, based on the association between short-term $\text{PM}_{2.5}$ and all-cause non-accidental mortality.

Step 5. Compare the AQG level across critical health outcomes: cause-specific mortality

The cause-specific mortality outcomes that were investigated all yielded bigger RRs for $\text{PM}_{2.5}$ compared with the RR for all-cause mortality, with RRs of 1.0092 (95% CI: 1.0061–1.0123) per 10 $\mu\text{g}/\text{m}^3$ for cardiovascular mortality, 1.0073 (95% CI: 1.0029–1.0016) for non-malignant respiratory mortality and 1.0072 (95% CI: 1.0012–1.0132) for cerebrovascular mortality. The certainty of the evidence was rated as high for cardiovascular mortality and moderate for both non-malignant respiratory mortality and cerebrovascular mortality. With these RRs for cause-specific mortality per 10 $\mu\text{g}/\text{m}^3$, the estimated excess mortality on such a day would be 0.72–0.92% for $\text{PM}_{2.5}$. The same considerations apply as for all-cause non-accidental mortality (as discussed in step 3). The data obtained for cause-specific mortality also support a short-term AQG level of no more than 15 $\mu\text{g}/\text{m}^3$ for $\text{PM}_{2.5}$.

Step 6. Assess certainty of the evidence

As mentioned in step 1, the certainty of the evidence linking short-term PM concentration variations to short-term mortality variations is high. In addition, as shown in Fig. 5A of Di et al. (2017b), there is evidence that this association persists to very low levels of exposure.

Step 7. Consider new evidence

Several new studies have been published since the autumn of 2018. Only one of these (the 652 cities study by Liu et al. (2019)) is discussed in the systematic review by Orellano et al. (2020). The results of this new, very large study were in line with those of the systematic review. A full search of studies reported since autumn 2018 was not done nor has been reported. As dozens of studies were already included in the systematic review by Orellano et al. (2020) and the Liu et al. (2019) study showed similar results, the GDG does not expect that inclusion of the new studies would change the assessment of the systematic review.

Step 8. Reconsider causality

All PM–outcome associations were deemed to be causal or likely causal in the 2016 outcome prioritization framework (see [section 2.3.3](#)). These judgements have not changed in more recent authoritative assessments.

3.2.3.1 Interim targets

Interim targets are proposed as incremental steps in a progressive reduction of air pollution and are intended for use in areas where pollution is high. For a more detailed rationale for establishing and using interim targets, see [section 2.5.3](#).

The recommendation is a short-term (24-hour) PM_{2.5} AQG level of 15 µg/m³, defined as the 99th percentile (equivalent to 3–4 exceedance days per year) of the annual distribution of 24-hour average concentrations.

The GDG recommends maintaining the 2005 interim targets and introducing an interim target 4 at the level of the 2005 air quality guideline, as shown in [Table 3.6](#).

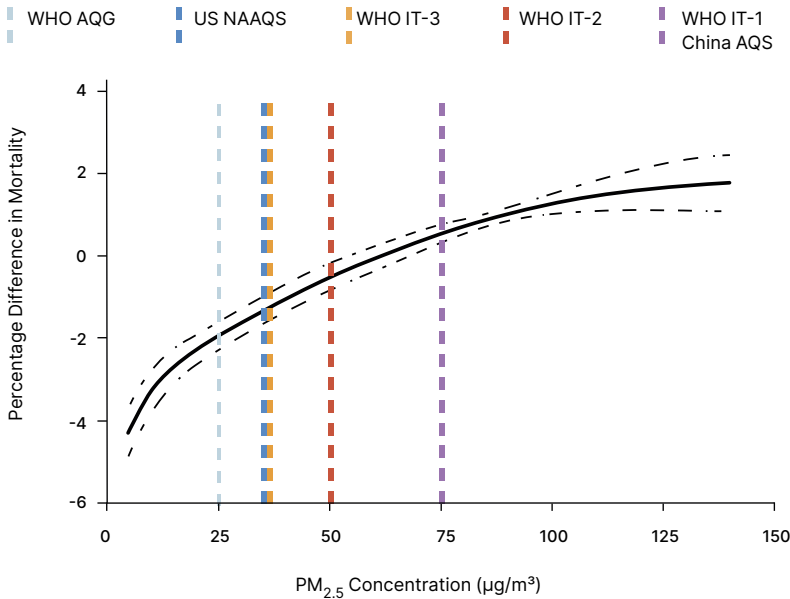
Table 3.6. Recommended short-term (24-hour) AQG level and interim targets for PM_{2.5}^a

Recommendation	PM _{2.5} (µg/m ³)
Interim target 1	75
Interim target 2	50
Interim target 3	37.5
Interim target 4	25
AQG level	15

^a Defined as the 99th percentile of the annual distribution of 24-hour average concentrations (equivalent to 3–4 exceedance days per year).

If mortality in a population exposed to PM_{2.5} at the AQG level is arbitrarily set at 100, then it will be 104, 102, 101 and 101, respectively, in populations exposed at PM_{2.5} at interim target 1, 2, 3 and 4 levels. These projections are based on the linear HR of 1.0065 per 10-µg/m³ increase in PM_{2.5} for all non-accidental mortality reported in the systematic review. At higher concentrations, the CRF may no longer be linear, which would change the numbers in this example.

Fig. 3.6. CRF of 24-hour average PM_{2.5} concentrations (µg/m³) and daily all-cause mortality, as observed in a joint analysis of data from 652 cities worldwide^a



AQG: Air Quality Guidelines; AQG: Air Quality Standard; EU AQD: European Union Air Quality Directive; IT-1: interim target 1; IT-2: interim target 2; IT-3: interim target 3; US NAAQS: United States National Ambient Air Quality Standard.

^a The y-axis represents the percentage difference from the pooled mean effect on mortality (as derived from the entire range of PM concentrations at each location). Zero on the y-axis represents the pooled mean effect, and the portion of the curve below zero denotes a smaller estimate than the mean effect.

Source: reprinted from Liu et al. (2019) with permission from the Massachusetts Medical Society. Copyright © 2019 Massachusetts Medical Society.

3.3 PM₁₀

3.3.1 Recommended AQG level for long-term exposure to PM₁₀

Based on the methods for deriving an AQG level outlined in the guideline development protocol in [Chapter 2](#), this section provides a recommended AQG level for long-term PM₁₀ that is based on non-accidental mortality and cause-specific mortality ([Table 3.7](#)).

The epidemiological evidence underpinning the AQG level is discussed in a systematic review commissioned by WHO, as explained in more detail in [section 2.4](#). The review (Chen & Hoek, 2020) was published in *Environment International* (Whaley et al., 2021) as open access.

As discussed in [section 2.3](#), there has been no separate, independent assessment of the mechanistic, toxicological and human clinical studies relating ambient particles to human health.

This section follows the eight steps outlined in the protocol for AQG level development. Tables and figures mentioned during the eight steps are listed at the end of the discussion of each recommendation.

Step 1. Assess RR estimate and, when available, CRFs

The systematic review by Chen & Hoek (2020) on PM₁₀ and all non-accidental mortality reported a meta-analytic effect estimate of RR = 1.04 (95% CI: 1.03–1.06) per 10 µg/m³ PM₁₀, assuming a linear relationship.

The certainty of the evidence was considered high according to GRADE. Only one study (Fischer et al., 2015) provided a CRF; it concluded that the association between PM₁₀ and non-accidental mortality did not deviate significantly from linear (Fig. 3.7).

Step 2. Determine the lowest level of exposure measured

For 13 of the 17 studies included in the meta-analysis, the 5th percentile of the exposure distribution was reported or could be calculated from the reported mean and standard deviation. As the concentration distributions are often lognormal, this calculation is not straightforward. In all cases where a 5th percentile was reported in the paper or obtained from the study authors upon request, the GDG gave preference to that number (see Table 3.8). The five lowest levels reported or estimated in these studies were 13.7 µg/m³ (Beelen et al., 2014), 15.0 µg/m³ (Bentayeb et al., 2015), 15.1 µg/m³ (Puett et al., 2008), 15.9 µg/m³ (Carey et al., 2013) and 16.0 µg/m³ (Hart et al., 2011). The average 5th percentile across the five studies with the lowest concentrations was 15.1 µg/m³. The sum of weights in the meta-analysis was 21% for the lowest five studies, indicating that they made a significant contribution to the effect estimate from the meta-analysis. All of these studies had positive effect estimates with lower confidence limits of 1.00 or more.

Step 3. Determine the minimal relevant increase in health outcomes

The GDG decided to consider as relevant any increase in risk for an adverse health outcome related to long-term exposure to a pollutant.

Step 4. Determine the starting point for AQG level determination as the long-term concentration of the pollutant from which the minimal relevant amount of the health outcome will result

The average of the five lowest 5th percentile levels measured in these five studies was the starting point for deriving a AQG level: 15.1 µg/m³ PM₁₀.

The data obtained so far support a long-term AQG level of no more than 15 µg/m³, based on the association between long-term PM₁₀ and all non-accidental mortality.

Step 5. Compare the AQG level across critical health outcomes: cause-specific mortality

The RRs estimated by the review of Chen & Hoek (2020) meta-analysis for effects of PM₁₀ exposure were 1.06 (95% CI: 1.01–1.10) for IHD, 1.12 (95% CI: 1.06–1.19) for respiratory and 1.08 (95% CI: 1.04–1.13) for lung cancer mortality, all per 10 µg/m³. The certainty of the evidence was considered high for respiratory mortality and lung cancer mortality and moderate for IHD mortality, according to GRADE. For the remaining causes of mortality considered (circulatory, COPD and stroke mortality), the estimates of RR exceeded 1 but with 95% CIs that included 1. Most of the studies addressing cause-specific mortality were based on the same populations as the studies of all non-accidental mortality. For the few studies based on different populations, PM₁₀ exposure levels were higher than in those used to derive the starting point for AQG level. Therefore, there is no evidence from cause-specific mortality studies supporting a decrease of the AQG level below that suggested by all non-accidental cause mortality studies.

Step 6. Assess certainty of the evidence

None of the studies that reported the lowest levels measured in the studies of all non-accidental mortality were considered at high RoB; thus, there is no reason to change the AQG level because of low certainty of the evidence in the lowest level studies.

Step 7. Consider new evidence

Two new studies were published between autumn 2018 and the summer of 2020 (Fischer et al., 2020; Hvidtfeldt et al., 2019). They are discussed in Chen & Hoek (2020). The effect estimates for PM₁₀ (RR = 1.12 (95% CI: 1.09–1.14) and RR = 1.12 (95% CI: 1.03–1.22) respectively) were higher in those studies than the estimates from the meta-analysis of earlier studies, but the PM₁₀ exposure levels were higher than those in the studies selected to support the derivation of the AQG level. Therefore, this new evidence does not change the recommended AQG level for long-term PM₁₀ concentrations.

Step 8. Reconsider causality

All PM–outcome associations were deemed to be causal or likely causal in the 2016 outcome prioritization framework (see [section 2.3.3](#)). These judgements have not changed in more recent authoritative assessments. For more details, see [Table 2.1](#) in [section 2.3.3](#).

The 5th percentile and mean or median of the exposure distributions in studies on PM₁₀ and mortality meta-analysis is indicated in [Table 3.8](#) based on data from the systematic review by Chen & Hoek (2020).

3.3.1.1 Interim targets

Interim targets are proposed as incremental steps in a progressive reduction of air pollution and are intended for use in areas where pollution is high. For a more detailed rationale for establishing and using interim targets, see [section 2.5.3](#).

The recommendation is an annual PM₁₀ AQG level of 15 µg/m³. This is based on an evaluation of the studies on the long-term effects of PM₁₀ on mortality only, without taking into consideration that a large proportion of PM₁₀ is made up of PM_{2.5}. As in most situations PM_{2.5} is about 50–80% of PM₁₀ by weight, the annual PM₁₀ AQG level of 15 µg/m³ is less protective than the annual AQG level for PM_{2.5}. In all situations where both PM_{2.5} and PM₁₀ measurements are available, preference should be given to the PM_{2.5} AQG level.

The GDG recommends maintaining the 2005 interim targets and introducing an interim target 4 at the level of the 2005 air quality guideline, as shown in [Table 3.7](#).

Table 3.7. Recommended annual mean AQG level and interim targets for PM₁₀

Recommendation	PM ₁₀ (µg/m ³)
Interim target 1	70
Interim target 2	50
Interim target 3	30
Interim target 4	20
AQG level	15

If mortality in a population exposed to PM₁₀ at the AQG level were arbitrarily set at 100, then it will be 122, 114, 106 and 102, respectively, in populations exposed to PM₁₀ at the interim target 1, 2, 3 and 4 levels. These projections are based on the linear HR of 1.04 per 10-µg/m³ increase in PM₁₀ for all non-accidental mortality reported in the systematic review. At higher concentrations, the CRF may no longer be linear, which would change the numbers in this example.

Table 3.8. Studies on long-term PM₁₀ exposure and all non-accidental mortality included in the systematic review by Chen & Hoek (2020), ordered by me(di)an concentration

Study	Me(di)an (µg/m ³)	SD	P5	P25	HR (95% CI) ^a
Carey et al. (2013)	19.7	2.3	15.9 ^b	–	1.07 (1.00–1.14)
Hansell et al. (2016) ^c	20.7	2.5	16.5 ^b	–	1.24 (1.15–1.32)
Beelen et al. (2014)	20.9	–	13.7 ^b	17.1	1.04 (1.00–1.09)
Puett et al. (2008)	21.6	4.3	15.1 ^b	–	1.16 (1.05–1.28)
Bentayeb et al. (2015)	25.0	5.5	15.0 ^b	–	1.18 (1.06–1.32)
Hart et al. (2011)	26.8	6.0	16.0 ^b	–	1.07 (1.02–1.11)
Puett et al. (2011)	27.9	5.8	19.1 ^b	–	0.92 (0.84–0.99)
Dockery et al. (1993)	28.9	–	–	–	1.09 (1.03–1.15)
Fischer et al. (2015)	29.0	–	24.0 ^b	–	1.08 (1.07–1.09)
Lipsett et al. (2011)	29.2	9.7	18.2 ^b	–	1.00 (0.97–1.04)
Ueda et al. (2012)	34.9	–	–	–	0.98 (0.92–1.04)
Badaloni et al. (2017)	36.6	5.1	28.2 ^d	–	1.02 (1.01–1.03)
Heinrich et al. (2013)	43.7	–	–	39.8	1.22 (1.06–1.41)
Abbey et al. (1999)	51.2	16.6	23.9 ^d	–	1.01 (0.94–1.08)
Kim, Kim & Kim (2017)	56.0	6.5	45.3 ^d	–	1.05 (0.99–1.11)
Zhou et al. (2014)	104.0	–	–	–	1.02 (1.01–1.03)
Chen et al. (2016)	144.0	3.6	–	126.0	1.01 (1.01–1.01)

–, data unavailable; P5: 5th percentile (of the distribution of concentrations assigned to study participants); P25: 25th percentile; SD: standard deviation.

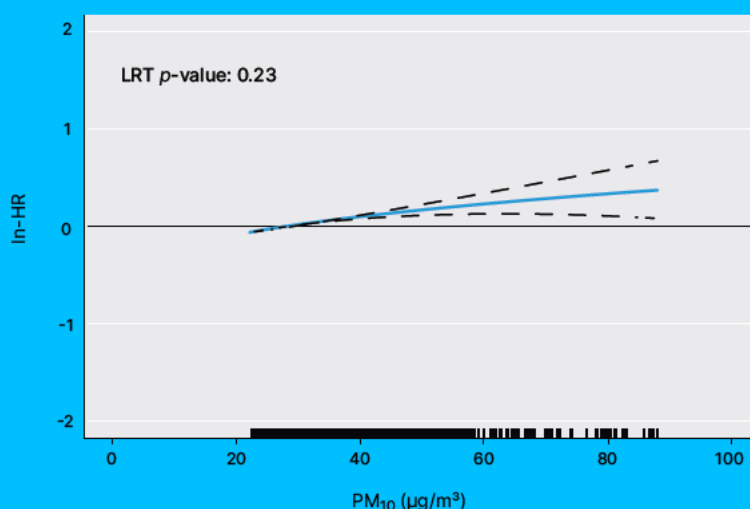
^a Per 10 µg/m³.

^b Reported in paper or by authors.

^c Study classified as having high RoB due to potentially insufficient control for confounding.

^d Calculated from mean and standard deviation using the following formula: Me(di)an – 1.645 × SD.

Fig. 3.7. Estimated concentration–response curve for non-accidental mortality and annual PM_{10} exposure ($\mu\text{g}/\text{m}^3$)



In: natural logarithm; LRT: likelihood ratio test.

Note: Solid blue line: estimated concentration–response curve; dashed lines: 95% CIs.

Source: reproduced from Fischer et al. (2015) with the permission of the lead author.

3.3.2 Recommended AQG level for short-term exposure to PM_{10}

Based on the methods for deriving an AQG level outlined in the guideline development protocol in [Chapter 2](#), this section provides a recommended AQG level for short-term, 24-hour average PM_{10} that is based on all-cause non-accidental mortality and cause-specific mortality ([Table 3.9](#)).

The epidemiological evidence underpinning the AQG level is discussed in a systematic review commissioned by WHO, as explained in more detail in [section 2.4](#). The review (Orellano et al., 2020) was published in *Environment International* (Whaley et al., 2021) as open access.

As discussed in [section 2.3](#), there has been no separate, independent assessment of the mechanistic, toxicological and human clinical studies relating ambient particles to human health.

This section follows the eight steps outlined in the protocol for AQG level development. Tables and figures mentioned during the eight steps are listed at the end of the discussion of each recommendation.

Step 1. Assess RR estimates and, when available, CRFs

The systematic review by Orellano et al. (2020) on PM₁₀ and all-cause non-accidental mortality reported a meta-analytic effect estimate of RR = 1.0041 (95% CI: 1.0034–1.0049) per 10 µg/m³ PM₁₀, assuming a linear relationship. The evidence was considered to be of high certainty according to GRADE. The authors found an indication of a supralinear relationship, suggesting a steeper risk increase at lower exposure levels. In contrast to PM_{2.5}, no individual studies published graphical representations of CRFs.

Step 2. Determine the lowest level of exposure measured

As discussed in the protocol for deriving AQG levels, the lowest concentrations in time-series studies of effects of daily variations in air pollution concentrations are often very low. Therefore, the 5th percentiles of these daily distributions cannot be used as starting points for AQG level development. In such cases, the protocol suggests identifying the 99th percentile of common distributions of daily air pollution concentrations corresponding to an average long-term concentration equivalent to the annual AQG level. Thus, once the air quality complies with the proposed annual mean AQG level, daily means would be expected to be higher than the short-term AQG level not more than three to four times per year. The proposed annual mean AQG level is 15 µg/m³ for PM₁₀. Common distributions observed in large numbers of cities around the world (data from Liu et al. (2019)) suggest that the 99th percentiles of daily concentrations are about three times higher than the annual mean PM₁₀ concentration.

Step 3. Determine the minimal relevant increase in health outcomes

The GDG decided to consider as relevant any increase in risk for an adverse health outcome related to long-term exposure to a pollutant. For short-term exposures, the CRFs from the systematic review by Orellano et al. (2020) were used to calculate the increase in mortality expected on a day with a PM₁₀ concentration of 45 µg/m³ compared with a day with a PM₁₀ concentration of 15 µg/m³. With an RR for all-cause mortality of 1.0041 per 10 µg/m³, the estimated excess mortality on such a day would be 1.23%. Under compliance with the annual mean AQG level, days with such high daily mean concentrations will be rare and most days will have concentrations below the annual mean AQG level. Thus, the health burden related to a few days with higher concentrations corresponds to a very small fraction of the total air pollution-related burden.

Step 4. Determine the starting point for AQG level determination as the 99th percentile, as mentioned in step 3

The data obtained support a short-term AQG level of no more than 45 µg/m³, based on the association between short-term PM₁₀ and all-cause non-accidental mortality.

Step 5. Compare the AQG level across critical health outcomes: cause-specific mortality

All cause-specific mortality outcomes that were investigated yielded slightly bigger RRs for PM₁₀ compared with the RR for all-cause mortality. The certainty of the evidence was rated as high for cardiovascular mortality and moderate for cerebrovascular mortality and non-malignant respiratory mortality. The data obtained for cause-specific mortality also support a short-term AQG level of no more than 45 µg/m³ for PM₁₀.

Step 6. Assess certainty of the evidence

As mentioned in step 1, the evidence linking short-term PM concentration variations to short-term mortality variations was of high certainty.

Step 7. Consider new evidence

The GDG noted that several new time-series studies, almost all from Asia, were published after the inclusion deadline of September 2018. A full search of studies reported since autumn 2018 was not done or has not been reported. As dozens of studies were already included in the systematic review by Orellano et al. (2020), the GDG did not expect that inclusion of new studies would change the assessment of the systematic review.

Step 8. Reconsider causality

All PM–outcome associations were deemed to be causal or likely causal in the 2016 outcome prioritization framework (see [section 2.3.3](#)). These judgements have not changed in more recent authoritative assessments.

3.3.2.1 Interim targets

Interim targets are proposed as incremental steps in a progressive reduction of air pollution and are intended for use in areas where pollution is high. For a more detailed rationale for establishing and using interim targets, see [section 2.5.3](#).

The recommendation is a short-term (24-hour) PM₁₀ AQG level of 45 µg/m³, defined as the 99th percentile (equivalent to three to four exceedance days per year) of the annual distribution of 24-hour average concentrations.

The GDG recommends maintaining the 2005 interim targets and introducing an interim target 4 at the level of the 2005 air quality guideline, as shown in [Table 3.9](#).

Table 3.9. Recommended short-term (24-hour) AQG level and interim targets for PM₁₀^a

Recommendation	PM ₁₀ (µg/m ³)
Interim target 1	150
Interim target 2	100
Interim target 3	75
Interim target 4	50
AQG level	45

^a Defined as the 99th percentile of the annual distribution of 24-hour average concentrations (equivalent to 3–4 exceedance days per year).

If mortality in a population exposed to PM₁₀ at the AQG level is arbitrarily set at 100, then it will be 104, 102, 101 and 100.2, respectively, in populations exposed to PM₁₀ at the interim target 1, 2, 3 and 4 levels. These projections are based on the linear HR of 1.0041 per 10-µg/m³ increase in PM₁₀ for all non-accidental mortality reported in the systematic review. At higher concentrations, the CRF may no longer be linear, which would change the numbers in this example.

3.4 Ozone

3.4.1 General description

The general description comes from *Global update 2005*.

Ozone (O₃) and other photochemical oxidants are pollutants that are not directly emitted by primary sources. Rather, they encompass a group of chemical species formed through a series of complex reactions in the atmosphere driven by the energy transferred to nitrogen dioxide (NO₂) molecules when they absorb light from solar radiation

The precursors that contribute most to the formation of oxidant species in polluted atmospheres are nitrogen dioxide and non-methane volatile organic compounds (VOCs), especially unsaturated VOCs. Methane is much less reactive than the other VOCs but is present at much higher concentrations, having risen in concentration over the past 100 years owing to its increasing use as fuel, and is released from rice fields and farm animals. Photochemistry involving methane accounts for much of the rise in ozone over the oceans and remote land areas, from about 30 µg/m³ to about 75 µg/m³ (WHO Regional Office for Europe, 2006).

Conversion factors for ozone: at 20 °C and 1013 hPa, 1 part per million (ppm) = 1.9957 mg/m³ and 1 mg/m³ = 0.5011 ppm.

3.4.2 Recommended AQG level for long-term exposure to ozone

Based on the methods for deriving an AQG level outlined in the guideline development protocol, this section provides an AQG level for long-term, peak-season ozone that is based on all non-accidental mortality and respiratory mortality ([Table 3.10](#)).

The epidemiological evidence underpinning the AQG level is discussed in a systematic review commissioned by WHO, as explained in more detail in [section 2.4](#). The review (Huangfu & Atkinson, 2020) was published in *Environment International* (Whaley et al., 2021) as open access.

As discussed in [section 2.3](#), there has been no separate, independent assessment of the mechanistic, toxicological and human clinical studies relating ambient ozone to human health.

The long-term AQG level for ozone is linked to the so-called peak-season exposure. Peak season is defined as the six consecutive months of the year with the highest six-month running-average ozone concentration. In regions away from the equator, this period will typically be in the warm season within a single calendar year (northern hemisphere) or spanning two calendar years (southern hemisphere). Close to the equator, such clear seasonal patterns may not be obvious, but a running-average six-month peak season will usually be identifiable from existing monitoring or modelling data.

This section follows the eight steps outlined in the protocol for AQG level development. Tables and figures mentioned during the eight steps are listed at the end of the discussion of each recommendation.

Step 1. Assess RR estimates and, when available, CRFs

The systematic review by Huangfu & Atkinson (2020) on ozone and all non-accidental mortality reported a meta-analytic effect estimate of RR = 1.01 (95% CI: 1.00–1.02) per 10 µg/m³ increase in peak-season average of daily maximum 8-hour mean ozone concentrations, assuming a linear relationship. For ozone, it is customary to calculate daily maximum of 8-hour mean concentrations rather than 24-hour averages because of the strong diurnal variation in ozone concentration. In most of the quoted studies, peak season was defined as the warm season, that is, the warmest five or six months of the year, for example May–September in studies from Canada and April–September in several of the studies from the

United States. The certainty of the evidence was considered moderate according to GRADE. CRFs were provided in one study (Di et al., 2017a), which documented a linear function starting from the 5th percentile of the observed warm-season concentrations of about 60 $\mu\text{g}/\text{m}^3$ (Fig. 3.8). From the series of Canadian Census Health and Environment Cohort (CanCHEC) studies, the more recent Cakmak et al. (2018) study was included instead of the earlier study by Crouse et al. (2015), which did document a monotonic dose–response relationship (Fig. 3.9).

Step 2. Determine the lowest level of exposure measured

For all seven studies included in the meta-analysis, a 5th percentile of the exposure distribution was reported or could be calculated from the reported mean and standard deviation. As the concentration distributions are often lognormal, this calculation is not straightforward. Therefore, in most cases it was replaced by actual reports of the relevant numbers obtained from the study authors (for details, see Table 3.11 and Table 3.12). The three lowest 5th percentile concentrations reported or estimated in these studies were the peak-season averages of 55 $\mu\text{g}/\text{m}^3$ (Weichenthal, Pinault & Burnett, 2017), 56 $\mu\text{g}/\text{m}^3$ (Cakmak et al., 2018) and 68 $\mu\text{g}/\text{m}^3$ (Di et al., 2017a). The study by Weichenthal, Pinault & Burnett (2017) was considered in the systematic review to be at high RoB. If this study is ignored, then the next lowest 5th percentile concentration was 68 $\mu\text{g}/\text{m}^3$ (Lipsett et al., 2011). The average of the three lowest 5th percentile values is either approximately 60 or 64 $\mu\text{g}/\text{m}^3$ (depending on whether or not the study by Weichenthal, Pinault & Burnett (2017) is included). Three of these four studies found statistically significant positive associations between ozone and all non-accidental mortality. The sum of weights of these four studies in the meta-analysis was well over 60%.

Step 3. Determine the minimal relevant increase in health outcomes

The GDG decided to consider as relevant any increase in risk for an adverse health outcome related to long-term exposure to a pollutant.

Step 4. Determine the starting point for AQG level determination as the long-term concentration of the pollutant from which the minimal relevant amount of the health outcome will result

Thus, the average of the three lowest 5th percentile levels measured in these studies was the starting point for deriving an AQG level: 60 $\mu\text{g}/\text{m}^3$ ozone, based on the average concentrations of either 60 $\mu\text{g}/\text{m}^3$ or 64 $\mu\text{g}/\text{m}^3$. The data obtained support a long-term, peak-season AQG level of no more than 60 $\mu\text{g}/\text{m}^3$, based on the association between long-term ozone and all non-accidental mortality.

Step 5. Compare the AQG level across critical health outcomes: respiratory mortality

The other outcome that was investigated was respiratory mortality, which yielded a bigger RR for peak-season ozone, compared with the RR for all non-accidental mortality, with an RR of 1.02 (95% CI: 0.99–1.05) per 10 $\mu\text{g}/\text{m}^3$. The certainty of the evidence, however, was rated low for non-malignant respiratory mortality because the prediction interval of 0.96–1.08 included unity and was exactly twice the meta-analytic 95% CI. For an explanation of the prediction interval, see [section 2.4.4](#). In addition, because none of the studies had explicitly considered the shape of the CRF, no upgrade was applied for dose–response. [Table 3.12](#) shows the findings for non-malignant respiratory mortality. The starting points for AQG level determination for this additional health outcome would not be further supported by including respiratory mortality, although three of the four studies are included in the all non-accidental mortality analysis and the fourth is on the same cohort as all-cause mortality (Crouse et al. (2015) versus Cakmak et al. (2018)). For further discussion, see step 7.

Step 6. Assess certainty of the evidence

The certainty of the evidence was rated as moderate for non-accidental mortality and low for respiratory mortality. One of the studies that made up the lowest levels measured in all non-accidental mortality studies (Weichenthal, Pinault & Burnett, 2017) was considered at high RoB, so the GDG calculated the starting point for AQG level determination with and without that study, as previously mentioned.

Step 7. Consider new evidence

Several new studies were published between autumn 2018 and the summer of 2020. The systematic review discussed these but did not include them in the assessment, so the GDG made its own assessment of these studies. These new studies are largely the same as those identified and included in the revision of the systematic review of long-term PM effects on mortality (Chen & Hoek, 2020). [Table 3.13](#) shows these studies, ordered by mean or median exposure level for all non-accidental mortality. These include two studies from Canada (Brauer et al., 2019; Pappin et al., 2019) and three new studies from the United States (Lefler et al., 2019; Lim et al., 2019; Kazemiparkouhi et al., 2020). Two of the five were administrative database studies with no adjustment (Brauer et al., 2019) or with area-level adjustment (Kazemiparkouhi et al., 2020) for lifestyle factors such as smoking. The other three were cohort studies with adequate information on lifestyle covariates. Adding these studies to the meta-analysis produced an HR of 1.013 (95% CI: 1.002–1.023) for non-accidental mortality. The effect estimate from the systematic review was 1.01 (95% CI: 1.00–1.02; see step 1).

The Kazemiparkouhi et al. (2020) study was based on 1-hour maximum concentrations, not 8-hour maximum concentrations. The 8-hour maximum concentrations usually correlate very highly with the 1-hour maximum concentrations but are 10–40% lower. Therefore, in principle, one would expect effect estimates expressed over the same concentration range to be somewhat higher when using 8-hour maximum concentrations as the denominator. However, a large study from Europe (Gryparis et al., 2004) found no difference in effect estimates based on 1-hour versus 8-hour maximum concentrations and expressed over the same concentration range. Therefore, the GDG did not change the effect estimate from the Kazemiparkouhi et al. (2020) study. Adding these studies to the meta-analysis produced an HR of 1.013 (95% CI: 1.006–1.021) and a prediction interval of 0.997–1.030. For an explanation of the prediction interval, see [section 2.4.4](#). Note that this prediction interval includes unity and is slightly larger than twice the HR 95% CI, so this would justify a downgrade of the certainty of evidence due to inconsistency. As argued before, the GDG finds the evidence of dose–response sufficient for an upgrade of certainty, so that the net result for the association between peak-season ozone and non-accidental mortality would be moderate certainty.

Two cohort studies also reported effect estimates for respiratory mortality ([Table 3.14](#)). Adding these studies to the meta-analysis produced an HR for respiratory mortality of 1.023 (95% CI: 1.007–1.038) with a prediction interval of 0.993–1.053. As this prediction interval is less than twice the meta-analytic 95% CI, there is no need to downgrade the certainty of the evidence due to inconsistency. The effect estimate from the systematic review was an RR of 1.02 (95% CI: 0.99–1.05) per 10 µg/m³. In addition, as [Fig. 3.10](#) shows, one of the new studies (Lim et al., 2019) supports a dose–response for respiratory mortality down to slightly less than 60 µg/m³.

The GDG notes that these very recent studies almost doubled the number of studies available for inclusion. If they had been part of the review, the AQG level starting point based on the three lowest 5th percentile values, excluding the studies at high RoB, would be even somewhat lower, at $(50 + 56 + 62) / 3 = 56$ µg/m³. There is no reason, based on these new findings, to change the proposed long-term AQG level.

Step 8. Reconsider causality

The long-term ozone–outcome associations were deemed to be likely causal (for respiratory effects) or suggestive of being causal (for total mortality) in the 2016 outcome prioritization framework (see [section 2.3.3](#)). These judgements were primarily based on the 2013 US EPA ISA of ozone (US EPA, 2013) and a 2013 Health Canada report (Health Canada, 2013). The 2020 EPA ISA (US EPA,

2020) did not change these classifications. As discussed in step 7 and shown in [Table 3.13](#) and [Table 3.14](#), a number of very recent studies have provided further support for associations between long-term ozone concentrations and both total and respiratory mortality.

The 5th percentile and mean or median of exposure distributions in studies in the ozone and mortality meta-analyses are shown in [Table 3.11](#) and [Table 3.12](#) based on data from the systematic review by Huangfu & Atkinson (2020) and in [Table 3.13](#) and [Table 3.14](#) for the new studies that were identified.

3.4.2.1 Interim targets

Interim targets are proposed as incremental steps in a progressive reduction of air pollution and are intended for use in areas where pollution is high. For a more detailed rationale for establishing and using interim targets, see [section 2.5.3](#).

Interim targets were not specified for long-term ozone in *Global update 2005*. The GDG recommends a peak-season average ozone concentration of 100 µg/m³ as interim target 1, as this is a level already shown to be achievable in many parts of the world. As interim target 2, a concentration of 70 µg/m³ is proposed; this is the threshold in the widely used SOMO35 metric. SOMO35 is the accumulated ozone concentration (daily maximum 8-hour mean) in excess of 35 parts per billion (ppb; equivalent to 70 µg/m³) (EEA, 2020).

The recommendation is a peak season ozone AQG level of 60 µg/m³ (the average of daily maximum 8-hour mean ozone concentrations). The peak season is defined as the six consecutive months of the year with the highest six-month running-average ozone concentration. In regions away from the equator, this period will typically be in the warm season within a single calendar year (northern hemisphere) or spanning two calendar years (southern hemisphere). Close to the equator, such clear seasonal patterns may not be obvious, but a running-average six-month peak season will usually be identifiable from existing monitoring or modelling data.

An interim target 1 of 100 µg/m³ and an interim target 2 of 70 µg/m³ are proposed, as shown in [Table 3.10](#).

If mortality in a population exposed to ozone at the AQG level is arbitrarily set at 100, then it will be 104 and 101, respectively, in populations exposed to ozone at the interim target 1 and 2 levels. These projections are based on the linear HR of 1.01 per 10-µg/m³ increase in ozone of all non-accidental mortality reported in the systematic review. For respiratory mortality, the numbers will be 108 and 102, respectively, at the interim target 1 and 2 levels, based on the linear HR of 1.02 of respiratory mortality reported in the systematic review. At higher concentrations, the CRF may no longer be linear, which would change the numbers in this example.

Table 3.10. Recommended peak season^a AQG level and interim targets for ozone

Recommendation	O ₃ (µg/m ³)
Interim target 1	100
Interim target 2	70
AQG level	60

^a Average of daily maximum 8-hour mean O₃ concentration in the six consecutive months with the highest six-month running-average O₃ concentration.

Table 3.11. Studies on peak-season, long-term ozone exposure and all non-accidental mortality included in the systematic review by Huangfu & Atkinson (2020), ordered by me(di)an concentration

Study	Me(di)an (µg/m ³)	SD	P5	P25	HR (95% CI) ^a
Weichenthal, Pinault & Burnett (2017) ^b	76.6	–	55.2 ^c	67.3	1.0290 (1.024–1.033)
Cakmak et al. (2018)	78.4	13.4	56.4 ^d	–	1.0400 (1.010–1.070)
Di et al. (2017a)	90.0	14.0	68.0 ^c	–	1.0115 (1.011–1.012)
Turner et al. (2016)	94.2	11.8	71.4 ^c	88.4	1.0100 (1.010–1.015)
Lipsett et al. (2011)	96.2	17.4	67.6 ^d	–	0.9900 (0.990–1.000)
Bentayeb et al. (2015)	101.0	8.5	87.0 ^d	–	0.9800 (0.900–1.060)
Lipfert et al. (2006)	173.4	18.6	142.8 ^d	–	1.0000 (0.990–1.020)

–, data unavailable; P5: 5th percentile (of the distribution of concentrations assigned to study participants); P25: 25th percentile; SD: standard deviation.

^a Per 10 µg/m³.

^b Considered to be at high RoB.

^c Reported in paper or by authors on request.

^d Calculated from mean and standard deviation using the following formula: Me(di)an – 1.645 × SD.

Table 3.12. Studies on peak-season, long-term ozone exposure and respiratory mortality included in the systematic review by Huangfu & Atkinson (2020), ordered by me(di)an concentration

Study	Me(di)an ($\mu\text{g}/\text{m}^3$)	SD	P5	P25	HR (95% CI) ^a
Weichenthal, Pinault & Burnett (2017) ^b	76.6	–	55.2 ^c	67.3	1.020 (1.006–1.035)
Crouse et al. (2015)	78.0	–	56.0 ^d	68.6	0.985 (0.975–0.994)
Turner et al. (2016)	94.2	11.8	71.4 ^c	88.4	1.05 (1.035–1.060)
Lipsett et al. (2011)	96.2	17.4	67.6 ^e	–	1.02 (0.990–1.040)

–, data unavailable; P5: 5th percentile (of the distribution of concentrations assigned to study participants); P25: 25th percentile; SD: standard deviation.

^a Per 10 $\mu\text{g}/\text{m}^3$.

^b Considered to be at high RoB.

^c Reported in paper or by authors on request.

^d Similar distribution assumed as in the paper by Weichenthal, Pinault & Burnett (2017), based on the same CanCHEC cohort.

^e Calculated from mean and standard deviation using the following formula: Me(di)an – 1.645 × SD.

Table 3.13. New studies on peak-season, long-term ozone exposure and all non-accidental mortality published since autumn 2018, ordered by me(di)an concentration

Study	Me(di)an ($\mu\text{g}/\text{m}^3$)	SD	P5	P25	HR (95% CI) ^a
Brauer et al. (2019) – CanCHEC subjects	72.0	15.0	52.3 ^b	–	1.036 (1.034–1.036)
Brauer et al. (2019) – CCHS subjects	72.0	15.0	50.0 ^b	–	1.025 (1.015–1.035)
Lim et al. (2019)	92.4	15.2	62.3 ^b	–	1.000 (0.995–1.005)
Lefler et al. (2019)	94.9	10.6	77.5 ^c	–	1.016 (1.010–1.022)
Kazemiparkouhi et al. (2020)	110.0	–	–	100.0	1.006 (1.006–1.007)

–, data unavailable; CCHS: Canadian Community Health Survey; P5: 5th percentile (of the distribution of concentrations assigned to study participants); P25: 25th percentile; SD: standard deviation.

^a Per 10 $\mu\text{g}/\text{m}^3$.

^b Reported in paper or by authors on request.

^c Calculated from mean and standard deviation using the following formula: Me(di)an – 1.645 × SD.

Table 3.14. New studies on peak-season, long-term ozone exposure and respiratory mortality published since autumn 2018, ordered by me(di)an concentration

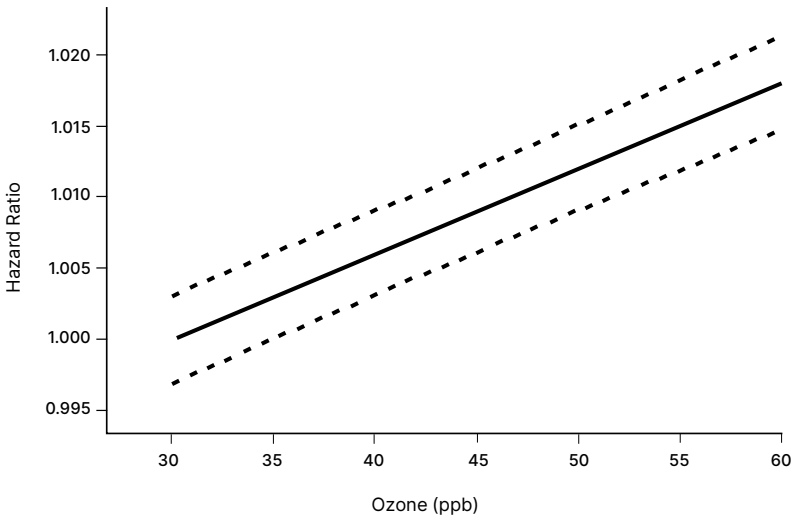
Study	Me(di)an (µg/m³)	SD	P5	P25	HR (95% CI) ^a
Lim et al. (2019)	92.4	15.2	62.3 ^b	–	1.040 (1.020–1.060)
Kazemiparkouhi et al. (2020)	110.0	–		100.0	1.018 (1.016–1.020)

–, data unavailable; P5: 5th percentile (of the distribution of concentrations assigned to study participants); P25: 25th percentile; SD: standard deviation.

^a Per 10 µg/m³.

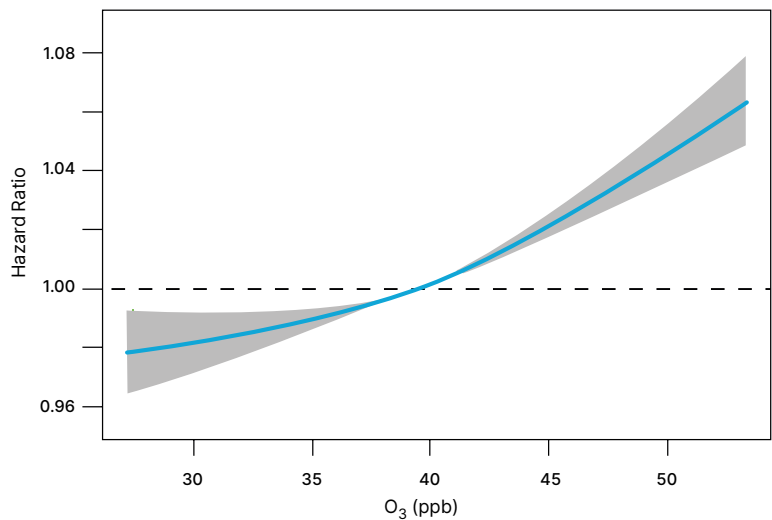
^b Reported in paper or by authors on request.

Fig. 3.8. Association between peak-season, long-term ozone exposure (ppb) and all non-accidental mortality^a



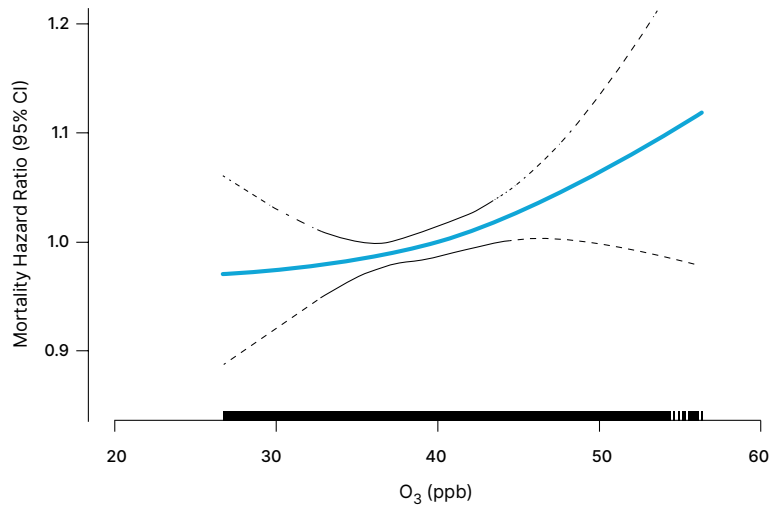
^a Note that the units for ozone are in ppb; these need to be multiplied by 2 to arrive at concentrations expressed in µg/m³. HR is expressed relative to the 5th percentile of the distribution of ozone concentrations, which was 30 ppb. Source: reprinted from Di et al. (2017a) with permission from the Massachusetts Medical Society. Copyright © 2017 Massachusetts Medical Society.

Fig. 3.9 The association between peak-season, long-term ozone exposure (ppb) and all-cause mortality^a



^a Note that the units for ozone are in ppb; these need to be multiplied by 2 to arrive at concentrations expressed in $\mu\text{g}/\text{m}^3$. HRs are expressed relative to the mean ozone concentration of 39.6 ppb.
Source: reproduced from Crouse et al. (2015) with permission of the lead author.

Fig. 3.10 The association between peak-season, long-term ozone exposure (ppb) and respiratory mortality^a



^a Note that the units for ozone are in ppb; these need to be multiplied by 2 to arrive at concentrations expressed in $\mu\text{g}/\text{m}^3$. HRs are expressed relative to the mean ozone concentration of 46.2 ppb.
Source: adapted from Lim et al. (2019) with permission of the American Thoracic Society. Copyright © 2019 American Thoracic Society. All rights reserved. Note that the authors, editors and the American Thoracic Society are not responsible for errors or omissions in adaptations.

3.4.3 Recommended AQG level for short-term exposure to ozone

Based on the methods for deriving an AQG level outlined in the guideline development protocol, this section provides an AQG level for short-term, daily maximum 8-hour average ozone that is based on all-cause non-accidental mortality ([Table 3.15](#)).

The epidemiological evidence underpinning the AQG level is discussed in a systematic review commissioned by WHO, as explained in more detail in [section 2.4](#). The review (Orellano et al., 2020), was published in *Environment International* (Whaley et al., 2021) as open access.

As discussed in [section 2.3](#), there has been no separate, independent assessment of the mechanistic, toxicological and human clinical studies relating ozone to human health. However, comprehensive evaluations by authoritative bodies such Health Canada, the United Kingdom's Committee on Medical Effects of Air Pollution and US EPA were taken into account in the development of the AQG levels. This was especially relevant when assessing causality of the associations examined in the systematic reviews (see step 8).

This section follows the eight steps outlined in the protocol for AQG level development. Tables and figures mentioned during the eight steps are listed at the end of the discussion of each recommendation.

Step 1. Assess RR estimates and, when available, CRFs

The systematic review by Orellano et al. (2020) on ozone and all-cause non-accidental mortality reported a meta-analytic effect estimate of $RR = 1.0043$ (95% CI: 1.0034–1.0052) per $10 \mu\text{g}/\text{m}^3$ ozone, assuming a linear relationship. This effect estimate is for 8-hour maximum concentrations. The certainty of the evidence was considered high according to GRADE. CRFs were provided by several studies. Many studies have found that associations persisted at daily levels of $100 \mu\text{g}/\text{m}^3$ ozone or lower. An example is provided in Fig. 5B of the original study (Di et al., 2017b), which was a very large study conducted in the United States of the entire Medicare population. Another example is from the multicity study by Vicedo-Cabrera et al. (2020), which was published after the systematic review search was completed ([Fig. 3.11](#)). This was a worldwide study combining evidence from 406 locations in 20 countries.

Step 2. Determine the lowest level of exposure measured

As discussed in the protocol for deriving AQG levels, the lowest concentrations in time-series studies of effects of daily variations in air pollution concentrations are often very low.

Therefore, the 5th percentiles of these daily distributions cannot be used as starting points for AQG level development.

In such cases, the protocol suggests identifying the 99th percentile of common distributions of daily air pollution concentrations corresponding to an average long-term concentration equivalent to the annual AQG level. The proposed long-term AQG level is $60 \mu\text{g}/\text{m}^3$ for ozone, as a warm-season average of daily maximum 8-hour concentrations. Common distributions observed in large numbers of cities around the world (data from Vicedo-Cabrera et al. (2020)) suggest that the 99th percentiles of daily concentrations are on average 2.05 (rounded to 2) times higher than the annual mean ozone concentrations. However, the long-term AQG level for ozone is for a peak-season average, which is always higher than the annual average. Note that the definitions of peak season and warm season vary slightly from study to study, sometimes restricted to the three summer months, sometimes using the (northern hemisphere) May–September period. A study from the United States (Turner et al., 2016) observed an annual mean of modelled daily 8-hour maximum ozone concentrations of $76.4 \mu\text{g}/\text{m}^3$ and a warm-season mean of $94.2 \mu\text{g}/\text{m}^3$ (ratio of 1.23). A very large database from Europe documented a ratio of 1.24 based on actual ozone measurements (de Hoogh et al., 2018). Therefore, using this ratio, the chosen peak-season AQG level of $60 \mu\text{g}/\text{m}^3$ corresponds to an annual mean of $48.7 \mu\text{g}/\text{m}^3$. Calculating the short-term AQG level using a ratio of 2 between the 99th percentile and annual mean produced a value of $120 \mu\text{g}/\text{m}^3$, and dividing that number by the 1.24 ratio of the peak (warm) season to annual average concentrations produced a value of $97 \mu\text{g}/\text{m}^3$, which was rounded up to a proposed short-term AQG level of $100 \mu\text{g}/\text{m}^3$.

Step 3. Determine the minimal relevant increase in health outcomes

The GDG decided to consider as relevant any increase in risk for an adverse health outcome related to long-term exposure to a pollutant. For short-term exposures, the CRFs from the systematic review by Orellano et al. (2020) were used to calculate the increase in mortality expected on a day with an 8-hour maximum ozone concentration of $100 \mu\text{g}/\text{m}^3$ compared with a day with an 8-hour maximum ozone concentration of $60 \mu\text{g}/\text{m}^3$. With an RR for all-cause mortality of 1.0043 per $10 \mu\text{g}/\text{m}^3$, the estimated excess mortality on such a day would be 1.72%. However, under compliance with the long-term peak-season AQG level, days with concentrations close to $100 \mu\text{g}/\text{m}^3$ will correspond to the far upper tail of the distribution of daily exposures. Most days will have much lower values and almost half will have concentrations below or far below the peak-season AQG level. The health burden related to a few days with higher concentrations corresponds to a very small fraction of the total air pollution-related burden.

Step 4. Determine the starting point for AQG level determination as the 99th percentile, as mentioned in step 3

The data obtained support a short-term AQG level of no more than 100 $\mu\text{g}/\text{m}^3$, based on the association between short-term ozone and all-cause non-accidental mortality.

Step 5. Compare the AQG level across critical health outcomes: cause-specific mortality and asthma hospital admissions and emergency room visits

Studies on short-term associations and cause-specific mortality were not reviewed. However, another systematic review assessed the evidence for associations between ozone and daily hospital and emergency room admissions for asthma (Zheng et al., 2021). The review found an effect estimate of $\text{RR} = 1.012$ (95% CI: 1.008–1.016) per 10 $\mu\text{g}/\text{m}^3$, which would produce an excess morbidity of 4.8% for a day at the proposed short-term AQG level of 100 $\mu\text{g}/\text{m}^3$ compared with a day at the proposed long-term AQG level of 60 $\mu\text{g}/\text{m}^3$. As mentioned in step 3, such days will be rare events under compliance with the peak-season long-term AQG level; thus, the short-term burden due to the few days with higher values is relatively small.

Step 6. Assess certainty of the evidence

As mentioned in step 1, the certainty level is high for evidence linking short-term ozone concentration variations to short-term mortality variations. In addition, as shown in Fig. 5B of Di et al. (2017b) and [Fig. 3.11](#), there is evidence that this association persists to very low levels of exposure.

Step 7. Consider new evidence

Several new studies have been published since autumn 2018. Of note is the very large study conducted by Vicedo-Cabrera et al. (2020). This study reported an effect estimate of $\text{RR} = 1.0018$ (95% CI: 1.0012–1.0024) per 10 $\mu\text{g}/\text{m}^3$, which is considerably lower than the RR of 1.0043 reported by Orellano et al. (2020). Whereas this new effect estimate would lower the estimated excess mortality at the proposed short-term AQG level, it would not change the proposed AQG level because this was calculated according to the methods explained in [section 2.5](#).

Step 8. Reconsider causality

The association between short-term ozone concentrations and all-cause mortality was judged as likely causal in the 2016 outcome prioritization framework (see [section 2.3.3](#)). This judgement was changed in the US EPA ISA of 2020 to suggestive of a causal relationship. A discussion of these changes is provided in [section 2.5](#) of this report. The relationship between short-term ozone and respiratory effects (including mortality) was classified as causal.

As mentioned in step 7, new results from a very large worldwide study (Vicedo-Cabrera et al., 2020) provide further support for an association between short-term ozone and all-cause mortality. The GDG judged it prudent to propose a short-term AQG level for ozone, also in view of the large proportions of the world population exposed to relatively high ozone concentrations and the prospect that concentrations may go up rather than down as a result of climate change.

3.4.3.1 Interim targets

Interim targets are proposed as incremental steps in a progressive reduction of air pollution and are intended for use in areas where pollution is high. For a more detailed rationale for establishing and using interim targets, see [section 2.5.3](#).

The recommendation is a short-term daily maximum 8-hour ozone AQG level of 100 µg/m³, defined as the 99th percentile (equivalent to three to four exceedance days per year) of the annual distribution of daily maximum 8-hour average concentrations.

An interim target 1 of 160 µg/m³ is retained from *Global update 2005*. An interim target 2 of 120 µg/m³ is also proposed, as shown in [Table 3.15](#).

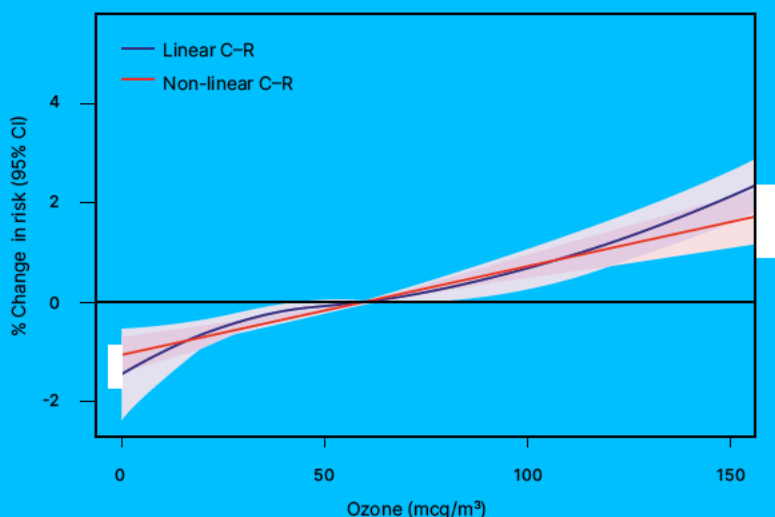
Table 3.15. Recommended short-term (8-hour) daily maximum AQG level and interim targets for ozone^a

Recommendation	O ₃ (µg/m³)
Interim target 1	160
Interim target 2	120
AQG level	100

^a Defined as the 99th percentile of the annual distribution of daily maximum 8-hour average concentrations (equivalent to 3–4 exceedance days per year).

If mortality in a population exposed, on a given day, to ozone at the AQG level is arbitrarily set at 100, then it will be 103 and 101, respectively, in populations exposed, on a given, high pollution day to ozone at the interim target 1 and 2 levels. These projections are based on the linear HR of 1.0043 per 10-µg/m³ increase in ozone for all non-accidental mortality reported in the systematic review. At higher concentrations, the CRF may no longer be linear, which would change the numbers in this example.

Fig. 3.11. Exposure–response curve for 8-hour ozone exposure ($\mu\text{g}/\text{m}^3$) and all-cause mortality^a



C-R: concentration–response.

^a The change in risk is expressed relative to a mean ozone concentration of about $60 \mu\text{g}/\text{m}^3$.

Source: Vicedo-Cabrera et al. (2020).

3.5 Nitrogen dioxide

3.5.1 General description

The general description comes from *Global update 2005*.

Many chemical species of nitrogen oxides exist, but the air pollutant species of most interest from the point of view of human health is nitrogen dioxide. Nitrogen dioxide is a reddish brown gas with a characteristic pungent odour. Nitric oxide spontaneously produces the dioxide when exposed to air. Nitrogen dioxide gas is a strong oxidant, and reacts with water to produce nitric acid and nitric oxide.

Nitrogen dioxide is an important atmospheric trace gas not only because of its health effects but also because: (a) it absorbs visible solar radiation and contributes to impaired atmospheric visibility; (b) it absorbs visible radiation and has a potentially direct role in global climate change; (c) it is, along with nitric oxide, a chief regulator of the oxidizing capacity of the free troposphere by controlling the build-up and fate of radical species, including hydroxyl radicals; and (d) it plays a critical role in determining ozone concentrations in the troposphere because the photolysis of nitrogen dioxide is the only key initiator of the photochemical formation of ozone, whether in polluted or in non-polluted atmospheres (US EPA, 1993, 1995).

Nitrogen dioxide is subject to extensive further atmospheric transformations that lead to the formation of strong oxidants that participate in the conversion of nitrogen dioxide to nitric acid and sulfur dioxide to sulfuric acid and subsequent conversions to their ammonium neutralization salts. Thus, through the photochemical reaction sequence initiated by solar-radiation-induced activation of nitrogen dioxide, the newly generated pollutants are an important source of organic, nitrate and sulfate particles currently measured as PM₁₀ or PM_{2.5}. For these reasons, nitrogen dioxide is a key precursor of a range of secondary pollutants whose effects on human health are well-documented (WHO Regional Office for Europe, 2006).

Conversion factors: at 20 °C and 1013 hPa, 1 ppm = 1.914 mg/m³ and 1 mg/m³ = 0.523 ppm.

3.5.2 Recommended AQG level for long-term exposure to nitrogen dioxide

Based on the methods for deriving an AQG level outlined in the guideline development protocol, this section provides a recommendation for an AQG level for long-term nitrogen dioxide that is based on all non-accidental mortality and cause-specific, respiratory mortality (Table 3.16).

The epidemiological evidence underpinning the AQG level is discussed in a systematic review commissioned by WHO, as explained in more detail in section 2.4. The review (Huangfu & Atkinson, 2020) was published in *Environment International* (Whaley et al., 2021) as open access.

As discussed in section 2.3, there has been no separate, independent assessment of the mechanistic, toxicological and human clinical studies relating nitrogen dioxide to human health.

This section follows the eight steps outlined in the protocol for AQG level development. Tables and figures mentioned during the eight steps are listed at the end of the discussion of each recommendation.

Step 1. Assess RR estimates and, when available, CRFs

The systematic review by Huangfu & Atkinson (2020) on nitrogen dioxide and all non-accidental mortality reported a meta-analytic effect estimate of RR = 1.02 (95% CI: 1.01–1.04) per 10 µg/m³ nitrogen dioxide, assuming a linear relationship. The certainty of the evidence was considered moderate according to GRADE. The authors found an indication of a supralinear relationship, suggesting a steeper risk increase at lower exposure levels. CRFs were provided by a few studies.

They are shown in [Fig. 3.12](#) and [Fig. 3.13](#) for those studies with information on low to very low levels of exposure measured (step 2).

Step 2. Determine the lowest level of exposure measured

For 19 of the 24 studies included in the meta-analysis, the 5th percentile of the exposure distribution was reported or could be calculated from the reported mean and standard deviation ([Table 3.17](#)). As the concentration distributions are often lognormal, this calculation is not straightforward. Therefore, in most cases it was replaced by actual reports of the relevant numbers obtained from the study authors. The three lowest levels reported or estimated in these studies are $-2.7 \mu\text{g}/\text{m}^3$ (Yorifuji et al., 2013) and $4.0 \mu\text{g}/\text{m}^3$ (Bentayeb et al., 2015) (both estimated) and $6.3 \mu\text{g}/\text{m}^3$ (Weichenthal, Pinault & Burnett, 2017). The GDG ignored these three numbers because the first two were a function of very high standard deviations in studies with otherwise not very low mean concentrations. The GDG ignored the third study because it was considered to be at a high RoB (see below). The next five lowest 5th percentile concentrations were $7.3 \mu\text{g}/\text{m}^3$ (Tonne & Wilkinson, 2013), $8.3 \mu\text{g}/\text{m}^3$ in two separate studies (Hart et al., 2011, 2013), $9.6 \mu\text{g}/\text{m}^3$ (Turner et al., 2016) and $10.3 \mu\text{g}/\text{m}^3$ (Carey et al., 2013). The average of these five 5th percentile values was $8.8 \mu\text{g}/\text{m}^3$; all of these studies found positive associations between nitrogen dioxide and all non-accidental mortality, of which three were statistically significant by themselves. The sum of weights in the meta-analysis was $> 25\%$, indicating that these studies made an important contribution to the meta-analysis.

Step 3. Determine the minimal relevant increase in health outcomes

The GDG decided to consider as relevant any increase in risk for an adverse health outcome related to long-term exposure to a pollutant.

Step 4. Determine the starting point for AQG level determination as the long-term concentration of the pollutant from which the minimal relevant amount of the health outcome will result

Thus, the average of the five lowest 5th percentile levels measured in these five studies was the starting point for deriving an AQG level: $8.8 \mu\text{g}/\text{m}^3$ nitrogen dioxide. The data obtained support a long-term AQG level of no more than $10 \mu\text{g}/\text{m}^3$, based on the association between long-term nitrogen dioxide and all non-accidental mortality.

Step 5. Compare the AQG level across critical health outcomes: cause-specific mortality

The cause-specific mortality outcomes that were investigated all yielded bigger RRs than the RR for all non-accidental mortality, with RRs of 1.03 (95% CI: 1.01–1.04),

1.03 (95% CI: 1.01–1.05) and 1.06 (95% CI: 1.02–1.10) per 10 $\mu\text{g}/\text{m}^3$ for COPD, respiratory and acute lower respiratory infection mortality, respectively. The certainty of the evidence was rated as high for COPD mortality and moderate for non-malignant respiratory mortality and acute lower respiratory infection mortality. [Table 3.18](#) shows the findings for non-malignant respiratory mortality. Starting points for AQG level determination for this additional health outcome would not change the analysis much, as the studies are essentially a large proportion of those in [Table 3.17](#). Therefore, the data obtained for cause-specific mortality also support a long-term AQG level of no more than 10 $\mu\text{g}/\text{m}^3$.

Step 6. Assess certainty of the evidence

One of the studies that made up the lowest levels measured in the non-accidental mortality studies (Weichenthal, Pinault & Burnett, 2017) was considered at high RoB, so the GDG did not include that study in further calculations.

Step 7. Consider new evidence

Several new studies were published between autumn 2018 and the summer of 2020. The systematic review did not include these, so the GDG had to make its own overview of these studies. These new studies were largely the same as those identified and included in the revision of the systematic review of long-term PM effects on mortality (Chen & Hoek, 2020). As they were included in the PM review, they are now also discussed in the context of nitrogen dioxide. [Table 3.19](#) shows these studies, ordered by the mean or median exposure level for all non-accidental mortality. These include two studies from Australia (Dirgawati et al., 2019; Hanigan et al., 2019) and two from Canada (Brauer et al., 2019; Pappin et al., 2019), all of which had mean or median nitrogen dioxide levels well below 20 $\mu\text{g}/\text{m}^3$. There are two new studies from the United States (Lefler et al., 2019; Eum et al., 2019), one from Denmark (Hvidtfeldt et al., 2019) and one from the Netherlands (Klompmaaker et al., 2020). Two of these were administrative database studies with no adjustment (Brauer et al., 2019) or with area-level adjustment (Eum et al., 2019) for lifestyle factors such as smoking. The last three studies also reported effect estimates for respiratory mortality ([Table 3.20](#)).

There was no reason, based on these new findings, to change the calculation of the proposed AQG level or the assessment of the certainty of the evidence.

Step 8. Reconsider causality

Most nitrogen dioxide–outcome associations were deemed to be suggestive of being causal or likely causal in the 2016 outcome prioritization framework (see [Table 2.1](#) in [section 2.3.3](#)). COMEAP published a report in 2018, Associations of long-term average concentrations of nitrogen dioxide with mortality, which

is somewhat more supportive of a causal role for long-term nitrogen dioxide in increasing all non-accidental and, especially, respiratory mortality (PHE, 2018). A 2018 review by the German Environment Agency (in German, with a summary in English) also supports a role for long-term nitrogen dioxide in causing cardiovascular mortality (Schneider et al., 2018). None of the more recent reviews were able to include the rather large number of new studies listed in [Table 3.19](#) and [Table 3.20](#), which provided further support for associations between long-term nitrogen dioxide concentrations and all-cause and respiratory mortality.

The GDG noted that one review specifically investigated how sensitive the associations between long-term nitrogen dioxide concentrations and mortality were to adjustment for different PM metrics (Faustini, Rapp & Forastiere, 2014). Associations with nitrogen dioxide were found to be generally robust.

The 5th percentile (where available) and mean or median of exposure distributions in studies included in the nitrogen dioxide and mortality meta-analysis are indicated in [Table 3.17](#) and [Table 3.18](#) based on data from the Huangfu & Atkinson (2020) systematic review and in [Table 3.19](#) and [Table 3.20](#) for the newly identified studies.

3.5.2.1 Interim targets

Interim targets are proposed as incremental steps in a progressive reduction of air pollution and are intended for use in areas where pollution is high. For a more detailed rationale for establishing and using interim targets, see [section 2.5.3](#).

Interim targets were not specified for nitrogen dioxide in *Global update 2005*. As evident from [Table 3.17](#), [Table 3.18](#), [Table 3.19](#) and [Table 3.20](#), the mean or median concentrations of nitrogen dioxide were well below 40 µg/m³ in most studies.

The GDG recommends using the long-term air quality guideline from *Global update 2005* of 40 µg/m³ as interim target 1, as this is a level already shown to be achievable in many parts of the world.

As interim target 2, a level of 30 µg/m³ is proposed and, as interim target 3, a level of 20 µg/m³ is proposed. Proposing two additional interim targets provides reasonable guidance to policy-makers on how to bridge the gap between the 2005 air quality guideline and the new, much lower, AQG level.

The recommendation is an annual nitrogen dioxide AQG level of 10 µg/m³.

An interim target 1 of 40 µg/m³, an interim target 2 of 30 µg/m³ and an interim target 3 of 20 µg/m³ are proposed, as shown in [Table 3.16](#).

Table 3.16. Recommended AQG level and interim targets for nitrogen dioxide

Recommendation	NO ₂ (µg/m ³)
Interim target 1	40
Interim target 2	30
Interim target 3	20
AQG level	10

If all-cause mortality in a population exposed to nitrogen dioxide at the AQG level is arbitrarily set at 100, then it will be 106, 104 and 102, respectively, in populations exposed to nitrogen dioxide at the interim target 1, 2 and 3 levels. For respiratory mortality, the numbers would be 109, 106 and 103, respectively, at the interim target 1, 2 and 3 levels. These projections are based on the linear HRs of 1.02 and 1.03 per 10-µg/m³ increase in nitrogen dioxide for all non-accidental and respiratory mortality, respectively, as reported in the systematic review. At higher concentrations, the CRF may no longer be linear, which would change the numbers in this example.

Table 3.17. Studies on long-term nitrogen dioxide exposure and all non-accidental mortality included in the systematic review by Huangfu & Atkinson (2020), ordered by me(di)an concentration

Study	Me(di)an (µg/m ³)	SD	P5	P25	HR (95% CI) ^a
Tonne & Wilkinson (2013)	18.5	6.8	7.3 ^b	–	1.01 (0.98–1.04)
Weichenthal, Pinault & Burnett (2017) ^c	21.6	–	6.3 ^d	12.1	1.04 (1.03–1.04)
Crouse et al. (2015)	21.8	–	–	11.3	1.03 (1.03–1.04)
Turner et al. (2016)	21.8	9.6	9.6 ^d	–	1.02 (1.01–1.03)
Yorifuji et al. (2013)	22.0	15.0	–2.7 ^b	–	1.12 (1.07–1.18)
Carey et al. (2013)	22.5	7.4	10.3 ^b	–	1.02 (1.00–1.05)
Beelen et al. (2014)	22.2	–	15.3 ^d	19.9	1.01 (0.99–1.03)

Table 3.17 contd

Study	Me(di)an ($\mu\text{g}/\text{m}^3$)	SD	P5	P25	HR (95% CI) ^a
Hart et al. (2013)	26.1	–	8.3 ^d	19.0	1.01 (1.00–1.03)
Hart et al. (2011)	26.7	13.3	8.3 ^d	–	1.05 (1.02–1.08)
Bentayeb et al. (2015)	28.0	14.6	4.0 ^b	–	1.07 (1.00–1.15)
Krewski et al. (2003)	30.3	–	–	–	1.08 (1.02–1.14)
Fischer et al. (2015)	31.0	–	19.0 ^d	26.0	1.03 (1.02–1.04)
Hartiala et al. (2016)	35.9	3.4	30.3 ^b	–	1.00 (0.75–1.34)
Filleul et al. (2005)	36.5	–	–	–	1.14 (1.03–1.26)
Lipfert et al. (2006)	37.2	–	16.5 ^d	–	1.03 (0.99–1.07)
Brunekreef et al. (2009) ^b	38.0	–	22.0 ^d	–	1.03 (1.00–1.05)
Jerrett et al. (2009)	39.1	–	32.0 ^d	–	1.23 (1.00–1.51)
Chen et al. (2016)	40.7	1.6	38.1 ^b	27.1	0.92 (0.90–0.95)
Cesaroni et al. (2013) ^b	43.6	8.4	29.8 ^b	38.5	1.03 (1.02–1.04)
Desikan et al. (2016) ^b	44.6	4.3	37.5 ^b	41.8	0.94 (0.76–1.17)
Rosenlund et al. (2008) ^b	48.5	–	–	–	0.95 (0.89–1.02)
Lipsett et al. (2011)	63.1	18.0	33.5 ^b	–	0.98 (0.95–1.02)
Abbey et al. (1999)	69.2	24.4	29.1 ^a	–	1.00 (0.99–1.01)
Yang et al. (2018)	104.0	–	–	91.0	1.00 (0.99–1.01)

–, data unavailable; P5: 5th percentile (of the distribution of concentrations assigned to study participants); P25: 25th percentile; SD: standard deviation.

^a Per 10 $\mu\text{g}/\text{m}^3$.

^b Calculated from the mean and SD using the following formula: Me(di)an – 1.645 * SD.

^c Considered to be at high RoB.

^d Reported in paper or by authors on request.

Table 3.18. Studies on long-term nitrogen dioxide exposure and respiratory mortality included in the systematic review by Huangfu & Atkinson (2020), ordered by me(di)an concentration

Study	Me(di)an ($\mu\text{g}/\text{m}^3$)	SD	P5	P25	HR (95% CI) ^a
Weichenthal, Pinault & Burnett (2017) ^b	21.6	–	6.3 ^c	12.1	1.06 (1.04–1.08)
Crouse et al. (2015)	21.8	–	–	11.3	1.02 (1.01–1.04)
Turner et al. (2016)	21.8	9.6	9.6 ^d	–	1.02 (1.00–1.04)
Yorifuji et al. (2013)	22.0	15.0	–2.7 ^d	–	1.19 (1.06–1.34)
Dimakopoulou et al. (2014)	22.2	–	15.3 ^c	19.9	0.97 (0.89–1.04)
Carey et al. (2013)	22.5	7.4	10.3 ^d	–	1.08 (1.04–1.13)
Hart et al. (2011)	26.7	13.3	8.3 ^c	–	1.04 (0.95–1.14)
Fischer et al. (2015)	31.0	–	19.0 ^c	26.0	1.02 (1.01–1.03)
Katanoda et al. (2011)	32.0	–	–	–	1.07 (1.03–1.12)
Brunekreef et al. (2009) ^a	38.0	–	22.0 ^c	–	1.11 (1.00–1.23)
Jerrett et al. (2009)	39.1	–	32.0 ^c	–	1.08 (0.64–1.84)
Cesaroni et al. (2013) ^a	43.6	8.4	29.8 ^d	38.5	1.03 (1.00–1.06)
Lipsett et al. (2011)	63.1	18.0	33.5 ^d	–	0.96 (0.86–1.08)
Abbey et al. (1999)	69.2	24.4	29.1 ^d	–	0.99 (0.98–1.01)
Yang et al. (2018)	104.0	–	–	91.0	1.00 (0.97–1.02)

–, data unavailable; P5: 5th percentile (of the distribution of concentrations assigned to study participants); P25: 25th percentile; SD: standard deviation.

^a Per 10 $\mu\text{g}/\text{m}^3$.

^b Considered to be at high RoB.

^c Reported in paper or by authors on request.

^d Calculated from mean and standard deviation using the following formula: Me(di)an – 1.645 × SD.

Table 3.19. New studies on long-term nitrogen dioxide exposure and all non-accidental mortality published since autumn 2018, ordered by me(di)an concentration

Study	Me(di)an (µg/m³)	SD	P5	P25	HR (95% CI) ^a
Dirgawati et al. (2019)	13.4	4.1	6.7 ^b	–	1.060 (1.000–1.120)
Brauer et al. (2019) – CCHS subjects	16.2	11.1	7.2 ^c	–	1.024 (1.016–1.040)
Brauer et al. (2019); Pappin et al. (2019) – CanCHEC subjects	16.2	–	5.9 ^c	–	1.004 (1.002–1.007)
Hanigan et al. (2019)	17.8	4.8	9.9 ^b	14.3	1.060 (0.960–1.140)
Lefler et al. (2019)	20.1	10.7	2.5 ^b	–	1.010 (1.002–1.017)
Klompaker et al. (2020)	23.1	–	–	19.3	0.990 (0.960–1.010)
Hvidtfeldt et al. (2019)	25.0	–	17.9 ^c	–	1.070 (1.040–1.100)
Eum et al. (2019)	26.7	–	–	18.2	1.027 (1.027–1.029)

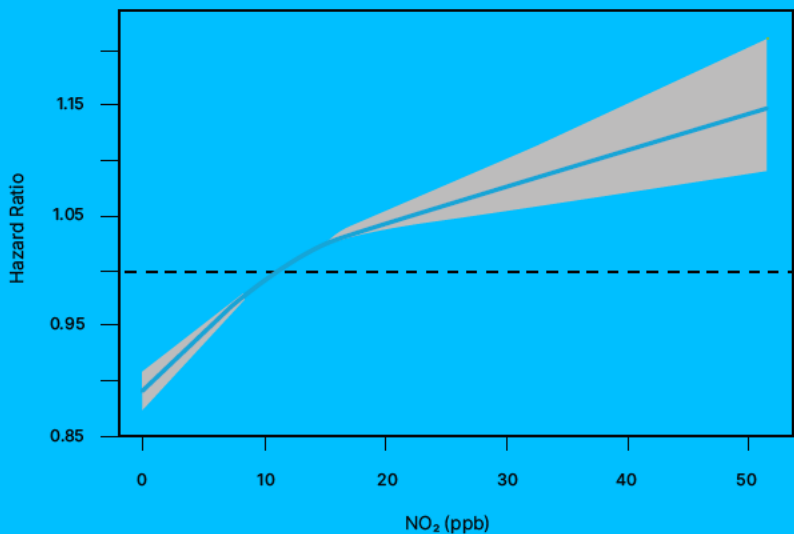
–, data unavailable; CCHS: Canadian Community Health Survey; P5: 5th percentile (of the distribution of concentrations assigned to study participants); P25: 25th percentile; SD: standard deviation.
^a Per 10 µg/m³.
^b Calculated from the mean and SD using the following formula: Me(di)an – 1.645 * SD.
^c Reported in paper or by authors on request.

Table 3.20. New studies on long-term nitrogen dioxide exposure and respiratory mortality published since autumn 2018, ordered by me(di)an concentration

Study	Me(di)an (µg/m³)	SD	P5	P25	HR (95% CI) ^a
Klompaker et al. (2020)	23.1	–	–	19.3	0.980 (0.890–1.070)
Hvidtfeldt et al. (2019)	25.0	–	17.9 ^b	–	1.030 (0.970–1.100)
Eum et al. (2019)	26.7	–	–	18.2	1.027 (1.023–1.030)

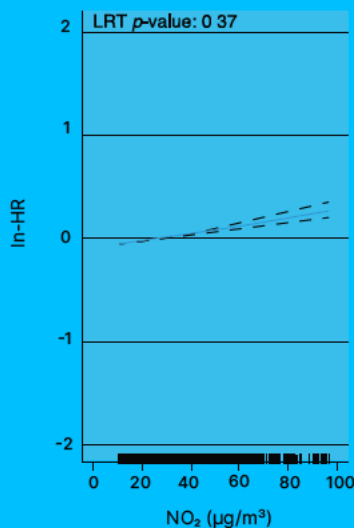
–, data unavailable; P5: 5th percentile (of the distribution of concentrations assigned to study participants); P25: 25th percentile; SD: standard deviation.
^a Per 10 µg/m³.
^b Reported in paper or by authors on request.

Fig. 3.12. CRFs for long-term nitrogen dioxide exposure (ppb) and all non-accidental mortality in Canada^a



^a HRs are relative to the mean concentration of 11.6 ppb (= 22.9 µg/m³).
Source: reproduced from Crouse et al. (2015) with permission of the lead author.

Fig. 3.13. CRFs for long-term nitrogen dioxide exposure (µg/m³) and all non-accidental mortality in the Netherlands^a



ln: natural logarithm; LRT: likelihood ratio test.
^a ln-HR = log HR, relative to the mean nitrogen dioxide concentration. The likelihood-ratio test P value indicates that there was no significant deviation from linearity.
Source: reproduced from Fischer et al. (2015) with permission of the lead author.

3.5.3 Recommended AQG level for short-term exposure to nitrogen dioxide

Based on the methods for deriving an AQG level outlined in the guideline development protocol, this section provides an AQG level for short-term, daily average nitrogen dioxide that is based on all-cause non-accidental mortality and asthma hospital admissions and emergency room visits ([Table 3.21](#)).

The epidemiological evidence underpinning the AQG level is discussed in two systematic reviews commissioned by WHO, as explained in more detail in [section 2.4](#). The reviews, conducted by Orellano et al. (2020) and Zheng et al. (2021), were published in *Environment International* (Whaley et al., 2021) as open access.

As discussed in [section 2.3](#), there has been no separate, independent assessment of the mechanistic, toxicological and human clinical studies relating nitrogen dioxide to human health. However, comprehensive evaluations by authoritative bodies such as COMEAP, Health Canada and US EPA were taken into account in the development of the AQG levels. This was especially relevant when assessing causality of the associations examined in the systematic reviews (see step 8).

This section follows the eight steps outlined in the protocol for AQG level development. Tables and figures mentioned during the eight steps are listed at the end of the discussion of each recommendation.

Step 1. Assess RR estimates and, when available, CRFs

The systematic review by Orellano et al. (2020) on 24-hour average nitrogen dioxide and all-cause non-accidental mortality reported a meta-analytic effect estimate of $RR = 1.0072$ (95% CI: 1.0059–1.0085) per $10 \mu\text{g}/\text{m}^3$ nitrogen dioxide, assuming a linear relationship. The certainty of the evidence was considered high according to GRADE. CRFs were provided by several studies. An example from a study in Austria shows an association between nitrogen dioxide and all-cause mortality at very low levels of exposure ([Fig. 3.14](#)) (Moshhammer et al., 2020).

Step 2. Determine the lowest level of exposure measured

As discussed in the protocol for deriving AQG levels, the lowest concentrations in time-series studies of effects of daily variations in air pollution concentrations are often very low. Therefore, the 5th percentiles of these daily distributions cannot be used as starting points for AQG level development. In such cases, the protocol suggests identifying the 99th percentile of common distributions of daily air pollution concentrations corresponding to an average long-term concentration equivalent to the proposed annual AQG level. This is $10 \mu\text{g}/\text{m}^3$ for nitrogen dioxide.

Common distributions observed in large numbers of cities around the world (data from Liu et al. (2019)) suggest a ratio of about 2.5 for 99th percentiles of daily concentrations to the annual mean nitrogen dioxide. Therefore, a short-term AQG level of 25 $\mu\text{g}/\text{m}^3$ is suggested.

Step 3. Determine the minimal relevant increase in health outcomes

The GDG decided to consider as relevant any increase in risk for an adverse health outcome related to long-term exposure to a pollutant. For short-term exposures, the CRFs from the systematic review by Orellano et al. (2020) were used to calculate the increase in mortality expected on a day with a 24-hour nitrogen dioxide concentration of 25 $\mu\text{g}/\text{m}^3$ compared with a day with a 24-hour nitrogen dioxide concentration of 10 $\mu\text{g}/\text{m}^3$. With an RR for all-cause mortality of 1.0072 per 10 $\mu\text{g}/\text{m}^3$, the estimated excess mortality on such a day would be 1.1%. However, under compliance with the long-term AQG level, days with concentrations close to 25 $\mu\text{g}/\text{m}^3$ will correspond to the far upper tail of the distribution of daily exposures. Most days will have much lower values, with close to half having concentrations below or far below the annual AQG level. The health burden related to a few days with higher concentrations corresponds to a very small fraction of the total air pollution-related burden.

Step 4. Determine the starting point for AQG level determination as the 99th percentile, as mentioned in step 3

The data obtained support a short-term AQG level of no more than 25 $\mu\text{g}/\text{m}^3$, based on the association between short-term nitrogen dioxide and all-cause non-accidental mortality.

Step 5. Compare the AQG level across critical health outcomes: cause-specific mortality and asthma hospital admissions and emergency room visits

Studies on short-term associations and cause-specific mortality were not reviewed. However, another systematic review commissioned by WHO assessed the evidence for associations between nitrogen dioxide and daily hospital admissions for asthma (Zheng et al., 2021). This review found an effect estimate of RR = 1.014 (95% CI: 1.009–1.019) per 10 $\mu\text{g}/\text{m}^3$, which would produce an excess morbidity 2.1% on a day at the proposed short-term AQG level of 25 $\mu\text{g}/\text{m}^3$ compared with a day at the proposed long-term AQG level of 10 $\mu\text{g}/\text{m}^3$. As is the case when considering mortality in step 3, under compliance with the long-term AQG level, days with concentrations close to 25 $\mu\text{g}/\text{m}^3$ will correspond to the far upper tail of the distribution of daily exposures. Most days will have much lower values, with close to half having concentrations below or far below the annual AQG level. The health burden related to a few days with higher concentrations corresponds to a very small fraction of the total air pollution-related burden.

Step 6. Assess certainty of the evidence

As mentioned in step 1, the certainty level is high for the evidence linking short-term nitrogen dioxide concentration variations to short-term mortality variations. In addition, as shown in [Fig. 3.14](#), there is evidence that this association persists to very low levels of exposure.

Step 7. Consider new evidence

Several new studies have been published since autumn 2018. The GDG did not make an inventory of all new time-series studies. The MCC Collaborative Research Network has reported new findings from a very large database on short-term mortality effects of PM_{2.5} and ozone (Liu et al., 2019; Vicedo-Cabrera et al., 2020); an analysis from the same database on short-term effects of nitrogen dioxide was also published (Meng et al., 2021). The effect estimates from this new analysis are in agreement with those from the WHO-commissioned systematic review.

Step 8. Reconsider causality

The association between short-term nitrogen dioxide concentrations and all-cause mortality was judged to be suggestive of a causal relationship in the 2016 outcome prioritization framework (see [section 2.3.3](#)), following authoritative evaluations by Health Canada, US EPA and other bodies. However, the association between short-term nitrogen dioxide concentrations and respiratory effects was judged to be causal. This judgement provides strong support for a short-term AQG level for nitrogen dioxide in view of the reported association with asthma hospital admissions and emergency room visits.

The GDG noted that one review specifically investigated how sensitive the associations between short-term nitrogen dioxide and mortality were to adjustment for different PM metrics (Mills et al., 2016). Associations with nitrogen dioxide were found to be generally robust.

3.5.3.1 Interim targets

Interim targets are proposed as incremental steps in a progressive reduction of air pollution and are intended for use in areas where pollution is high. For a more detailed rationale for establishing and using interim targets, see [section 2.5.3](#).

An interim target 1 of 120 µg/m³ is proposed – which is roughly comparable to the existing 1-hour 2005 air quality guideline of 200 µg/m³. An interim target 2 of 50 µg/m³ is also proposed. Both interim targets use the same definition of 99th percentiles of the distribution of 24-hour concentrations over a one-year period.

The recommendation is a short-term (24-hour) nitrogen dioxide AQG level of 25 µg/m³, defined as the 99th percentile (equivalent to three to four exceedance days per year) of the annual distribution of 24-hour average concentrations.

An interim target 1 of 120 µg/m³ and an interim target 2 of 50 µg/m³ are proposed, as shown in Table 3.21.

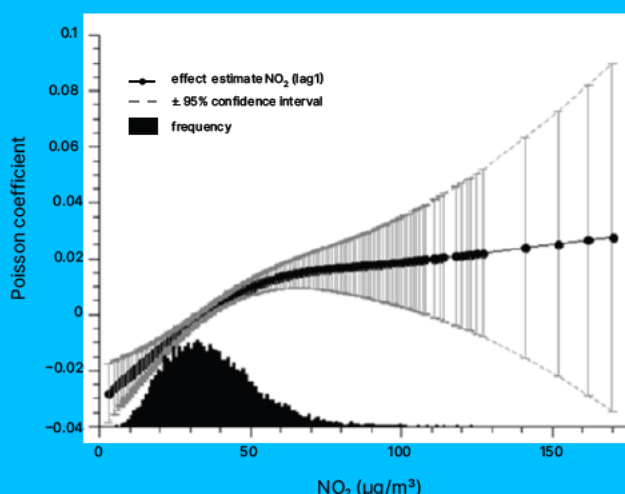
Table 3.21. Recommended short-term (24-hour) AQG level and interim targets for nitrogen dioxide^a

Recommendation	NO ₂ (µg/m³)
Interim target 1	120
Interim target 2	50
AQG level	25

^a Defined as the 99th percentile of the annual distribution of 24-hour average concentrations (equivalent to 3–4 exceedance days per year).

If mortality in a population exposed to nitrogen dioxide for a day at the AQG level of 25 µg/m³ is arbitrarily set at 100, then it will be 107 and 102, respectively, in populations exposed to nitrogen dioxide at the interim target 1 and 2 levels. These projections are based on the linear HR of 1.0072 HR per 10-µg/m³ increase in nitrogen dioxide of all non-accidental mortality reported in the systematic review. At higher concentrations, the CRF may no longer be linear, which would change the numbers in this example.

Fig. 3.14. Association between 24-hour average nitrogen dioxide concentrations ($\mu\text{g}/\text{m}^3$) and mortality in Vienna, Austria^a



^a The corresponding linear effect estimate is a 0.21% increase in total mortality per previous-day NO_2 increase of $10 \mu\text{g}/\text{m}^3$.

Source: Moshhammer et al. (2020).

3.6 Sulfur dioxide

3.6.1 General description

The general description comes from *Global update 2005*.

Historically, sulfur dioxide and PM derived from the combustion of fossil fuels have been the main components of air pollution in many parts of the world. The most serious problems have been experienced in large urban areas where coal has been used for domestic heating or for poorly controlled combustion in industrial installations. In such situations, the complex of pollutants has generally been considered collectively, drawing on findings from epidemiological studies carried out decades ago in areas formerly heavily polluted. Guidelines developed in this way had been related to averaging times of 24 hours in respect of acute effects and one year in respect of chronic effects.

Separate attention has been paid to sulfur dioxide alone, based largely on findings from controlled human exposure studies. These allow guidelines to be developed in terms of shorter averaging periods of the order of one hour. These are relevant to exposures to peak concentrations that may arise from sources burning coal or heavy oil, whether or not accompanied by substantial concentrations of PM.

Epidemiological studies published in the last decade [i.e. 1995–2004] provide suggestive evidence on the health effects of sulfur dioxide. Thus, a section has been introduced in this revision focusing on epidemiological results in locations where the sources of sulfur dioxide are mainly motor vehicles and various industries.

Sulfur dioxide is derived from the combustion of sulfur-containing fossil fuels and is a major air pollutant in many parts of the world. Oxidation of sulfur dioxide, especially at the surface of particles in the presence of metallic catalysts, leads to the formation of sulfurous and sulfuric acids. Neutralization, by ammonia, leads to the production of bisulfates and sulfates.

Sulfur dioxide is a colourless gas that is readily soluble in water. Sulfuric acid is a strong acid formed from the reaction of sulfur trioxide (SO₃) with water. Sulfuric acid is strongly hygroscopic. As a pure material it is a colourless liquid with a boiling point of 330 °C. Ammonium bisulfate (NH₄HSO₄), which is also a strong acid but is less acidic than sulfuric acid as a pure material, is a crystalline solid with a melting point of 147 °C. The formation of very small droplets of sulfuric acid occurs by nucleation. Many vapours are able to condense on the surface of existing very fine nuclei and lead to the growth of composite particles. (WHO Regional Office for Europe, 2006).

Conversion factors: at 20 °C and 1013 hPa, 1 ppm = 2660 µg/m³ and 1 mg/m³ = 0.3759 ppm.

3.6.2. Recommended AQG level for 24-hour exposure to sulfur dioxide

Based on the methods for deriving an AQG level outlined in the guideline development protocol, the GDG recommends an AQG level for short-term, 24-hour mean sulfur dioxide concentration based on its relationship with asthma hospital admissions and emergency room visits, daily non-accidental mortality and respiratory mortality (Table 3.22). As discussed in Chapter 2, the association between sulfur dioxide and mortality was added to the list of pollutant–outcome pairs at a later stage to improve continuity with *Global update 2005*.

The epidemiological evidence underpinning the AQG level is discussed in a systematic review commissioned by WHO on asthma hospital admissions and emergency room visits (Zheng et al., 2021) and another on daily sulfur dioxide mortality (Orellano, Reynoso & Quaranta, 2021). These reviews were published in *Environment International* (Whaley et al., 2021) as open access.

As discussed in [section 2.3](#), there has been no separate, independent assessment of the mechanistic, toxicological and human clinical studies relating sulfur dioxide to human health.

This section follows the eight steps outlined in the protocol for AQG level development. Tables and figures mentioned during the eight steps are listed at the end of the discussion of each recommendation.

Step 1. Assess RR estimates and, when available, CRFs

The systematic review by Zheng et al. (2021) on sulfur dioxide and asthma hospital admissions and emergency room visits reported a meta-analytic effect estimate of $RR = 1.010$ (95% CI: 1.001–1.020) per $10 \mu\text{g}/\text{m}^3$ sulfur dioxide, assuming a linear relationship. The certainty of the evidence was considered low according to GRADE. More elaborate analyses of the CRF shape were not provided by any of the studies on asthma included in the systematic review. The systematic review by Orellano, Reynoso & Quaranta (2021) on sulfur dioxide and daily mortality reported a meta-analytic effect estimate of $RR = 1.0059$ (95% CI: 1.0046–1.0071) per $10 \mu\text{g}/\text{m}^3$ sulfur dioxide, assuming a linear relationship. For respiratory mortality, the meta-analytic effect estimate was $RR = 1.0067$ (95% CI: 1.0025–1.0109) per $10 \mu\text{g}/\text{m}^3$ sulfur dioxide, assuming a linear relationship. The certainty of the evidence was considered high according to GRADE for all non-accidental mortality and moderate for respiratory mortality.

Step 2. Determine the lowest level of exposure measured

As discussed in the protocol for deriving AQG levels, the lowest concentrations in time-series studies of effects of daily variations in air pollution concentrations are often very low. The minimum concentration reported by most of the studies included in the systematic reviews by Zheng et al. (2021) and Orellano, Reynoso & Quaranta (2021) was below $1 \mu\text{g}/\text{m}^3$. The protocol suggests identifying as the daily AQG level the 99th percentile of a distribution of daily air pollution concentrations corresponding to an average long-term concentration equivalent to the annual AQG level. However, in the case of sulfur dioxide, there is no annual AQG level that can be used as a point of departure, so this approach cannot be applied.

Step 3. Determine the minimal relevant increase in health outcomes

The GDG decided to consider as relevant any increase in risk for an adverse health outcome related to long-term exposure to a pollutant. For short-term exposures, the assumption of a linear CRF and a risk coefficient from the systematic reviews by Zheng et al. (2021) and Orellano, Reynoso & Quaranta (2021) were used to calculate the increase in asthma hospital admissions and emergency room

visits and daily non-accidental mortality and respiratory mortality relative to a daily mean sulfur dioxide concentration of $0 \mu\text{g}/\text{m}^3$. With an RR of 1.010 per $10 \mu\text{g}/\text{m}^3$, any $10\text{-}\mu\text{g}/\text{m}^3$ increase would produce a 1% increase in asthma hospital admissions and emergency room visits. The increases in non-accidental mortality and respiratory mortality would be 0.6% and 0.7%, respectively, per $10 \mu\text{g}/\text{m}^3$.

Step 4. Determine the starting point for AQG level determination as the 99th percentile, as mentioned in step 3

In the proposed short-term AQG levels for $\text{PM}_{2.5}$, PM_{10} , ozone and nitrogen dioxide, a comparison was made between the expected excess deaths or asthma hospital admissions and emergency room visits at the 99th percentiles of daily distributions corresponding to a distribution that is in compliance with the proposed long-term AQG levels for these pollutants. For non-accidental mortality, these excess estimates were up to 1.72% for deaths related to ozone and 4.8% for asthma hospital admissions and emergency room visits related to ozone. Similar percentage increases related to sulfur dioxide, relative to a $0 \mu\text{g}/\text{m}^3$ concentration, would be expected at a daily mean of about $30 \mu\text{g}/\text{m}^3$ (3% increase in asthma hospital admissions and emergency room visits, 1.8% increase in daily non-accidental mortality). The MCC Collaborative Research Network database (A. Gasparrini, London School of Hygiene and Tropical Medicine, unpublished data, 23 June 2020; Liu et al., 2019) documented a ratio of 3.9 between the 99th percentile of daily concentrations and the annual mean sulfur dioxide concentration across hundreds of cities from all over the world. Following the same logic used for pollutants for which there is a proposed long-term AQG level, the starting point for a short-term sulfur dioxide AQG level would be $40 \mu\text{g}/\text{m}^3$. The rationale is that with a ratio of about 4 between the 99th percentile and annual mean, $40 \mu\text{g}/\text{m}^3$ would correspond to an increase of $30 \mu\text{g}/\text{m}^3$ over an annual mean of $10 \mu\text{g}/\text{m}^3$, which is about the same as the overall mean concentration observed across almost 400 locations worldwide in the MCC Collaborative Research Network database (A. Gasparrini, London School of Hygiene and Tropical Medicine, unpublished data, 23 June 2020; Liu et al., 2019). The GDG recognizes that the choice for a background of $10 \mu\text{g}/\text{m}^3$ is, to some extent, arbitrary but notes that the estimated excess mortality at days with concentrations at the recommended AQG level is small and is roughly comparable across all pollutants considered in this report.

Step 5. Compare the AQG level across critical health outcomes

No other health outcomes were evaluated in the systematic reviews.

Step 6. Assess certainty of the evidence

As mentioned in step 1, the evidence base supporting an association between 24-hour average sulfur dioxide and asthma hospital admissions and emergency

room visits was considered to be of low certainty. For all non-accidental mortality, it was considered to be of high certainty.

Step 7. Consider new evidence

No new studies on the relation between sulfur dioxide exposure and asthma hospital admissions and emergency room visits and non-accidental or respiratory mortality were considered.

Step 8. Reconsider causality

The association between short-term sulfur dioxide concentrations and asthma hospital admissions and emergency room visits was judged to be causal for respiratory effects in the 2016 outcome prioritization framework (see [section 2.3.3](#)), based on assessments by Health Canada and the US EPA. The US EPA published a new ISA on sulfur oxides in 2017 (US EPA, 2017) that did not change that assessment, and which classifies the short-term association with mortality as suggestive of a causal relationship.

3.6.2.1 Interim targets

Interim targets are proposed as incremental steps in a progressive reduction of air pollution and are intended for use in areas where pollution is high. For a more detailed rationale for establishing and using interim targets, see [section 2.5.3](#).

Recommended interim targets are the same as in *Global update 2005*. There are still some places in the world where such high sulfur dioxide concentrations occur, and these areas would benefit from maintaining the existing interim targets.

The recommendation is a short-term (24-hour) sulfur dioxide AQG level of 40 $\mu\text{g}/\text{m}^3$, defined as the 99th percentile (equivalent to three to four exceedance days per year) of the annual distribution of 24-hour average concentrations.

An interim target 1 of 125 $\mu\text{g}/\text{m}^3$ and an interim target 2 of 50 $\mu\text{g}/\text{m}^3$ are proposed, as shown in [Table 3.22](#).

If mortality in a population exposed to sulfur dioxide for a day at the AQG level of 40 $\mu\text{g}/\text{m}^3$ is arbitrarily set at 100, then it will be 105 and 101, respectively, in populations exposed to sulfur dioxide at the interim target 1 and 2 levels. These projections are based on the linear HR of 1.0059 per 10- $\mu\text{g}/\text{m}^3$ increase in sulfur dioxide of all non-accidental mortality reported in the systematic review. At higher concentrations, the CRF may no longer be linear, which would change the numbers in this example.

Table 3.22. Recommended short-term (24-hour) AQG level and interim targets for sulfur dioxide^a

Recommendation	SO ₂ (µg/m ³)
Interim target 1	125
Interim target 2	50
AQG level	40

^a Defined as the 99th percentile (equivalent to 3–4 exceedance days per year) of the annual distribution of 24-hour average concentrations.

3.7 Carbon monoxide

3.7.1 General description

The general description comes from the *WHO guidelines for indoor air quality: selected pollutants*.

Carbon monoxide (CO) is a colourless, non-irritant, odourless and tasteless toxic gas. It is produced by the incomplete combustion of carbonaceous fuels such as wood, petrol, coal, natural gas and kerosene. ...

The molecular weight of carbon monoxide is similar to that of air (28.01 vs approximately 29). It mixes freely with air in any proportion and moves with air via bulk transport. It is combustible, may serve as a fuel source and can form explosive mixtures with air. It reacts vigorously with oxygen, acetylene, chlorine, fluorine and nitrous oxide. Carbon monoxide is not detectable by humans either by sight, taste or smell. It is only slightly soluble in water, blood serum and plasma; in the human body, it reacts with haemoglobin to form carboxyhaemoglobin (COHb) (WHO Regional Office for Europe, 2010).

Conversion factors: at 20 °C and 1013 hPa, 1 ppm = 1.165 mg/m³ and 1 mg/m³ = 0.858 ppm.

3.7.2 Recommended AQG level for 24-hour exposure to carbon monoxide

Based on the methods for deriving an AQG level outlined in the guideline development protocol, this section provides an AQG level for short-term, 24-hour mean carbon monoxide concentration based on its association with hospital admissions and mortality from myocardial infarction (Table 3.23).

The epidemiological evidence underpinning the AQG level is discussed in a systematic review commissioned by WHO, as explained in more detail in [section 2.4](#). The review, conducted by Lee et al. (2020), was published in *Environment International* (Whaley et al., 2021) as open access.

As discussed in [section 2.3](#), there has been no separate, independent assessment of the mechanistic, toxicological and human clinical studies relating carbon monoxide to human health.

This section follows the eight steps outlined in the protocol for AQG level development. Tables and figures mentioned during the eight steps are listed at the end of the discussion of each recommendation.

Step 1. Assess RR estimates and, when available, CRFs

The systematic review by Lee et al. (2020) on carbon monoxide and hospital admissions and mortality from myocardial infarction reported a meta-analytic effect estimate of $RR = 1.052$ (95% CI: 1.017–1.089) per 1 mg/m^3 carbon monoxide, assuming a linear relationship. The certainty of the evidence was considered moderate according to GRADE. More elaborate analyses of the CRF shape were not provided by any of the myocardial infarction studies included in the systematic review. However, the effects were seen mostly in studies with higher carbon monoxide levels, with the effect estimate being $RR = 1.019$ (95% CI: 1.011–1.027) in studies with a median carbon monoxide level exceeding 1.15 mg/m^3 compared with $RR = 1.00$ (95% CI: 0.998–1.003) in the rest of the studies.

Step 2. Determine the lowest level of exposure measured

As discussed in the protocol for deriving AQG levels, the lowest concentrations in time-series studies of effects of daily variations in air pollution concentrations are often very low. The minimum concentration reported by most of the studies included in the systematic review by Lee et al. (2020) was below 0.5 mg/m^3 and the mean carbon monoxide level ranged from 0.35 mg/m^3 to 4.56 mg/m^3 ; in half of the studies, the median carbon monoxide level was below 1.15 mg/m^3 . The protocol suggests identifying as the daily AQG level the 99th percentile of a distribution of daily air pollution concentrations corresponding to an average long-term concentration equivalent to the annual AQG level. However, in the case of carbon monoxide, there is no annual AQG level that can be used as a point of departure, so this approach cannot be applied.

Step 3. Determine the minimal relevant increase in health outcomes

The GDG decided to consider as relevant any increase in risk for an adverse health outcome related to long-term exposure to a pollutant. For short-term exposures,

the assumption of a linear CRF and a risk coefficient from the systematic review by Lee et al. (2020) were used to calculate the increase in myocardial infarction hospital and emergency room admissions and mortality relative to a daily mean carbon monoxide concentration of 0 mg/m³. With an RR of 1.052 per 1 mg/m³, any 1 mg/m³-increase would produce a 5.2% increase in events. However, the Lee et al. (2020) review showed that the magnitude of the RR estimate was highly dependent on inclusion of three partly overlapping studies from East Asia conducted in low carbon monoxide, high nitrogen dioxide and high PM atmospheres (Hsieh et al., 2010; Cheng, Tsai & Yang, 2009; Tsai et al., 2012). Excluding these studies produced an RR of 1.016 (95% CI: 1.009–1.023). In addition, the review showed that there were only three effect estimates for myocardial infarction mortality, none of which suggested an effect from carbon monoxide. The additional exclusion of these estimates produced an RR for myocardial infarction admissions of 1.015 (95% CI: 1.007–1.024). As previously mentioned, the effects were mostly seen in studies with higher carbon monoxide levels, with an effect estimate of RR = 1.019 (95% CI: 1.011–1.027) in studies with a median carbon monoxide level exceeding 1.15 mg/m³ compared with RR = 1.00 (95% CI: 0.998–1.003) in the rest of the studies. For guideline development, the GDG considered the RR of 1.019 that was observed in studies with a median carbon monoxide of more than 1.15 mg/m³ to be more relevant because it excludes obvious outliers, is focused on one outcome (myocardial infarction admissions) rather than two (admissions plus mortality) and is restricted to the concentration range over which effects were actually demonstrated. Using this RR, the expected excess myocardial infarctions would be 5.4% on a 4-mg/m³ day compared with a day with a carbon monoxide concentration of 1.15 mg/m³. The excess would be 11.1% at the 2010 WHO indoor 24-hour guideline for carbon monoxide of 7 mg/m³ (WHO Regional Office for Europe, 2010).

Step 4. Determine the starting point for AQG level determination as the 99th percentile, as mentioned in step 3

A 99th percentile of 4 mg/m³ corresponds to an estimated annual mean of 1.33 mg/m³, based on a 3 : 1 ratio between the 99th percentile and annual mean observed in the large MCC Collaborative Research Network database (A. Gasparrini, London School of Hygiene and Tropical Medicine, unpublished data, 23 June 2020; Liu et al., 2019; Chen et al., 2021). Such a mean would roughly correspond to the median of 1.15 mg/m³, above which the studies included in Lee et al. (2020) showed an elevated risk of exposure. In the development of the short-term AQG levels for PM_{2.5}, PM₁₀, ozone and nitrogen dioxide, a calculation was always made of the differences in events between the mean and the 99th percentile. In the case of carbon monoxide, that difference would be 5.1%. The GDG recommends a short-term AQG level, defined as 99th percentile of daily

mean concentrations in a year, of no more than 4 mg/m³, based on the association between short-term carbon monoxide and hospital admissions and emergency room visits for myocardial infarctions. Although the risk of myocardial infarction hospital admissions and emergency room visits is expected to be elevated by about 5% on such days, the overall health burden related to a few days with higher concentrations corresponds to a very small fraction of the total air pollution-related burden.

Step 5. Compare the AQG level across critical health outcomes

No other health outcomes were evaluated in the systematic review.

Step 6. Assess certainty of the evidence

As mentioned in step 1, the evidence base supporting an association between 24-hour average carbon monoxide and hospital admissions and emergency room visits due to myocardial infarction was considered to be of moderate certainty.

Step 7. Consider new evidence

No new studies were found on the relation between myocardial infarction admissions/deaths and carbon monoxide exposure.

Step 8. Reconsider causality

The association between short-term carbon monoxide concentrations and myocardial infarctions was judged to be likely causal in the 2016 outcome prioritization framework (see [section 2.3.3](#)), based on assessments by Health Canada and US EPA, both of which date back to 2010 and have not been revised since. Of note, US EPA did not develop a standard for 24-hour carbon monoxide at the time, despite evidence of associations persisting at levels below 1 mg/m³ or 2 mg/m³ (Bell et al., 2009).

3.7.2.1 Interim targets

Interim targets are proposed as incremental steps in a progressive reduction of air pollution and are intended for use in areas where pollution is high. For a more detailed rationale for establishing and using interim targets, see [section 2.5.3](#).

The recommendation is a short-term (24-hour) carbon monoxide AQG level of 4 mg/m³, defined as the 99th percentile (equivalent to three to four exceedance days per year) of the annual distribution of 24-hour average concentrations.

An interim target 1 of 7 mg/m³ is proposed, as a point of reference to the existing 24-hour indoor WHO air quality guideline.

Table 3.23. Recommended short-term (24-hour) AQG level and interim targets for carbon monoxide^a

Recommendation	CO (mg/m ³)
Interim target 1	7
AQG level	4

^a Defined as the 99th percentile (equivalent to 3–4 exceedance days per year) of the annual distribution of 24-hour average concentrations.

If the number of myocardial infarctions in a population exposed to carbon monoxide for a day at the AQG level of 4 mg/m³ is arbitrarily set at 100, the number will be 106 in populations exposed to carbon monoxide at the interim target 1 level. This projection is based on the linear HR of 1.019 per 1-mg/m³ increase in carbon monoxide for hospital admissions due to myocardial infarction. At higher concentrations, the CRF may no longer be linear, which would change the numbers in this example.

3.8 Summary of recommended air quality guideline levels and interim targets

Table 3.24 summarizes the recommended AQG levels and interim targets for all pollutants. The evidence underlying all of the recommended AQG levels was rated as of high or moderate certainty and all recommendations are classified as strong according to the adapted GRADE approach (discussed in Chapter 2).

Table 3.25 shows the air quality guidelines for nitrogen dioxide, sulfur dioxide and carbon monoxide for short averaging times that were not re-evaluated and, therefore, remain valid.

Table 3.24. Summary of recommended long- and short-term AQG levels and interim targets

Pollutant	Averaging time	Interim target				AQG level
		1	2	3	4	
PM _{2.5} , µg/m ³	Annual	35	25	15	10	5
	24-hour ^a	75	50	37.5	25	15
PM ₁₀ , µg/m ³	Annual	70	50	30	20	15
	24-hour ^a	150	100	75	50	45
O ₃ , µg/m ³	Peak season ^b	100	70	–	–	60
	8-hour ^a	160	120	–	–	100
NO ₂ , µg/m ³	Annual	40	30	20	–	10
	24-hour ^a	120	50	–	–	25
SO ₂ , µg/m ³	24-hour ^a	125	50	–	–	40
CO, mg/m ³	24-hour ^a	7	–	–	–	4

^a 99th percentile (i.e. 3–4 exceedance days per year).
^b Average of daily maximum 8-hour mean O₃ concentration in the six consecutive months with the highest six-month running-average O₃ concentration.

Table 3.25. Air quality guidelines for nitrogen dioxide, sulfur dioxide and carbon monoxide (for short averaging times) that remain valid

Pollutant	Averaging time	Air quality guideline that remain valid
NO ₂ , µg/m ³	1-hour	200
SO ₂ , µg/m ³	10-minute	500
CO, mg/m ³	8-hour	10
	1-hour	35
	15-minute	100

Table 3.26 shows a side-by-side comparison of the 2005 air quality guidelines and the 2021 AQG levels.

Table 3.26. Recommended 2021 AQG levels and 2005 air quality guidelines

Pollutant	Averaging time	2005 air quality guideline	2021 AQG level
PM_{2.5}, µg/m³	Annual	10	5
	24-hour ^a	25	15
PM₁₀, µg/m³	Annual	20	15
	24-hour ^a	50	45
O₃, µg/m³	Peak season ^b	–	60
	8-hour ^a	100	100
NO₂, µg/m³	Annual	40	10
	24-hour ^a	–	25
SO₂, µg/m³	24-hour ^a	20	40
CO, mg/m³	24-hour ^a	–	4

^a 99th percentile (i.e. 3–4 exceedance days per year).

^b Average of daily maximum 8-hour mean O₃ concentration in the six consecutive months with the highest six-month running-average O₃ concentration.

3.8.1 Important AQG level updates to *Global update 2005*

The most important updates in these guidelines are listed below.

1. The PM_{2.5} annual AQG level has been lowered from 10 µg/m³ to 5 µg/m³. This reflects the new evidence of effects on mortality occurring at concentrations below 10 µg/m³. In this update of the air quality guidelines, an analysis was introduced to identify the most appropriate level of the long-term air quality guidelines that is more formalized than what was used in 2005. However, the change from 10 µg/m³ to 5 µg/m³ primarily reflects the new evidence about effects occurring at low levels of exposure.
2. The 24-hour AQG level for PM_{2.5} changed from 25 µg/m³ to 15 µg/m³. In 2005 a ratio of 2.5 was assumed between the 99th percentile of 24-hour average concentrations and annual averages. This ratio was changed to 3 based on empirical data from the very large MCC Collaborative Research Network (A. Gasparrini, London School of Hygiene and Tropical Medicine, unpublished data, 23 June 2020; Liu et al., 2019).

3. The PM₁₀ annual AQG level has been reduced from 20 µg/m³ to 15 µg/m³. This reflects the new evidence of effects on mortality occurring at concentrations below 20 µg/m³. In this update of the air quality guidelines, an analysis was introduced to identify the most appropriate level of the long-term air quality guidelines that is more formalized than what was used in 2005. However, the change from 20 µg/m³ to 15 µg/m³ primarily reflects the new evidence about effects occurring at low levels. It is important to note that the assessment of PM₁₀ was based on studies that had actually measured PM₁₀, without taking into consideration the ratios between PM₁₀ and PM_{2.5}. In 2005 based on empirical data, a PM₁₀ : PM_{2.5} ratio of 2 was used to establish the PM₁₀ AQG levels. The GDG notes that the empirical PM₁₀ : PM_{2.5} ratios have not changed, but the method used to derive the AQG levels has changed. The resulting PM₁₀ annual AQG level is less protective than the PM_{2.5} annual AQG level in most practical circumstances.
4. The 24-hour AQG for PM₁₀ changed from 50 µg/m³ to 45 µg/m³. In 2005 a ratio of 2.5 was assumed between the 99th percentile of 24-hour average concentrations and annual averages. This ratio was changed to 3 based on empirical data from the very large MCC Collaborative Research Network (A. Gasparrini, London School of Hygiene and Tropical Medicine, unpublished data, 23 June 2020; Liu et al., 2019). As a result of the combined effects of the new derivation procedure and the changed ratio, the 24-hour AQG level for PM₁₀ is not much lower in 2021 than in 2005. The resulting PM₁₀ 24-hour AQG level is less protective than the PM_{2.5} 24-hour AQG level in most practical circumstances.
5. A new long-term peak-season average ozone AQG level has been established. This is based on new evidence on the long-term effects of ozone on total mortality and respiratory mortality. The short-term AQG level was re-calculated using the protocols outlined in [section 2.5](#). The resulting short-term AQG level of 100 µg/m³ is the same as the 2005 short-term air quality guideline, which was based on morbidity and lung function effects. Therefore, in practical terms, the guidance for ozone has not changed.
6. The annual AQG level for nitrogen dioxide changed from 40 µg/m³ to 10 µg/m³. This was primarily because this update of the air quality guidelines is based on the effects of long-term nitrogen dioxide on all-cause mortality and respiratory mortality. The 2005 air quality guideline was based on morbidity effects observed in children exposed indoors to nitrogen dioxide from gas cooking. The chosen level was originally proposed in a document prepared by the International Labour Organization, UNEP and WHO (International Programme on Chemical Safety, 1997). It was justified as follows:

On the basis of a background level of 15 µg/m³ (0.008 ppm) and the fact that significant adverse health effects occur with an additional level of 28.2 µg/m³ (0.015 ppm) or more, an annual guideline value of 40 µg/m³ (0.023 ppm) is proposed. This value will avoid the most severe exposures (International Programme on Chemical Safety, 1997).

As is evident from this quotation, the annual AQG of 40 µg/m³ was in fact expected to be associated with “significant adverse health effects”. A background of 15 µg/m³ is not all that different from the AQG level of 10 µg/m³ that is recommended in this report.

7. Following the protocol established in [section 2.5](#), a new 24-hour AQG level of 25 µg/m³ for nitrogen dioxide was recommended. The 2005 1-hour AQG level of 200 µg/m³ was not re-evaluated. The GDG points out that in most practical circumstances, the 24-hour AQG level in this update is more stringent than the 2005 1-hour AQG level.
8. Following the protocol established in [section 2.5](#), a 24-hour AQG level for sulfur dioxide of 40 µg/m³ was recommended. This is based on a new evaluation of the effects of short-term sulfur dioxide concentrations on all-cause mortality and respiratory mortality. This AQG level is higher than the 2005 24-hour air quality guideline of 20 µg/m³. The 2005 air quality guideline was also primarily based on an evaluation of the short-term effects of sulfur dioxide on mortality. No formal method was applied to derive a guideline value in 2005. The considerations at the time were:

In consideration of (a) the uncertainty of sulfur dioxide in causality, (b) the practical difficulty of reaching levels that are certain to be associated with no effects and (c) the need to provide greater degrees of protection than those provided by the guidelines published in 2000, and assuming that reduction in exposure to a causal and correlated substance is achieved by reducing sulfur dioxide concentrations, there is a basis for revising the 24-hour guideline for sulfur dioxide downwards, adopting a prudent precautionary approach (WHO Regional Office for Europe, 2006).

The GDG argues that in comparison the recommended 24-hour AQG level of 40 µg/m³ is better justified, and coherent with the approaches followed in the recommendations for short-term AQG levels for the other pollutants covered in this report.

9. Following the protocol established in [section 2.5](#), a 24-hour AQG level for carbon monoxide of 4 mg/m³ was recommended. This is based on a new evaluation of the effects of short-term carbon monoxide concentrations on hospital admissions for myocardial infarction.

3.9 Supporting burden of disease calculations

To support discussions on the updating of AQG levels, WHO performed a rapid scenario analysis to explore the reductions in disease burden attributable to annual ambient PM_{2.5} globally (WHO, 2018) that would occur if the 2016 levels were reduced to the current interim target 1 (35 µg/m³), interim target 2 (25 µg/m³), interim target 3 (15 µg/m³), interim target 4 (10 µg/m³) and AQG levels.

The methods and results are described in more detail in Evangelopoulos et al. (2020). The methodology of this calculation was the same as in the GBD 2016 study, which used a set of non-linear, cause-specific exposure–response functions. These are not directly comparable to the linear CRFs reported in the systematic reviews produced for the purpose of AQG level derivation in this document. In addition, Evangelopoulos et al. (2020) did not perform a scenario analysis for the current AQG level, which was decided after their publication. However, the analysis was conducted for this document. For further methodological details, see GBD 2016 Risk Factors Collaborators (2017).

[Table 3.27](#) illustrates the total estimated number of deaths attributable to ambient PM_{2.5} in 2016 by WHO region and worldwide. In all these scenarios, the indicated levels are assumed to reflect the population-weighted mean exposure. The population-weighted mean is the average concentration in a sub-area (region or country) weighted by the distribution of the population within that sub-area, relative to its total population. This accounts for spatial relationships between locations of populations and concentrations, in contrast to area-weighting, which is simply the average concentration within a sub-area, irrespective of where the population may reside.

As an illustration, results show that if interim target 4 (equivalent to the 2005 air quality guideline) had been achieved in 2016, then in terms of population-weighted average, the estimated burden of disease would have been reduced substantially: achievement of interim target 4 would have resulted in a 47.8% decrease in total deaths attributed to PM_{2.5} exposure compared with the number calculated using the 2016 levels of exposure worldwide. The highest impact would have been observed in the WHO South-East Asia and African regions (reductions of 57% and 60%, respectively).

Meeting the interim targets would also have had a notable benefit on health, especially in those regions where exposures far exceed interim targets. Even if interim target 1 had been met, reductions of 20% and 14%, respectively, in burden of disease attributable to ambient PM_{2.5} would have been observed in the South-East Asia and Eastern Mediterranean regions.

Table 3.27. Region-specific and global deaths attributable to ambient PM_{2.5} under 2016 air pollution levels and percentage reduction through achievement of the recommended interim targets or AQG level^a

WHO region	Global/regional deaths & % reduction through achievement of interim target or AQG level ^a					
	Air pollution level, 2016	Interim target 1	Interim target 2	Interim target 3	Interim target 4	AQG level
African Region						
<i>n</i> (UI), in 000s	474 (411–547)	403 (329–481)	349 (270–429)	255 (182–351)	188 (126–284)	60 (30–142)
% reduction (UI)	–	14.5 (9.5–21.9)	26.2 (17.4–37.0)	45.9 (32.0–59.1)	60.4 (44.0–72.0)	87.3 (71.6–93.6)
Region of the Americas						
<i>n</i> (UI), in 000s	249 (204–306)	249 (204–306)	247 (202–304)	230 (185–286)	203 (159–258)	89 (49–144)
% reduction (UI)	–	0.0 (0.0–0.0)	0.6 (0.4–0.9)	7.4 (5.6–9.5)	18.2 (14.4–22.5)	64.1 (50.6–79.4)
South-East Asian Region						
<i>n</i> (UI), in 000s	1 351 (1193–1515)	1 078 (940–1 244)	948 (804–1 110)	742 (610–906)	580 (460–732)	223 (128–353)
% reduction (UI)	–	19.7 (16.3–25.1)	29.5 (24.7–36.55)	44.6 (38.0–52.8)	56.8 (49.3–64.5)	83.3 (74.8–90.3)
European Region						
<i>n</i> (UI), in 000s	464 (383–552)	463 (382–551)	457 (376–545)	436 (356–523)	385 (308–471)	157 (85–253)
% reduction (UI)	–	0.2 (0.1–0.2)	1.5 (1.2–1.9)	6.2 (5.1–7.7)	17.1 (14.2–20.4)	65.9 (52.0–81.5)

Table 3.27 contd

WHO region	Global/regional deaths & % reduction through achievement of interim target or AQG level ^a					
		Interim target 1	Interim target 2	Interim target 3	Interim target 4	AQG level
Eastern Mediterranean Region						
<i>n</i> (UI), in 000s	336 (301–369)	289 (255–322)	253 (220–287)	199 (169–236)	158 (130–194)	64 (37–96)
% reduction (UI)	–	13.8 (11.5–16.9)	24.3 (20.4–28.9)	40.4 (34.4–46.4)	52.6 (45.7–58.9)	80.7 (72.2–88.4)
Western Pacific Region						
<i>n</i> (UI), in 000s	1 278 (1 119–1 449)	1 160 (1 009–1 324)	1 024 (876–1 191)	818 (673–978)	643 (512–796)	248 (138–386)
% reduction (UI)	–	9.2 (7.9–11.2)	19.8 (17.2–23.9)	36.1 (31.7–42.5)	49.7 (44.2–56.5)	80.6 (71.8–88.8)
Global						
<i>n</i> (UI), in 000s	4 155 (3 685–4 662) ^b	3 646 (3 179–4 188)	3 276 (2 818–3 840)	2 677 (2 237–3 222)	2 155 (1 736–2 674)	848 (484–1 310)
% reduction (UI)	–	12.0 (9.7–15.5)	20.8 (17.0–26.1)	35.2 (29.4–42.3)	47.8 (40.8–55.2)	79.5 (70.1–87.9)

UI: uncertainty interval.

^a Based on 2016 figures and assuming all other relevant health factors remain unchanged.

^b These values are slightly different than the ones reported in the WHO Burden of Disease 2016 report (WHO, 2018) due to rounding.

Note: for the definition of uncertainty interval, see WHO (2018).

The scenario analysis showed that if the interim targets were achieved, the greatest benefit in terms of reduced health impact would be observed in countries with high PM_{2.5} concentrations and large populations. If population-weighted concentrations were to comply with the AQG level, then premature mortality could be reduced by as much as 45–50 deaths per 100 000 people.

On the other hand, much smaller changes in premature mortality would occur in high-income countries because in most cases the ambient PM_{2.5} concentrations are already below the interim targets.

The derived reductions in the health burden relate to national or WHO regional level, population-weighted mean concentrations. However, policy-makers may require compliance with the AQG level not just at the level of the population average but in all areas where people live. Therefore, [Table 3.27](#) underestimates the health benefits of full compliance with the AQG level for all locations.

Estimates of the ultimate population-weighted mean concentrations once interim targets or AQG levels have been achieved everywhere are not yet available; thus, the related benefits have not been described here. However, an impact assessment study provided estimates for a scenario in which the new PM_{2.5} interim target 4 (10 µg/m³) had been achieved throughout Switzerland, including at hot spots (Castro et al., 2020). Under this scenario, the population-weighted mean concentration of PM_{2.5} is expected to be only 83% of the interim target 4 value.

4

**Good practice
statements about
other PM types**

4.1 Introduction

The GDG decided not to formulate air quality guideline (AQG) levels for the specific types of PM (i.e. BC/EC, SDS and UFP) that were prioritized during the preliminary phase. This decision was made because the GDG considered that the quantitative evidence on independent adverse health effects from these pollutants was still insufficient at the time of deriving the AQG levels. The GDG decided that the best manner for addressing these pollutants in the guideline document was to formulate good practice statements (discussed in [section 2.5.3](#)), as outlined in the *WHO handbook for guideline development, 2nd edition* (WHO, 2014a). That is, when a GDG is confident that a large body of diverse evidence that is hard to synthesize indicates that the desirable effects of a particular course of action far outweigh its undesirable effects (WHO, 2014c).

[Section 4.4](#) (on SDS) is substantially more detailed than [sections 4.2](#) (on BC/EC) and [4.3](#) (on UFP), and includes several statements on the mitigation measures for population exposure to pollution from SDS. This is intentional, since the mitigation of exposure to pollution from SDS requires different, less standard, approaches than those related to anthropogenic pollution (black carbon and UFP), that focus on source emission reduction.

4.2 Black carbon/elemental carbon

There is concern over the potential impacts on health of black carbon, and a review of the literature by WHO (WHO Regional Office for Europe, 2013a) concluded that evidence links black carbon particles with cardiovascular health effects and premature mortality, for both short- (24-hour) and long-term (annual) exposures. In studies that take black carbon and PM_{2.5} into account simultaneously, associations remained robust for black carbon (WHO Regional Office for Europe, 2013a). Even when black carbon may not be the causal agent, black carbon particles are a valuable additional air quality metric for evaluating the health risks of primary combustion particles from traffic, including organic particles, that are not fully taken into account with PM_{2.5} mass levels. An assessment by US EPA also summarized the evidence of associations between a series of health effects and black carbon concentrations, with conclusions similar to those of the earlier WHO review (US EPA, 2019a).

Black carbon is a measure of airborne soot-like carbon that is determined with optical methods. It is closely related to the mass concentration of elemental carbon (i.e. carbon in various crystalline forms) that is ascertained chemically. BC/EC is typically formed through the incomplete combustion of fossil fuels, biofuel and biomass, and is emitted from both anthropogenic and natural sources.

It consists of pure carbon in several forms, and the relevant particle size fraction can include known carcinogens and other toxic species. Black carbon is a powerful climate-warming agent that acts by absorbing heat in the atmosphere and by reducing albedo (the ability to reflect sunlight) when deposited on snow and ice (Bond et al., 2013).

To address concerns about the health and environmental effects of BC/EC, three good practice statements (Box 4.1) have been formulated. The following sections provide a rationale for each of the statements.

Box 4.1. Good practice statement – BC/EC

Based on insufficient evidence to propose an AQG level, the GDG decided to formulate the following three good practice statements on BC/EC directed to countries and regional authorities.

1. Make systematic measurements of black carbon and/or elemental carbon. Such measurements should not replace or reduce the existing monitoring of pollutants for which guidelines currently exist.
2. Undertake the production of emission inventories, exposure assessments and source apportionment for BC/EC.
3. Take measures to reduce BC/EC emissions from within the relevant jurisdiction and, where considered appropriate, develop standards (or targets) for ambient BC/EC concentrations.

4.2.1 Rationale for statement 1 – measurement of black carbon and/or elemental carbon

Black carbon is a measure of airborne soot-like carbon that is defined operationally by the method used for its measurement, that is, the optical absorption of specific wavelengths by particles collected on a filter. The extent of optical absorption is then converted to black carbon concentrations expressed in units of $\mu\text{g}/\text{m}^3$ via a calibration based on a mass measurement of elemental carbon. Continuous measurements of black carbon are often made with aethalometers, which use

an optical approach and a standard conversion to mass concentration. Black carbon is a metric similar to elemental carbon, with the latter being a chemical measurement; both are measures of soot-like (graphitic) carbon. Elemental carbon is also defined operationally; it is usually determined by thermo-optical (chemical) techniques, in which the carbonaceous material is driven off the filter at high temperatures in an oxygen-rich environment. There is a close relationship between black carbon and elemental carbon mass measurements, which (to a very good approximation) is linear, but the slope may vary by the specific PM mixture and should be verified locally to reflect local conditions.

There are several measurement methods for black carbon. Hansen (2005) provides a detailed description of a common measurement method. EU Directive 2008/50/EC (European Parliament & Council of the European Union, 2008) requires measurements of elemental carbon, but filter measurements of black carbon or related optical parameters such as absorbance are much simpler and cheaper to make than elemental carbon measurements and, therefore, are much more applicable globally. For example, Jeronimo et al. (2020) describe a low-cost method of measurement(). It should be noted further that black carbon and its optical properties are more relevant to the climate than elemental carbon.

Elemental carbon is required to be measured by EU Directive 2008/50/EC, and the European Committee for Standardization (CEN) has developed a measurement method (CEN, 2017; Brown et al., 2017). As yet, no similar standard exists for black carbon but descriptions of methods of reporting have been given in the EU-funded Aerosol, Clouds and Trace Gases Research Infrastructure (ACTRIS, 2020) and described by the World Meteorological Organization (WMO) (Petzold et al., 2013). Although recommending a standard method for BC/EC monitoring is outside of the scope of WHO air quality guidelines, defining a standard and easy-to-apply method by relevant organizations would facilitate the recommended monitoring.

4.2.2 Rationale for statement 2 – production of emission inventories, exposure assessments and source apportionment for BC/EC

BC/EC emissions arise from incomplete or inefficient combustion and, hence, tend to come from local sources in urban areas and from specific combustion sources such as solid fuel or fuel-oil-fired power plants. Sources include passenger cars, buses, and trucks and other heavy goods vehicles, particularly diesel engines (both on-road and off-road); residential solid fuel use such as wood and coal, as well as liquid fuel such as kerosene; and power plants using heavy fuel oil and coal. Shipping, agricultural waste burning and wildfires are also sources of black BC/EC.

Emission factors for BC/EC are often uncertain, but guidance is available via several guidebooks (EEA, 2019; US EPA, 2019b).

The nature of these local sources means that, in general, exposures to BC/EC are more spatially variable than the total PM_{2.5}, so exposure assessments could be more challenging but more informative about the true spatial contrasts in exposures. Assessments could be based on models with fine spatial resolution as well as on measurements. Modelling approaches might involve small-scale urban dispersion models based on Gaussian plume methods, boundary-layer scaling plume models, urban and large-scale 3D chemical transport models, and land-use regression models. Use of well-formulated emission inventories coupled with dispersion air quality models will yield the source apportionment necessary to formulate abatement policies to reduce air pollutants.

4.2.3 Rationale for statement 3 – implementation of measures to reduce BC/EC, including the development of standards where appropriate

Epidemiological studies have already been carried out using black carbon and elemental carbon as exposure metrics (Janssen et al., 2012; US EPA, 2019a). Most studies have been in Europe and North America, and further work in other areas of the world – as well as in Europe and North America – would be valuable, particularly since there now exists recommendations for reporting black carbon measurements, as described above.

There has been considerable discussion in the past over the differential toxicity of the various components of PM_{2.5}, but with no clear consensus so far. However, the earlier review of the literature in the WHO REVIHAAP project did state that PM components deriving from combustion were particularly toxic (WHO Regional Office for Europe, 2013a). In addition, much of the consideration of this issue has focused on the question of whether or not there is a better metric than total PM_{2.5} mass to account for the associations demonstrated in the epidemiological studies. It seems unlikely that a clear answer to this question will be forthcoming in the near future and, indeed, in terms of actions to improve public health this may not be the right question to ask.

A more appropriate question to ask may be whether there is an additional metric or component that countries might target for emission reductions next to the total PM_{2.5} mass. For many countries or regions – where the incomplete or inefficient combustion of carbon-containing material is common and where a substantial part of population exposure to PM is due to BC/EC – actions to reduce BC/EC would seem to be an appropriate complementary strategy and a good practice to strengthen clean air policies. BC/EC particles contain known toxic constituents such as carcinogens and are co-emitted with other toxic pollutants that are also products of incomplete combustion, that is, carbon monoxide, polycyclic aromatic

hydrocarbons and VOCs. Using total $\text{PM}_{2.5}$ as a control metric could mean that targets could be met with no specific pressure to reduce the primary combustion particles and known toxic constituents of BC/EC. Moreover, control of BC/EC requires paying stronger attention to spatial hot spots of primary PM pollution, which are less well captured or identified with $\text{PM}_{2.5}$ mass concentrations; thus, compliance with $\text{PM}_{2.5}$ standards may not necessarily guarantee low enough levels of elemental carbon for compliance.

In addition, given the carcinogenicity of elemental carbon, the strategy to keep its concentrations as low as possible is in line with the prevailing risk reduction strategy generally pursued for carcinogens. On the other hand, the control of total $\text{PM}_{2.5}$ mass in many areas is not totally under the control of a single country or jurisdiction – in many areas long-range transport of secondary PM is a significant contributor of $\text{PM}_{2.5}$ mass. Including BC/EC as an indicator of local emission reductions might compensate for the limited ability to influence total $\text{PM}_{2.5}$ concentration. Finally, there are sound climatic reasons for reducing black carbon concentrations: along with methane and ozone, black carbon is one of the most important short-lived climate pollutants, the reduction of which could produce rapid improvements in actions to stop climate warming (Bice et al., 2009; Bond et al., 2013; Miller & Jin, 2019).

To illustrate typical ambient levels of black carbon, the results from the United Kingdom Black Carbon Network can be used (Butterfield et al., 2016). Annual mean concentrations of black carbon measured in 2015 were $0.2\text{--}0.4\text{ }\mu\text{g}/\text{m}^3$ in rural sites, $1.0\text{--}2.0\text{ }\mu\text{g}/\text{m}^3$ in urban background stations and $1.4\text{--}5.1\text{ }\mu\text{g}/\text{m}^3$ in roadside locations. Black carbon made up a significant proportion of PM mass concentration at roadside sites, contributing to 12–21% of PM_{10} and 18–32% of $\text{PM}_{2.5}$. In an urban background location, these proportions were 5% and 9%, respectively, and in rural background locations were 2–3% of each of the PM fractions.

Black carbon mean concentrations observed in epidemiological studies ranged from $0.65\text{ }\mu\text{g}/\text{m}^3$ to $3.9\text{ }\mu\text{g}/\text{m}^3$, while for elemental carbon the means generally ranged from $0.47\text{ }\mu\text{g}/\text{m}^3$ to $3.5\text{ }\mu\text{g}/\text{m}^3$ and reached $7.5\text{--}8.8\text{ }\mu\text{g}/\text{m}^3$ in individual studies from Asia (Khreis et al., 2017; Luben et al., 2017).

Illustrative annual mean concentrations where statistically significant associations with health outcomes have been found were $1.08\text{--}1.15\text{ }\mu\text{g}/\text{m}^3$ for black carbon and $0.5\text{--}0.8\text{ }\mu\text{g}/\text{m}^3$ for elemental carbon (Luben et al., 2017).

Although the evidence base is insufficient to set a certain AQG level to provide a basis for legally binding limit values, adoption of an air quality standard or

target (e.g. in the form of a concentration reduction obligation) might be a good instrument to force local actions on BC/EC reduction.

Strategies to control BC/EC emissions should consider local conditions. They may address emissions from biomass and other polluting fuels used for cooking and heating, emissions from diesel vehicles and off-road machinery (World Bank, 2014), and emissions from agricultural (and communal) waste burning and from wildfires.

4.3 Ultrafine particles

UFP are generally considered as particulates with a diameter less than or equal to 0.1 μm , that is, 100 nm (typically based on physical size, thermal diffusivity or electrical mobility). There was already considerable evidence on the toxicological effects of UFP at the time that *Global update 2005* was being prepared, which was acknowledged in the document (WHO Regional Office for Europe, 2006). However, it was stated that the evidence from epidemiology was insufficient to recommend guidelines for UFP. Since then, the body of epidemiological evidence has grown, and two systematic reviews have assessed scientific research papers published from 1997 to 2017 (HEI, 2013; Ohlwein et al., 2019), documenting the rising number of studies being conducted. The studies demonstrated short-term effects of exposure to UFP, including mortality, emergency department visits, hospital admissions, respiratory symptoms, and effects on pulmonary/systemic inflammation, heart rate variability and blood pressure; and long-term effects on mortality (all-cause, cardiovascular, IHD and pulmonary) and several types of morbidity. However, various UFP size ranges and exposure metrics were used, preventing a thorough comparison of results across studies (US EPA, 2019a). Therefore, there was a consensus in the GDG that the body of epidemiological evidence was not yet sufficient to formulate an AQG level.

At the same time, however, there is a large body of evidence from exposure science that is sufficient to formulate good practice advice. The most significant process generating UFP is combustion and, therefore, the main sources of the UFP include vehicles and other forms of transportation (aviation and shipping), industrial and power plants, and residential heating. All of these utilize fossil and biofuels, as well as biomass. Since everyone is exposed to the emissions from these sources, exposure to UFP is of concern.

To address concerns about the health and environmental effects of UFP, four good practice statements (Box 4.2) have been formulated. The following sections provide a rationale for each of the statements.

Box 4.2. Good practice statement – UFP

The GDG decided to formulate the following four good practice statements on UFP to guide national and regional authorities and research towards measures to reduce ambient ultrafine particle concentrations.

1. Quantify ambient UFP in terms of particle number concentration (PNC) for a size range with a lower limit of ≤ 10 nm and no restriction on the upper limit.
2. Expand the common air quality monitoring strategy by integration of UFP monitoring into existing air quality monitoring. Include size-segregated real-time PNC measurements at selected air monitoring stations in addition to, and simultaneously with, other airborne pollutants and characteristics of PM.
3. Distinguish between low and high PNC to guide decisions on the priorities of UFP source emission control. Low PNC can be considered < 1000 particles/cm³ (24-hour mean). High PNC can be considered $> 10\,000$ particles/cm³ (24-hour mean) or $20\,000$ particles/cm³ (1-hour).
4. Utilize emerging science and technology to advance approaches to the assessment of exposure to UFP for application in epidemiological studies and UFP management.

4.3.1 Rationale for statement 1 – quantification of ambient UFP

PNC is the most common measure used to characterize UFP, and the measurement technologies for this are well established; however, there is no agreed international (or national) standard method on this as yet. The existing instrumental methods for PNC measurement do not provide information on particles in the UFP-specific size range (< 100 nm), and both their lower and upper detection limits vary; the lower limit typically ranges from 2 nm to 20 nm. Therefore, the term quasi-ultrafine refers to particles substantially smaller than $1\ \mu\text{m}$ but larger than 100 nm. In this document, PNC refers to the number concentration of quasi-UFP. The choice of the lower cut-off of measurement is usually critical, since the majority of UFP are often within a smaller size range, particularly in environments affected by fresh combustion emissions; the upper range is less critical. The error (underestimation) for lower size limits up to 10 nm can be calculated and

corrected for. The uncertainty in the calibration of instruments measuring PNC is based on a standardized methodology (ISO 27891:2015 (ISO, 2015)) and varies between 30% for lower concentrations (< 1000 particles/cm³) to 10% for typical urban background concentrations (about 10 000 particles/cm³) (Morawska et al., 2008; Thinking Outside the Box team, 2019).

4.3.2 Rationale for statement 2 – expanding UFP monitoring

Whereas the theories underpinning UFP emission and formation processes are generally well developed, local understanding of the origin of UFP (primary/secondary, specific sources) and their chemical composition (solid/liquid, organic carbon/elemental carbon, metals and toxicity) is generally very limited in most parts of the world; UFP and precursor emission inventories and PNC source apportionments hardly exist. Generally, there is very little or no relationship between PNC and mass concentration of larger particles (PM_{2.5}), and the existence and degree of relationship between PNC and traffic-emitted gaseous pollutants (carbon monoxide and NO_x) or black carbon varies, depending on location. Therefore, no other pollutant is a good proxy for UFP. However, quantitative knowledge of UFP is needed, since focusing only on PM_{2.5} may result in overlooking the impact of UFP and there is no evidence that mitigating particle mass only (PM₁₀, PM_{2.5}), as the existing air quality measures do, will necessarily lead to a reduction in UFP (ANSES, 2019; Thinking Outside the Box team, 2019).

UFP monitoring would provide a good base for evaluation of effects of pollution mitigation and could be used for future epidemiological studies on the health effects of UFP and for distinguishing these effects from the effects of other pollutants. Note that the UFP measurements should not hinder the existing measurements of pollutants for which guidelines currently exist.

4.3.3. Rationale for statement 3 – distinction between low and high PNC

In urban areas, road traffic and other forms of transportation (aviation and shipping) are usually the main sources of UFP. These particles are emitted directly by the sources or formed in the air from gaseous precursors that are usually also emitted by the same sources. In addition, emissions from industrial sources, power plants, residential heating and biomass burning are sources of UFP, contributing to various extents to the UFP concentrations in urban air. Due to the nature of source emissions and particle formation, the spatiotemporal variation of the absolute level of PNC across a single city area is substantially larger than the spatiotemporal variation of larger particles (measured as particle mass concentration), for example PM_{2.5}. Based on literature review and expert opinion, there is general agreement that concentrations below 1000 particles/cm³ (24-hour mean), typically observed in environments not affected by anthropogenic emissions,

can be considered as low (de Jesus et al., 2019; Thinking Outside the Box team, 2019). It is proposed that 24-hour mean concentrations exceeding the typical levels observed in urban background areas (10 000 particles/cm³) or 1-hour mean concentrations exceeding levels found usually in all urban microenvironments (20 000 particles/cm³) can be considered high.

4.3.4 Rationale for statement 4 – utilization of emerging science and technology to advance population exposure assessment

Estimation of the population exposure to UFP in short- and long-term epidemiological studies (including repeated peak exposures) is significantly more complex than assessment of the exposure to PM_{2.5} and PM₁₀. It would be highly beneficial to develop and utilize standardized measurement procedures that enable meaningful comparison between the results from different studies, which is of particular significance for human exposure and epidemiological studies. Considering the complexity of the measurements, variety of instruments available and difference in the aims of the measurement/monitoring, it is not likely that standard methods to measure UFP will be accepted/established in the foreseeable future. However, scientific progress on many fronts makes personal exposure assessment possible by providing estimates of variation among the different results based on differences in the instruments being used or their settings. Furthermore, there are modelling tools that can allow obtaining the source contributions to UFP concentrations and can increase the robustness of meta-analysis of multicity data for epidemiological studies. Therefore, future long-term studies might consider modelling, increasing the number of monitors or utilizing mobile platforms to collect data across larger urban areas in order to cover the spatial variability in cities (ANSES, 2019; Thinking Outside the Box team, 2019).

4.4 Sand and dust storms

At their first meeting in 2016, the GDG members agreed that SDS needed to be addressed in this update of the WHO air quality guidelines. Dealing with SDS has become a growing priority within the global community, as reflected by the adoption of several resolutions by the UN General Assembly (UN, 2016, 2017, 2018b, 2019b). Improving the implementation of sustainable land management practices, taking measures to prevent and control the main factors of SDS, and improving the development of early warning systems as tools to combat SDS feature among the key priorities for action (UNEP, 2016b).

The discussion and arguments reported here have to take into account the fact that there are countries that are located in desert regions and countries that do not include desert land but are affected by desert dust. SDS events that originated

in specific regions can impact various countries owing to the proven long-range transport of dust over countries and, even, continents (Tanaka & Chiba, 2006; UNEP, WMO & UNCCD, 2016; Middleton, 2017). Indeed, a relevant issue to take into consideration is the difference between geographical regions such as the Middle East, the Sahel and northeast Asia, which have considerable SDS events, and others such as eastern Asia, southern Europe, parts of North America, and western Africa, that have experienced various episodes of transported desert dust. Desert dust is usually composed of mineral particles that originate from arid and semi-arid land surfaces, but “sometimes, after having travelled great distances, they may be observed over areas where no dust or sand covers the ground” (WMO, 2020b). SDS are usually prompted by intense winds that elevate large amounts of sand and dust from bare, dry soils into the air (WMO, 2020a). It has to be considered that there is no precise distinction between sand storms and dust storms, since there is a continuum of particle sizes in any storm. Importantly, desert dust events have coincided with substantial increases in measured concentrations of both the PM₁₀ and PM_{2.5} size fractions. Furthermore, research from southern Europe suggests an increased accumulation of anthropogenic pollutant concentration during events of transported dust, likely owing to a number of related meteorological phenomena (Querol et al., 2019a).

The WHO-commissioned toxicological review of 67 experimental studies concluded that SDS may be a significant risk factor for inflammatory and allergic lung diseases such as child and adult asthma. Studies, mainly using doses that reflect or at least approach real-world exposures during a dust event, have demonstrated that sand dust particles collected from surface soils (i.e. at the source) and dust-storm particles sampled at remote locations away from the source (and as such, mixed with industrial pollutants and microorganisms) induce inflammatory lung injury and aggravate allergen-induced tissue eosinophilia. No studies were identified that included specific cardiovascular end-points. In vitro findings suggest desert dust surface reactions may enhance the toxicity of aerosols in urban environments (Fussell & Kelly, 2021).

The WHO-commissioned systematic review of adverse health effects from SDS summarized the evidence from 93 studies conducted worldwide. The studies indicate an overall effect of desert dust on cardiovascular mortality and respiratory morbidity, but the evidence is still inconsistent when accounting for sources of PM in different geographical areas (Tobias et al., 2019a, 2019b). In addition, previously published reviews, systematic or not, reported inconsistent results across studies and geographical regions (de Longueville et al., 2013; Hashizume et al., 2010; Karanasiou et al., 2012; Zhang et al., 2016). An existing limitation in the scientific literature is the lack of studies on the long-term health effects of SDS. The health

outcomes studied more frequently include (i) daily mortality, natural-cause and cause-specific; (ii) cardiovascular and respiratory morbidity; and (iii) morbidity as documented in hospital admissions and emergency room visits, mainly for cardiovascular and respiratory diseases, including asthma and COPD. Overall, the four reviews (de Longueville et al., 2013; Hashizume et al., 2010; Karanasiou et al., 2012; Zhang et al., 2016) had similar conclusions, suggesting that potential health effects linked to SDS may include increased cardiovascular mortality and respiratory hospital admissions. A range of other health impacts, such as injuries and death from transport accidents due to reduced visibility or the potential implications for disease incidence of meningitis and coccidioidomycosis, have also been reported (Ashley et al., 2015; Baddock et al., 2013; Goudie, 2014). The published studies differed in terms of settings, assessment methods for SDS exposure, lagged exposures examined and epidemiological study designs applied. Moreover, none of the previous reviews attempted to assess the quality of the evidence across the published studies.

The available evidence comes from studies that assessed the health effects of dust events as a binary risk exposure (mainly conducted in eastern Asia), comparing the occurrence of health events during dust and non-dust days, and from studies that considered dust events as an effect modifier for the health effects of any given PM fraction (mainly in southern Europe). Studies considering the effects of desert dust and anthropogenic PM (APM) concentrations independently revealed different effects in eastern Asia (higher association with specific cardiovascular mortality outcomes and ambulance calls related to Asian dust than to suspended PM) and southern Europe (similar health effects for Saharan dust and APM). When the role of APM during dust events was considered, the health effects of APM appeared to be stronger during dust days than during non-dust days. It should be noted that only studies considering short-term exposure have been conducted; there has been no study on the health effects of long-term exposure to sand and desert dust. The populations most susceptible to suffering the short-term effects of suspended particulates are considered to be older persons, individuals with chronic cardiopulmonary disorders, and children (Goudie, 2014).

Based on the available studies, the GDG agreed that formulating an AQG level for SDS was not possible due to insufficient evidence on quantitative and qualitative health risk-related characteristics of SDS. The GDG decided that the best manner for addressing SDS in the guideline document was to formulate qualitative practical recommendations focused on the likely consequences of desert dust and on options for mitigating it. Potential interventions can be part of short- or long-term strategies. Examples of possible short-term options outlined by the GDG in different meetings included: (i) strengthening and/or establishing

air quality management programmes; (ii) measuring PM components for the purpose of source apportionment; (iii) conducting research on health impacts and epidemiological studies; and (iv) cleaning up road dust on streets. During the discussions other options were also mentioned: (i) alerting public health authorities and vulnerable populations of increased levels of SDS; (ii) reducing local emissions from anthropogenic sources of dust and other pollutants during dust episodes; (iii) informing the public about personal interventions to reduce outdoor and indoor air pollution sources; and (iv) demonstrating the impact of policies towards lowering anthropogenic pollution (Argyropoulos et al., 2020; Katra & Krasnov, 2020; Querol et al., 2019b).

Long-term mitigation interventions are more complex. A review by Middleton & Kang (2017) classified interventions to mitigate SDS hazards into measures to prevent wind erosion occurring at source and measures to address the atmospheric transport of the particles and their deposition. If wind erosion is reduced, land degradation can be halted and eventually reversed and, in turn, SDS impacts can be mitigated. In agriculture, for example, a number of techniques are available for wind erosion control, including those that minimize the actual risk (e.g. cultivation practices such as minimum tillage) and those that minimize the potential risk (e.g. planting windbreaks) (Middleton & Kang, 2017). In general terms, long-term strategies such as reforestation plans have been implemented at various scales and for many years in different places; these were also meant as climate change mitigation measures (Jindal, Swallow & Kerr, 2008; UNEP, WMO & UNCCD, 2016).

All of the actions that address the impacts of SDS associated with particle transport and deposition include a range of monitoring, early warning, forecasting and communication activities. It is worth emphasizing that there is always a need to understand the context when discussing or implementing the good practices recommended in [Box 4.3](#). Rationales for each of the good practice statements follow [Box 4.3](#).

At the local, national and regional levels, the potential success of the implementation of these good practices is conditioned by actions that address the impacts of SDS with a range of monitoring, early warning, forecasting and communication activities. Other planned short-term actions – in general, relevant and desirable for reducing the overall impact of air pollution – can, if implemented, also decrease the exposure to SDS. These include (i) alerting public health authorities and vulnerable populations of increased levels of air pollution, in particular of SDS; (ii) reducing local emissions from anthropogenic sources of dust and other pollutants, in particular during dust episodes; (iii) informing the public

about personal interventions to reduce outdoor and indoor air pollution sources, in particular during SDS episodes, as sheltering during SDS episodes is sometimes the only feasible intervention (indoor air quality should be better than outdoor); and (iv) demonstrating the impact of policies towards lowering anthropogenic pollution. These actions are the mandate of national or local authorities, and international organizations can support policies by providing data, expertise and support.

Box 4.3. Good practice statement – SDS

Considering the available evidence, the GDG decided to formulate the following five good practice statements on SDS for frequently affected areas.

1. Maintain suitable air quality management and dust forecasting programmes. These should include early warning systems and short-term air pollution action plans to alert the population to stay indoors and take personal measures to minimize exposure, and subsequent short-term health effects, during SDS incidents with high levels of PM.
2. Maintain suitable air quality monitoring programmes and reporting procedures, including source apportionment activities to quantify and characterize the PM composition and the percentage contribution of SDS to the overall ambient concentration of PM. This will enable local authorities to target local emissions of PM from anthropogenic and natural sources for reduction.
3. Conduct epidemiological studies, including those addressing long-term effects of SDS, and research activities aimed at better understanding the toxicity of the different types of PM. Such studies are especially recommended for areas where there is a lack of sufficient knowledge and information about the health risk due to frequent exposure to SDS.
4. Implement wind erosion control through the carefully planned expansion of green spaces that considers and is adjusted to the contextual ecosystem conditions. This calls for regional collaboration among countries in the regions affected by SDS to combat desertification and carefully manage green areas.

Box 4.3 contd

5. Clean the streets in those urban areas characterized by a relatively high population density and low rainfall to prevent resuspension by road traffic as a short-term measure after intense SDS episodes with high dust deposition rates. Cleaning can be done by washing and/or sweeping. For the former, non-drinking, underground water from the subway drainage system or treated urban waters should be used (Querol et al., 2019a). This intervention is not feasible in many countries where water is scarce. In such cases, minimizing some of the local urban sources of dust such as construction and demolition activities can be a better alternative intervention. Before planning street cleaning, local authorities should:
 - assess the magnitude of the problem;
 - evaluate rainfall statistics;
 - select the streets that are most critically affected by the dust load situation;
 - ascertain the accumulation rate of sediments; and
 - determine the most effective cleaning method (e.g. frequency, timing and cleaning machine characteristics).

In partnership with other UN agencies, in particular, WMO, research institutes and academic institutions, WHO can ensure expertise and support in relation to dust measurements and their impacts. For example, the WHO Global Ambient Air Quality Database on air pollution, which is updated on a voluntary basis, can strengthen the adoption of good practices by providing a global framework of analysis. This can occur if countries affected by SDS send the WHO Global Database on Air Quality, for a given year, lists of affected zones, cities and agglomerations; information on concentrations and sources; and evidence demonstrating that observed PM concentrations are attributable, at least in part, to SDS episodes. This may provide the basis for different health impact (mortality and morbidity) calculations of air pollution that take into account the SDS contribution. The influence of SDS on air quality management is potentially very significant in orienting decisions, for example on setting national or local standards. Although this process should be based on this update of the WHO air quality guidelines and its AQG levels as the benchmark for setting standards, the rules concerning compliance assessment could be adjusted to accommodate local SDS risks.

4.4.1 Rationale for statement 1 – strengthening and/or establishing air quality management programmes

Preparedness and emergency response procedures in depositional areas need to cover diverse sectors such as public health surveillance, hospital services, air and ground transportation services, and public awareness and resilience. Since emergency response services are generally applied at local level, further subnational-level reviews and planning are needed.

A review by Querol et al. (2019b) suggested that setting up early warning systems for SDS by relevant authorities is an appropriate action to (i) inform exposed and vulnerable populations about behavioural measures that minimize the risks of high dust exposure levels; and (ii) implement special policy and regulatory measures at the local and regional levels to decrease anthropogenic air pollution emissions during dust episodes.

WMO established the Sand and Dust Storm Warning Advisory and Assessment System (SDS-WAS) (WMO, 2020c) to improve capabilities for more reliable SDS forecasting, intended for 40 of its Member States, with the Northern Africa-Middle East-Europe Node hosted by Spain, the Asian Node hosted by China, and the Pan-American Node with its Regional Center hosted by the United States and Barbados, respectively. The SDS-WAS mission is to achieve comprehensive, coordinated and sustained observations and modelling capabilities for SDS in order to improve SDS monitoring to increase the understanding of the dust processes and enhance dust prediction capabilities (WMO, 2020c).

Akhlaq, Sheltami & Mouftah (2012) provided an overview of the tools available for SDS prediction and detection, including data requirements and modelling approaches. Technologies include lookout towers, video-surveillance, sensory information, satellite imagery and unmanned aerial vehicles. The authors note that the best approach to use depends on the type of SDS, but that a hybrid approach consisting of wireless sensor networks and satellite imagery is appropriate for detecting and predicting all types of SDS.

The authorities in charge of the warning system should assess the most appropriate means to disseminate alerts to the population. Several means may be considered, such as media coverage, dedicated websites, messaging through social media and dedicated smartphone apps. It is also important to define the target population and identify vulnerable populations that can be particularly affected by SDS, as well as the facilities and other infrastructure that may be needed for such events. The involvement of health professionals and, in particular, of the medical profession should be considered, for example, general

practitioners who, knowing the population, can rapidly identify susceptible individuals based on their age, comorbidities, socioeconomic status or social isolation. Although the evidence on adverse health effects from SDS remains preliminary, there is some literature suggesting the effectiveness of public health alerts in promoting behavioural change. Messages that are generally issued by authorities include the following: staying indoors (appropriate in many settings), avoiding exposure, refraining from exercise, following asthma plans (for asthmatic patients), driving with care (for cases of SDS affecting visibility such as dry thunderstorms or haboob), and visiting the doctor if respiratory or cardiovascular symptoms occur (Middleton & Kang, 2017; WHO, 2020a).

Although there is evidence of the cost-effectiveness of early warning systems, especially for those related to weather services, there is no direct evidence for SDS. To be cost-effective, four elements must be present in any early warning system: knowledge of risks, monitoring and alert services, communication, and response capability. Systems are typically cost-effective when the monitored event is relatively frequent, significant harms can occur and there are affordable preventive measures (Rogers & Tsirkunov, 2010; World Bank, 2019). Specifically, it is not just the frequency of events but their intensity that should be considered. However, there is no cut-off, that is, no specific number of episodes per year, to orient decisions. This issue is similar to considering alert systems for wildfires that can affect an area; tools are available to assess the air quality impacts of such events, including their frequency and intensity. If these events are only rare and mild, usually a conventional weather forecast is sufficient to warn the public. These systems and their structure should take into account existing time series of events and evaluate the potential health impacts using epidemiological methods and tools.

Querol et al. (2019b) provided an example of the system established in Portugal and Spain as good practice. The system consists of three modules that allow SDS predicting, detecting SDS when they occur, and quantifying the daily contributions of desert dust to ambient PM_{2.5} and PM₁₀ concentrations.

4.4.2 Rationale for statement 2 – strengthening air quality monitoring programmes through identification of dust sources

SDS are usually prompted by intense winds that elevate large amounts of sand and dust from bare, dry soils into the air and transport them for long distances. As a result of this phenomenon, approximately 40% of aerosols in the troposphere are dust particles derived from wind erosion. The main areas from which mineral dust originates are the arid regions of northern Africa, the Arabian Peninsula, and central and eastern Asia (WMO, 2020a). Saharan dust may contribute more than

60% of the total PM₁₀ concentration in Mediterranean countries and the Middle East during a strong dust pollution event (Pey et al., 2013; Querol et al., 2009). This may lead to exceedances of the daily average interim target 4 value for PM₁₀ of 50 µg/m³. Causes of SDS are affected by direct and indirect drivers in natural ecosystems, direct and indirect drivers in human-dominated ecosystems, and land degradation feedback processes (UNEP, WMO & UNCCD, 2016). In recent centuries, human activities and climate change have aggravated the problem of desert storm generation. The natural composition of desert dust can be affected by several human sources (Mori et al., 2003; Rodríguez et al., 2011). This makes the distinction between natural PM and APM sources and assessment of the health effects of desert dust difficult (Perez & Künzli, 2011; Querol et al., 2019b).

A review commissioned by WHO (Querol et al., 2019b) suggested that acquiring reliable exposure data for source apportionment is a first critical step for epidemiological and health impact assessment studies of SDS. For desert dust, Querol et al. (2019b), based on earlier work by Escudero et al. (2007), recommended the following procedure for source apportionment as a method to quantify desert dust contributions to PM levels for air quality reporting purposes.

- Collect daily PM_{2.5} and PM₁₀ data, measured at remote or regional background air quality monitoring stations close to the urban area under evaluation.
- Calculate the 30-day moving 40th percentile PM concentration without taking into account PM levels on the SDS days. The 40th percentile equates to the RBPM₁₀ and RBPM_{2.5} levels without the dust contribution.
- Determine the net dust PM (NDPM) levels in PM₁₀ and PM_{2.5} (NDPM₁₀ and NDPM_{2.5}) for the regional background by subtracting RBPM₁₀ and RBPM_{2.5} from the bulk PM₁₀ and PM_{2.5} levels at the reference regional background-monitoring site.
- At the nearby urban area, NDPM₁₀ and NDPM_{2.5} can be considered the net desert dust contribution for the specific area during the specific SDS day. The result of the subtraction of the NDPM₁₀ and NDPM_{2.5} values from the urban PM₁₀ and PM_{2.5} levels, are the APM loads during the dust days (APM₁₀ and APM_{2.5}).
- Once the series of NDPM and APM are obtained, the health effects could be evaluated for PM, NDPM and APM.

Source apportionment with receptor modelling, based on sampling and chemical analysis of PM, is also suggested. However, when there are other important sources of non-desert dust (e.g. local soil or urban dust), this approach may be unable to differentiate sources.

A potential solution is implementing the study at a reference rural/remote site. As the review by Querol et al. (2019a) showed, local pollution in areas far away from dust sources can be enhanced under intense SDS (by thinning of the boundary layer and the interaction of mineral dust and gaseous pollutants) and dust can be co-transported with pollutants and microorganisms such as fungi and spores.

Better monitoring systems can support decision-makers to establish to what extent disease outbreaks are the result of transported sand and dust and to assess the contribution that human activities have made to that process. That is, they can help better comprehend the impact of human activities on SDS and how these ultimately impact the environment and social systems.

4.4.3 Rationale for statement 3 – conducting health impacts research and epidemiological studies in areas affected by SDS

WHO has followed a systematic process to review the effects of desert SDS on human health. This has allowed for summarizing quantitatively, using meta-analysis, the effects of dust on several mortality and morbidity outcomes (Tobias et al., 2019b).

Various epidemiological studies on the health effects of dust events have formulated hypotheses in different ways. They have compared health outcomes between days without and with desert dust events, assessed differences in association between total PM and health on days without and with desert dust events, or looked for independent effects of dust-derived PM and APM on health.

The summary of the evidence of the systematic review on desert dust indicated inconsistent results, depending on the way of assessing the effect of dust on health and the geographical region where the studies were conducted. The comparability of short-term estimates of desert dust health effects obtained in different studies could be improved by standardizing the modelling of desert dust exposure, as proposed by Tobías & Stafoggia (2020). Furthermore, studies on long-term effects of SDS are needed.

4.4.4 Rationale for statement 4 – desertification and wind erosion reduction interventions

There is a recognized pathway that links the presence of green spaces and health benefits (Markevych et al., 2017; Rojas-Rueda et al., 2019). Green spaces play an important role that is under intense scrutiny, from both empirical studies and models, in terms of ecosystem services and co-benefits to improve (mental and physical) health, mitigate climate change and provide spaces for physical activities (Egorov et al., 2016).

Various techniques, mainly reforestation plans, have been implemented in different ways in many countries to reduce exposure to desert dust (FAO, 2009, 2021). Most of these techniques were developed to protect cultivated fields from soil loss (Nordstrom & Hotta, 2004), for carbon sequestration projects and to address desertification. Health impacts have rarely been taken into account in most of the projects (Donovan, 2017). Nevertheless, tree and shrub planting should be taken into account to reduce PM in areas heavily affected by desert dusts following careful studies of the environmental conditions of the land and areas where such plans are going to be implemented.

On an international level, there is well-established agreement that

[t]here is need for an integrated multi-scale approach for effective SDS control. Control measures at the field scale to protect soil and reduce wind speed locally, need to be combined with landscape measures over large areas to reduce wind speed, reduce sand and dust mobilization and increase deposition of sand and dust out of the atmosphere. Measures must simultaneously tackle different components of the landscape, including cropland, rangeland and deserts, as well as other sources, such as building sites, mines, etc. Integrated, landscape level measures are especially critical given the transboundary impacts of SDS.

Control of anthropogenic sources of SDS is synonymous with sustainable land management [...] and integrated landscape management [...] and requires a long-term vision (UNEP, WMO & UNCCD, 2016).

Such initiatives are successful in the long-term only if they carefully consider existing water resources and utilize well-adapted plant species.

It is worth considering that most of the published studies supporting greening interventions have been carried out in North American (e.g. Nowak & Heisler, 2010), European (Selmi et al., 2016) and some Asian cities (e.g. Yang et al., 2005); some research results are available from areas in desert regions (e.g. Cohen, Potchter & Schnell (2014)). Overall, however, there is a lack of systematic studies in cities and in rural areas heavily affected by desert dust. Most of the studies are mainly urban, although the impacts of desert dust are not negligible for populations living in rural areas. It is worth noting that water resource management can represent a more crucial issue than greening in various countries.

4.4.5 Rationale for statement 5 – urban street cleaning

A review of street cleaning as a measure to mitigate the impact of road dust offers indirect evidence of the benefits of this type of intervention (IDAEA, 2013). The authors found that sweeping alone did not decrease PM levels in the short term, although a reduction could not be excluded in the long term. In contrast, washing – alone or in combination with sweeping – yielded more promising findings, with PM₁₀ reductions observed in most reviewed studies. PM₁₀ reductions varied within 7–30% of the daily mean PM₁₀ concentration depending on the local situation, and were observed in a variety of settings, including Asia, Europe and North America.

In addition, street washing and sweeping can be cost-effective in reducing the health impacts of pollution from road traffic, as indicated in an analysis from the United Kingdom (Ballinger et al., 2016).

The practice of street cleaning should be carefully discussed before adoption due to the use of resources and energy that may not produce the expected overall public health benefits. Additionally, there are no studies that provide direct evidence of the effectiveness of street cleaning for reducing desert dust exposure and/or its adverse health effects after intense episodes with high dust deposition rates.

5

Dissemination of the guidelines

These guidelines will be distributed through multiple communication platforms to reach a wide range of audiences. This includes formal communication lines through WHO offices to relevant national authorities, professional organizations and agencies and nongovernmental organizations; more informal local-scale, on-the-ground platforms; and social media using tools to raise awareness and campaigns to foster engagement. In addition, collaboration platforms with other UN agencies, regional bodies and national partners will be used to facilitate dissemination.

It is important to widely distribute and disseminate the information provided in these updated guidelines, and using effective communication to do so will be key to successful uptake. Although these guidelines are universally applicable, additional and/or different approaches and strategies may be required to disseminate and communicate information about them in low- and middle-income countries, particularly where poverty and inequity could add to the complexity of the distribution and communication process.

The communication strategy for the guidelines aims to address all different types of audiences by modifying the language used to present the guidelines and diversifying the tools and channels used to maximize reach and impact to all relevant users. Communication is based on the provision of strong and clear messages and the establishment of relationships with all relevant stakeholders across sectors to ensure the usefulness, acceptability, understanding and uptake of the final product.

5.1 Tools and approaches to raise awareness of the guidelines

Several tools and approaches will be used, including dedicated WHO webpages, communication materials, awareness-raising campaigns, and specific information dissemination and communication approaches.

The WHO website is the major channel for disseminating information on the air quality guidelines to a range of users and for targeting policy-makers, health-care professionals, governmental agencies, the media, academia and the public. The website provides general information on the project and links to relevant documents and resources.

Lay versions, graphical materials, and materials developed in different official languages for promotion and awareness-raising purposes are available on social media platforms. Other means to communicate the guidelines include answers

to frequently asked questions, factsheets and key messages on air pollution and health addressed to policy-makers or health practitioners.

Advocacy and outreach activities in key high-level forums are planned as part of the road map for implementation of this update of the WHO air quality guidelines. A prime example is the joint BreatheLife campaign (led by WHO, the Climate and Clean Air Coalition (CCAC) and UNEP), which aims to mobilize individuals and cities to protect human health and the planet from the adverse effects of air pollution (WHO, CCAC & UNEP, 2018). Another example is the WHO Urban Health Initiative, which promotes the consideration of human health in city development (WHO, 2020c). A specific package to train health-care professionals in air pollution and health will also be launched.

Dissemination of the WHO air quality guidelines is a whole-of-society effort. This means that, while WHO will be targeting several strategic small- and large-scale communication forums, the availability and accessibility of the air quality guidelines will enable their wide distribution among interested parties. This includes civil society organizations, which can further share them through their related initiatives.

In addition, WHO aims to participate in relevant conferences, workshops and stakeholder meetings to introduce the guidelines to audiences globally. These include:

- large, high-profile events with a predominant policy focus;
- smaller workshops or meetings of end-users of the guidelines;
- meetings of professional medical societies;
- events and conferences of the scientific community working on air pollution and health;
- articles, opinions and/or editorials in leading scientific journals;
- meetings, conferences and personal engagements at the local and grass-roots levels, for example at relevant national association events or targeted consultation in affected communities;
- press releases to civil society by local organizations;
- engagement by governments and by WHO regional and country offices; and
- national-level patient groups and networks.

5.2 Risk communication

Effective risk communication enables and empowers people who are facing health risks to make informed decisions that can improve their personal well-being. These people, in turn, can educate others, which can

ultimately empower communities to take actions to reduce risks and increase healthy behaviours. The air quality guidelines provide the evidence base from which successful risk communication about air pollution effects on human health can take place. The provision of air quality guideline (AQG) levels, for instance, aims to prompt action to reduce health risks from exposure to air pollutants. By outlining who is most affected by exposure to air pollution, these guidelines are also able to provide direction in terms of to whom risk communication should be targeted in order to be most successful.

It should be noted that risk communication around air pollution is difficult and many factors need to be considered, including understanding how people perceive risk and ensuring that the risks of poor air quality are communicated in a way that empowers rather than disempowers people. In order to do this effectively, using the WHO air quality guidelines as a base, different stakeholders will need to play a role, including governments and civil society. This highlights the importance of dissemination of the guidelines in forms fit for different audiences, particularly for those who are most impacted by poor air quality. Specific information on the principles of risk communication is available in different WHO publications (WHO Regional Office for Europe, 2013b; WHO, 2017, 2020a).

5.3 Advocacy and engagement of stakeholders

The air quality guidelines advocate for services and regulatory frameworks that promote the management and reduction of air pollution to protect the health and well-being of individuals and communities. The successful dissemination and communication of the air quality guidelines aim to ensure the adoption of the guidelines into relevant institutional, community, national and international policies in order to transform existing systems and processes and, ultimately, improve human health.

Any successful advocacy strategy requires collaborative approaches and the effective engagement of relevant stakeholders across sectors. Participatory approaches are deemed valuable, particularly in lower-income contexts. This is because a consultative dialogue is often more successful at tangibly bringing across abstract concepts to communities, for instance, rather than one-way information sessions.

WHO will use its convening power to facilitate effective cooperation and ensure that key stakeholders (not only from different sectors but also from various perspectives, including local and national governments, civil society and academia) can share and benefit from their respective expertise, experience and resources.

6

Implementation of the guidelines

The WHO air quality guidelines set goals for protecting public health on a worldwide scale. They were established through a rigorous process of revision and evaluation of scientific evidence on the health effects of air pollutants and, like other WHO guidelines, are not legally binding recommendations. National standards are developed through a policy-making process by each country, have legal status and are based on the specific conditions of the country itself. Supranational (e.g. EU) and regional standards may also be developed, depending on the political structure of the area. The establishment of adequate legislation for protection of the population from the health effects of air pollutants is an essential step for all countries. The transfer of guidelines into practicable standards is an integral part of public health and environmental protection policy and is a challenge for most countries. The continuous improvement of air quality requires a formidable effort by those countries dedicated to addressing this major environmental health problem in order to progressively reduce the potential health effects, irrespective of the air pollution level at which they start. Abatement measures and air quality improvement should aim to achieve the interim targets and, finally, the air quality guideline (AQG) levels as expeditiously as possible (additional guidance on interim targets can be found in [section 2.5.3](#)). Up-to-date knowledge and information on levels of air pollution and guidance on interim targets can increase awareness and provide an incentive for the adoption of measures to reduce the level of pollutants, monitor progress and evaluate results.

This chapter examines that process and provides an overview of the general usefulness of the WHO air quality guidelines, with an emphasis on the careful assessment of national needs, capacity-building and the additional elements that are necessary in the development of national standards. Once standards have been established, there is a need for a proper implementation strategy and management of air quality with monitoring, training and enforcement. Health risk assessment is an essential tool to inform public policy decisions by providing an understanding of pollution-related disease burden and the potential for burden reduction. Collaborations of the health-care sector and many different stakeholders are essential to maintain public health protection.

6.1 Significance of the guidelines: an evidence-informed decision support tool

AQG levels are widely seen as a practical instrument for advancing emission reductions and the design of effective measures and policies. WHO guidelines equip policy-makers and other end-users across a range of different needs with the necessary evidence base to inform their decisions. They serve as a reference for assessing whether, and how much, the exposure of a population

(including particular vulnerable and/or susceptible subgroups) is associated with health concerns. For various target audiences and for each stakeholder group, they can function as a critical tool to be used in multiple ways and integrated into their work for years to come.

6.1.1 Use by authorities

Health risk assessments are an important tool for authorities (at international to local levels) when deciding on necessary emission reduction measures because they provide estimates of the health burden/impacts on the population and, therefore, allow a comparison of the consequences among different policy options. These options can include measures to reduce emissions from various sources, measures aimed at reducing concentrations of pollutants in ambient air, measures aimed at reducing exposure of individuals and the population, and/or measures related to urban planning. In principle, the priority should be to prevent emissions of pollutants and reduce them at source.

6.1.2 Use by technical experts and decision-makers

For technical experts and decision-makers, the guidelines are vital in providing information on concentration–response relationships that give insight into the consequences of certain regulations or standards on the associated health effects. They are essential quantitative inputs to quantify the impact of air pollution on health and can be useful at the national and international levels when developing air quality limits or standards as they provide the scientific basis to identify the levels at which air pollution can cause a significant and unacceptable health impact. They provide valuable information used in cost–effectiveness and cost–benefit analyses of various policies and, based on these recommendations, national governments and international organizations can be better informed when introducing air quality standards to ensure the protection of people's health.

6.1.3 Use by civil society, patient and other advocacy groups

They can also be used by civil society, patient and other advocacy groups to raise awareness and encourage actions to protect the population, including susceptible groups such as children, from exposure to air pollution. They can be used to help inform these groups to advocate to policy-makers to improve air quality levels. They are of great value for communicating the health risks and potential cost–effective solutions to reducing air pollution. Organizations responsible for risk communication and general awareness-raising can use these guidelines for promotion campaigns and appropriate risk communication. The guidelines provide scientific evidence on a range of health effects associated with air pollutants and facilitate appropriate risk communication to specific vulnerable and susceptible groups.

Therefore, they need to be promoted broadly to citizens, national and local authorities, and nongovernmental organizations responsible for risk communication.

6.1.4 Use by health/environmental impact assessment practitioners

For health/environmental impact assessment practitioners, these guidelines provide concentration–response relationships that give insight into the expected health effects at observed or expected air pollution levels under various future scenarios. They provide vital input to assist in deriving the health burden or impact of air pollution; in that sense, they can be used when conducting studies to obtain an evaluation of the magnitude of the health problem for a particular situation. The systematic reviews developed in support of these guidelines will support practitioners in raising awareness of the credibility of the issue of air pollution as a public health problem and in applying the recommended concentration–response relationships uniformly so as to justify their use in different countries.

6.1.5 Use by researchers and academics

Researchers and academics will also benefit from the guidelines as they clearly identify critical data gaps that need to be filled in the future through a structured research agenda in order to better protect the population from the harmful effects of air pollution. In addition, the importance of the burden of disease related to air pollution provides an opportunity to justify the inclusion of content related to the guidelines in university curricula for a variety of medical professionals and scientists.

6.2 Assessment of national needs and capacity-building

National needs, including the need for capacity-building, differ greatly among countries. They depend in great part on the existence and level of implementation of national, regional and international policies. In many countries, air pollution is now perceived as a major and growing environmental and public health problem. Nevertheless, significant differences are still evident in multiple areas:

- the existence and operation of air pollution monitoring systems;
- the availability of and public access to data;
- air quality management policies, regulations and standards;
- the availability of trained human resources to understand, assess and monitor health impacts; and
- implementation of universal health coverage and cross-sectoral collaboration.

The existence and operation of air pollution monitoring systems differs by country and city.

Conditions at the country and city levels, specifically for the annual mean PM_{2.5}, have been documented as interactive maps as part of the WHO Global Health Observatory (WHO, 2021a). Progress in combining satellite remote sensing, global chemical transport models, land-use regression models, high-resolution dispersion models and surface measurements (including those made using low-cost sensors) has made information on exposure increasingly available, including in some of the most highly polluted and data-poor regions. However, these estimates need to be grounded and evaluated with existing or new ground-based monitoring; further development of these methods depends to a large extent on the availability of surface measurements in all regions of the world.

The availability of and public access to data to assess population exposure to ambient air pollution and quantify the health impacts or burden related to air pollution for past and current scenarios or future projections also differs by country.

Differences also exist between countries in the development and implementation of air quality management policies, regulations and standards that take into consideration the latest research evidence on the health impacts of ambient air pollutants. Policies to reduce emissions of air pollutants, which are clearly preferable and should be the main focus of any air quality management plan, are highly context dependent: what might be effective and contribute to improving public health in one setting might not work in another. Therefore, understanding the particular situation, including the main emissions, sources and nature of the populations exposed, is critical to the development of effective risk management policies and strategies and is important for decision-making. Most critical is to understand the current level of air pollution in relation to the guidelines.

Lastly, there are differences in the implementation and strengthening of universal health coverage and in the level of cooperation of the health sector in decision-making with other sectors. These include the environment, transport, land planning, housing and energy, agriculture, industrial, and building sectors at the national, regional and, in some cases, international levels.

6.3 Moving from guidelines to air quality standards

The primary aims of these guidelines are to provide a uniform basis for the protection of public health from adverse effects of air pollution and to eliminate or reduce exposure to those pollutants known or likely to be hazardous. Based on the extensive scientific evidence available, the guidelines aim to identify the optimal level of air quality to protect public health in different contexts; they

provide a pathway to countries to transform the recommended AQG levels into legally enforceable standards. This section discusses ways in which this may be done, drawing from and expanding upon previous documents (WHO Regional Office for Europe, 1987, 1998, 2000b), each of which is a useful resource on this topic. The discussion here is limited to pollutants measured in ambient air and does not include the setting of emission standards.

6.3.1 Air quality standards

Air quality standards are the cornerstone of air quality management. Such standards are adopted and enforced by regulatory authorities to define the acceptable level of air pollution for a country or region. They define the level of an air pollutant, such as a concentration measured in ambient air for a specific averaging time. Unlike the case for a guideline value, several additional elements are usually specified in the definition of a standard. These include the averaging time, the measurement technique and strategy, data handling procedures (including quality assurance/quality control), and the statistics (for example, choice of a particular percentile) and form used to derive the value to be compared with the standard. The definition of a standard may also include a permitted number of exceedances of a certain numerical value in a given period.

Air quality standards may be based solely on scientific evidence and public health considerations. However, other features such as legal aspects, cost–benefit or cost–effectiveness may also be examined. In practice, there are generally several opportunities within a legal framework to address economic issues, as well as issues related to technological feasibility, infrastructural measures and sociopolitical considerations. These can be considered during the standard-setting process or when designing appropriate measures to control emissions. This process may result in the establishment of multiple standards, such as an adverse effect-oriented standard as a long-term goal and less stringent interim standards to be achieved within shorter periods of time.

Standards also depend on political choices about which health and environmental effects should be prevented and the extent to which populations should be protected. They also depend on the country's economic development level, capability in air quality management and other factors. Given that the benefits of clean air policies largely outweigh the cost of managing air pollution (Amann et al., 2017), the political choice for the adoption of rigorous standards may find broad societal support for economic reasons. Some countries have separate standards for the protection of public health and for the environment. Moreover, the stringency of a standard can be influenced by provisions designed to account for individuals or populations who might be more susceptible to the effects of

air pollution, such as children, older adults, and individuals with asthma or other pre-existing diseases. Consideration of environmental justice or other equity issues that affect disadvantaged segments of the population may be accounted for when deriving standards. It also might be important to specify whether effects are considered for individual pollutants or for a combined exposure to several pollutants. Air quality standards should be regularly reviewed and revised as new scientific evidence emerges on adverse effects on public health and the environment.

6.3.2 Legal aspects

Within established legal frameworks, and using the WHO air quality guidelines as a starting point, the development of standards involves a consideration of several aspects. These are in part determined by the emission sources, characteristics of populations and physical properties of the environment, and include the following determinations: (i) which pollutants should be regulated; (ii) the adverse health effects against which the population needs to be protected; (iii) which individuals or subpopulations are most at risk for the effects of air pollution; (iv) what level of risk and related costs for society are acceptable to the populations; (v) what uncertainties remain in the evidence base and how they will affect the decision-making process; and (vi) the feasibility of complying with the proposed standards (which includes assessing the costs and benefits of compliance).

Legislation on, as well as the format of, air quality standards varies from country to country but, in general, the following aspects should be considered:

- identification and selection of the pollutants to which the legislative instrument will apply;
- the numerical value of the standards for the various pollutants or the process for making decisions about the appropriate standards, applicable detection methods and monitoring methodology;
- actions to be taken to implement the standard, such as the definition of the time frame needed/allowed for achievement of compliance with the standard, considering emission control measures and necessary abatement strategies; and
- identification of the responsible enforcement authorities.

Depending on their position within a legislative framework, standards may or may not be legally binding. In some countries, the constitution contains provisions regarding the protection of public health and the environment. The development of a legal framework based on constitutional provisions generally comprises two regulatory actions.

The first is the enactment of a formal legal instrument, such as an act, law, ordinance or decree. The second is the development of regulations, by laws, rules and orders.

6.3.3 Factors to be considered in setting standards

The recommendations ([Chapter 3](#)) of these WHO air quality guidelines are based on serious health effects (mortality or hospital admissions/emergency room visits) in a general population and are not designed to focus on the protection of sensitive groups. It is notable that epidemiological studies of the general population include sensitive groups, and these sensitive groups contribute, in part, to the reported risk estimates. Furthermore, such studies often do not provide separate CRFs for various subgroups of the population. However, in setting a standard for the control of an environmental pollutant, consideration may be given to additional aspects, including the adverse effects that the standard will address. A hierarchy of effects on health can be identified, ranging from minor and temporary illnesses to acute, severe illness, chronic disease and death. Distinguishing between adverse and non-adverse effects can pose considerable difficulties (Thurston et al., 2017). Of course, more serious effects are generally accepted as adverse. In considering effects that are either temporary and reversible or involve biochemical or functional changes with uncertain clinical significance, judgements must be made as to which of these less serious effects should be considered adverse. With any definition of adversity, a significant degree of subjectivity and uncertainty remains. Judgements as to adversity may differ between countries because of factors including different cultural backgrounds and different levels of health status.

Susceptible populations or groups are defined here as those who are more sensitive because of impairment by concurrent disease or other physiological limitations and specific characteristics that make the health consequences of exposure more significant (e.g. the developmental phase in children and reduction in the physiological reserve capacity of older people). Other vulnerable groups may also be judged to be at special risk owing to their exposure patterns or to having an increased effective dose for a given exposure (e.g. outdoor workers, athletes). These populations may vary across countries owing to differences in the number of people with inadequate medical care; existence of endemic disease; prevailing genetic factors; or prevalence of debilitating diseases, nutritional deficiencies and lifestyle factors. The setting of air quality standards generally takes into account other considerations beyond public health impacts such as economic and technological aspects and, as such, is considered a political decision.

Another factor to be considered in developing standards is information about the concentration–response relationship for the pollutant of concern. Where adequate evidence is available, concentration–response relationships for a number of pollutants are presented in this update of the WHO air quality guidelines.

In developing standards, regulators should consider the degree of uncertainty about concentration–response relationships. Differences in the population structure (age, health status), climate (temperature and humidity) and geography (altitude, different ecosystems) can have an impact on the prevalence, frequency and severity of effects and may modify the concentration–response relationships provided in these guidelines in their application to a particular population.

Important factors to be considered in developing standards are the number of people who are exposed to concentrations of concern and the distribution of exposure among various population groups at current pollution concentrations and at the different concentrations at which standards might be set. As well as monitoring data, the results of exposure modelling can be used at this stage of a risk assessment. The origin of background air pollution, including long-range pollution transport and its contribution to ambient levels, should also be evaluated when considering standards. It is important that guidelines are health based and, therefore, do not consider background values, whereas standards may include considerations of background levels (e.g. in the case of ozone, background increases with a warming climate).

The extent to which ambient air quality estimates from monitoring networks or models correspond to personal exposure in the population should also be considered in standard setting. This will depend on the pollutant in question (e.g. personal exposure to carbon monoxide is poorly characterized by fixed-site monitors) and other local characteristics, including lifestyle, climatic conditions, spatial distribution of pollution sources and local determinants of pollution dispersion.

Other important exposure-related concerns include how much total human exposure is due to ambient, outdoor sources as opposed to indoor sources, and how to apportion the regulatory burden among the different routes of exposure (e.g. PM from outdoor sources versus PM from household cooking with fossil fuels) for pollutants where multiple routes of exposure are important. These may vary substantially between countries. For example, indoor air pollution levels are normally quite substantial in households in countries where fossil and/or biomass fuels in unvented stoves are used for cooking and heating in homes. However, further discussion of the evolving methods of exposure assessment is beyond the scope of these guidelines.

6.3.4 Risk assessment

Generally, the central question in developing air quality standards to protect public health is the degree of protection associated with the different pollution levels at which standards might be established. In the framework of quantitative risk assessment, various proposals for standards can be considered in health or ecological risk models. These models represent a tool that is increasingly used to inform decision-makers about some of the possible consequences of pollution associated with various options for standards (or, alternatively, the reduction in adverse effects associated with moving from current conditions to a particular standard). Regulatory risk assessments are likely to result in different risk estimates across countries owing to differences in exposure patterns and in the size and characteristics of susceptible and vulnerable populations at special risk.

It is important to recognize that there are many uncertainties at each stage of a regulatory risk assessment. The results of sensitivity and uncertainty analyses should be presented to characterize the impact of major uncertainties on the risk estimates. In addition, the methods used to conduct the risk assessments should be clearly described and the limitations and caveats associated with the analysis should be discussed. In addition, the degree of acceptability of risk may vary between countries because of differences in social norms and the degree of adversity and risk perception among the general population and various stakeholders. How the risks associated with air pollution compare with risks from other pollution sources or human activities may also influence risk acceptability (GBD 2019 Risk Factors Collaborators, 2020).

6.4 Air quality management

Risk to health from inhaled pollutants varies with the concentrations of pollutants inhaled and the mechanisms by which they cause adverse effects, which may be acute or chronic. The sources of exposure to airborne contaminants are myriad, even for the pollutants covered by the WHO air quality guidelines, and pollutants are encountered as people move through multiple environments throughout the day. The microenvironmental model is a comprehensive construct for exposures to inhaled agents and for considering risk reduction through air quality management (National Research Council, 2012). A microenvironment is a place where time is spent and that has a particular pollutant concentration profile during the time spent there; for example, a motor vehicle represents a microenvironment during the time spent commuting. A microenvironment with a high concentration of pollution, such as an urban street canyon, could make a substantial contribution to total exposure, even if only a brief period of time were

spent there. This model is useful for considering how air quality guidelines and standards can reduce personal exposures and for linking air quality management to benefit public health.

This model is also advantageous for considering the numerous microenvironments relevant to air pollution and associated risks to health, and how characteristics of the environment determine exposures. Table 6.1 lists some key microenvironments in urban environments, the pollution sources within these environments and some of the main pollutants present in them. The residence is particularly important because most people spend the majority of their time at home. In urban areas, the air contaminants in the home include those generated by indoor sources, such as cooking and tobacco smoking, and the indoor penetration of outdoor air pollutants, including PM and carbon monoxide generated by local traffic. Streets, which may have hot spots of air pollution generated by traffic or industrial sources, are another key and distinct microenvironment, and one that can be directly benefited by air quality management. The relative significance of different microenvironments across the world varies by where time is spent, the nature of buildings and housing, the distribution of sources and the stringency of measures taken to manage air quality (Samet, 2010).

Table 6.1. Sources of air pollution in urban microenvironments

Microenvironment	Sources	Pollutants
Home	Cooking, space heating, parked vehicles, hobbies, smoking, household products, pets, rodents, insects	PM, CO, NO _x , VOCs, allergens
Transportation environments	Vehicle and industrial emissions, road dust, background pollution, smoking	PM, including ultrafine PM, CO, NO _x , O ₃ , VOCs, aeroallergens, carcinogens
Streets	Vehicle emissions, road dust, background pollution	PM, including ultrafine PM, CO, NO _x , O ₃ , VOCs, carcinogens, lead
Work environments	Industrial processes, smoking, background pollution	PM, CO, VOCs, NO _x , carcinogens
Entertainment environments	Cooking and space heating, background pollution, smoking	PM, VOCs, carcinogens

CO: carbon monoxide; NO_x: nitrogen oxides; O₃: ozone.
Source: reproduced from Samet (2010) with permission from publisher.

The WHO air quality guidelines address air pollution and, hence, cover the many microenvironments where people spend time. At times, the increased breathing rate that results from certain activities may increase the dose of inhaled pollutants at a given concentration. In outdoor environments, there may be high-level exposures, sometimes transient, that may reflect particular industrial sources, traffic hot spots or more general sources, for example wildfires or agricultural burning. Risks for some adverse health effects, such as lung cancer or all-cause mortality, are driven by longer-term and cumulative exposures. Hence, the WHO air quality guidelines include both 24-hour (or even shorter time periods, such as 1 hour for nitrogen dioxide or 10 minutes for sulfur dioxide) and annual averaging times.

In many countries around the world, most time is spent indoors, making indoor microenvironments critical in determining the total exposure to air pollution. Ambient air pollution penetrates indoors, so exposures to pollutants that are covered by the guidelines also occur in homes and other indoor places. Conversely, indoor sources do contribute to outdoor air pollution. An example is the burning of biomass fuels for heating and cooking. The extent of penetration of ambient pollutants into indoor environments varies across pollutants. For PM, the degree of penetration depends on the size distribution of the ambient PM, whereas for gases the reactivity of the pollutant is key (e.g. ozone is highly reactive, which causes concentrations to quickly decay indoors). Also critical are the characteristics of the building, that is, how airtight it is and whether it has an air handling system (and, if so, its characteristics) or an air cleaning system for particles and gases. In higher-income countries, a central air handling system (i.e. a heating, ventilation and air-conditioning system) may be equipped to remove particles.

Modification of time–activity patterns is a widely used governmental and personal strategy to reduce pollution exposure. Air quality indices inform the public when concentrations have reached a level at which health is threatened. Typically, recommendations are tailored to the level reached and the susceptibility of those exposed, for example, people with asthma; avoiding outdoor environments and outdoor exercise is an anchoring strategy. In some locations, particularly those where air pollution is known to reach very high levels, people may use personal protection and air purifiers. These approaches vary in their effectiveness, but neither is a satisfactory alternative to governmental actions to reduce outdoor pollution concentrations.

The development of low-cost monitors for airborne PM allows people to measure one key air pollutant in their specific microenvironments (Lewis, von

Schneidemesser & Peltier, 2018). Although the accuracy of these monitors does not reach the level required for reference monitors used by regulatory agencies, they can provide a useful complement to reports from governmental agencies and can be a valuable resource when central site monitoring of known accuracy is not available. The results can be complementary if aggregated for so-called citizen science purposes, particularly by improving the spatial resolution over that provided by regulatory monitoring networks. People also use the personal monitoring results for guiding their time–activity patterns, particularly those related to time spent outdoors.

Air quality regulation and management include various policy measures to protect population health. Such policy measures need to be informed by previous evidence regarding their efficacy. A specific type of applied research activity, accountability research, assesses whether a certain policy has had an effect on reducing emissions and decreasing concentrations. Such research may also contribute to estimating the burden of disease that might be avoided if certain actions are taken (van Erp et al., 2008).

A proper evaluation of the evidence for effective air quality interventions is under development and a systematic review of the available evidence is accessible from the Cochrane Library (Burns et al., 2019). This document articulates the challenges and limitations of this kind of research. Few existing studies directly examine the effects of these interventions on environmental concentrations of pollutants or the resulting health outcomes. Therefore, the health benefits of interventions must be inferred from the reductions in emissions. In the future, as new policies are introduced, decision-makers should consider a built-in evaluation component, which could facilitate more systematic and comprehensive evaluations.

Specific evidence-informed suggestions for air quality management, according to a hierarchy of interventions, have been proposed (PHE, 2020). In this case, the first priority is preventing, reducing or replacing polluting activities to reduce emissions. The second priority is taking actions to reduce the concentration of air pollution once the polluting activity has occurred and the third is individual avoidance of exposure. The hierarchy for the most effective approaches starts with reducing emissions, followed by reducing concentrations and then reducing exposure. Five areas for potential action have been suggested:

- vehicles and fuels, including for heating
- spatial planning
- industry
- agriculture
- behavioural change.

In addition, high-level interventions have been identified with the potential to benefit health by reducing emissions, concentrations and exposures to the pollutants that cause harm. A report from a WHO consultation in 2019 (WHO, 2020a) provides an overview of the issues related to interventions that are critical for managing air pollution exposure at individual level (e.g. physical activity, use of face masks and air purifiers). A Cochrane review on the topic is also in press; the review protocol has been published (Janjua et al., 2019).

6.5 Methodological guidance for health risk assessment of air pollution

An air pollution health risk assessment estimates the health impact to be expected from measures that affect air quality in different socioeconomic, environmental and policy circumstances. As such, it is an important tool for informing public policy decisions. This section describes in broad terms how the health risks of outdoor air pollution and its sources are estimated and provides an overview of the general principles for the proper conduct of health risk assessment for various scenarios and purposes. This section draws from a previous document (WHO Regional Office for Europe, 2016b) to provide a general understanding of the concepts, scope and principles of health risk assessments.

Health risk assessments aim to estimate the risks of past, current or future exposure to air pollution and of the changes in exposure that may result from planned policies or other modifications of air quality. An air pollution health risk assessment may be quantitative or qualitative; it generally assesses (i) the amount of air pollution present (i.e. pollutant concentrations); (ii) the amount of contact (exposure) of the targeted population; and (iii) how harmful the concentration is to human health (i.e. the resulting health risks to the exposed population). The estimates provided by a health risk assessment are intended to inform the decisions of policy-makers and/or other stakeholders.

As an analytical tool, health risk assessments include a comprehensive assessment of the health impacts of policies, programmes and projects that affect environmental conditions – known as a health impact assessment. Health risk assessments and health impact assessments are different concepts, although the two terms are sometimes used interchangeably. A health impact assessment, which is an extension of the overall risk assessment, is often characterized by a combination of procedures, methods and tools used to judge the effects that a policy, programme or project may have on the health of a population and on the distribution of those effects within the population; it may also identify appropriate actions to manage those effects.

The main purpose of a health risk assessment is to answer policy questions about the likely health impacts of planned policies or modifications of those policies.

Air pollution health risk assessments are often used to answer the following policy questions.

- What is the public health burden associated with current levels of air pollution?
- What are the human health benefits associated with changing an air quality policy or applying a more stringent air quality standard?
- What are the human health impacts of emissions from specific sources or selected economic sectors, and what are the benefits of policies related to these?
- What are the human health impacts of current policy or implemented actions?
- What are the policy implications of the uncertainties of the assessment?

The first step in a health risk assessment is planning. This includes the definition of the policy question to be evaluated, determination of the availability of data and resources, and selection of appropriate methods and tools. Sources of data required for the health risk assessment include, but are not limited to, the level of air pollution, the exposed population and the health effect, and the relationship of risk to exposure (e.g. CRF). During the planning process, selection of the methods to be implemented may depend on data availability or may determine the data requirements. In addition, the identification of different tools that will be useful in the health risk assessment occurs in the planning step.

Estimating population exposure to air pollutants is the next step in the health risk assessment. Data on population exposure to air pollutants generally come from monitoring by local or national institutions. Estimates of population exposure based on measured air pollution data are often limited by the restricted geographical and time coverage of the data. Recently, predicted estimates of pollutant concentrations from statistical models have become more common and can be used to estimate exposure in locations that do not have air quality monitors. Progress in combining satellite remote sensing, global chemical transport modelling, land-use regression models and high-resolution local dispersion models in combination with existing ground-based monitoring has made information on key air pollutant indicators increasingly available, including in some of the most highly polluted and data-poor regions. It may be difficult to harmonize data from different locations, since measurements and model predictions are often made using different procedures and techniques.

When estimating the change in population exposure caused by a hypothetical change in emissions or pollutant concentrations, monitoring data may be used as a baseline level. However, air quality modelling is needed to estimate future concentration changes resulting from policies and technological innovations.

The next step in the health risk assessment is estimating the health risk. To provide useful advice aimed at answering a specific question, a specific health end-point or set of health end-points in a specific population must be identified. The health risk assessment is unlikely to cover the full range of possible adverse health effects in all possible groups of the population but may focus on those health effects that affect the most people or the most susceptible populations. The quantitative risk of air pollution to health in a population is usually represented by a CRF, which is typically based on a risk estimate from epidemiological studies.

Quantifying the health impact is the next step in the health risk assessment. Health risk assessments often report results in terms of the number of attributable deaths or cases of disease, years of life lost or disability adjusted life-years, or to the change in life expectancy attributable to the total exposure to air pollution or to a change in exposure. These metrics aggregate different types of health impact and can be used to highlight different aspects of the health status of a population. It is important to note that these metrics provide expected values for a whole population and cannot be applied to individuals in that population. Tools for health risk assessment calculation are widely available from WHO (AirQ+) or other sources (such as the US EPA BenMAP-CE) (Sacks et al., 2020).

In summary, an air pollution health risk assessment can quantify the health impact of air pollution or of changes in air pollution resulting from different socioeconomic, environmental or policy circumstances. In many countries, health risk assessments are formally required as part of the decision-making process for new programmes, projects, regulations and policies that may affect air quality. Those conducting a health risk assessment need to understand how to do it; know what data are available and needed, and where to find them; and know how to communicate the results. It is a challenging, yet important, task to find a balance between the complexity of information and tools used and the need to produce understandable results for policy-makers and others who do not necessarily have a technical background or expertise in the field.

6.6 Role of the health sector

Health-care professionals are now regularly faced with questions and concerns from patients about the impact that air pollution can have on their health. This holds particularly true for individuals who suffer from chronic conditions, such as asthma, COPD, diabetes, heart failure and IHD. Parents with young children also often have concerns. However, many health-care professionals working in different disease areas and settings are unable or unprepared to advise.

Engagement of the health community as trusted, connected and committed advocates is crucial. The health sector has a role in:

- raising awareness of the impact of air quality on health using evidence provided by the WHO air quality guidelines;
- advising the public and patients about how the impact of air pollutants above WHO air quality guidelines can be mitigated at an individual level; and
- joining advocacy efforts at the national and international levels to ensure that the health arguments for the WHO air quality guidelines are heard in national policy discussions.

Scientific evidence on the impact of air pollution on health is developing rapidly, and these new guidelines provide AQG levels for different pollutants based on a review of the latest evidence. However, the practical implications for patients and the public, specifically in relation to acute air pollution episodes and the impact on chronic conditions, are unclear to many in the health sector. For this reason, in addition to publishing the guidelines, further efforts are needed to promote the understanding, support and engagement of those in the health sector.

For the WHO air quality guidelines to have a significant impact on the lives of people most vulnerable and susceptible to the effects of air pollution, cooperation with professional societies is crucial to raise awareness of and strengthen the messages related to air pollution, as well as to ensure appropriate education and training for health-care workers. Examples include presenting the AQG levels and what they mean for health in a practical and easy-to-understand format, and providing guidance on what actions individuals can take to reduce exposure when the AQG levels are exceeded. Explaining the risk from air pollution to an individual in relation to other risk factors, such as smoking, is also important. There is a clear role for organizations such as medical societies and patient organizations to work with WHO to communicate the WHO air quality guidelines in the most accessible manner and tailored to the needs of different target groups.

6.7 Intersectoral and multistakeholder cooperation

In addition to the increased role that the health sector should play, intersectoral and multistakeholder action is crucial for the successful development and implementation of air quality policies, including achievement of the goals and targets of the 2030 Agenda for Sustainable Development (PHAC & WHO, 2008; WHO Regional Office for Europe, 2018). In many countries, responsibilities for air quality are shared among government institutions, but collaboration is not always optimal. Since air quality is influenced by policies formulated in diverse sectors, whole-system approaches are needed for protecting the public's health.

Key to effective air quality policy is the adoption of a whole-of-government approach. This approach involves downstream and upstream coordination among governance domains and levels, as well as horizontal cooperation across sectors, supported by the appropriate selection of interventions, financing mechanisms and legal instruments (WHO Regional Office for Europe, 2018). Specific models have been available at national level since the 1990s, such as the national environment health action plans (WHO Regional Office for Europe, 1999). An example of this model is the National Air Quality Cooperation Programme in the Netherlands, which fosters cooperation among different levels of government through consensus, legislation and public participation (Joint Task Force on the Health Aspects of Air Pollution, 2018).

In a similar vein, the Health in All Policies approach can help ensure that the health impacts of air pollution are considered in formulating policy outside the health sector (WHO, 2014d). For example, the California Health in All Policies Task Force convened a multisectoral working group to deal with the issues of transit-oriented development, including its impact on air pollution, active transportation and social cohesion (Government of South Australia & WHO, 2017). Among low- and middle-income countries, Thailand provides an example of promoting the Health in All Policies approach. In 2012 Thailand's National Health Assembly brought together all parties and sectors to exchange knowledge and formulate policy proposals on biomass burning from power plants and from forest fires related to agriculture (Government of South Australia & WHO, 2017; Rajan et al., 2017; NHCO, 2019).

Of particular importance is the exchange of knowledge and experiences, not only between government and the scientific community but also through engaging the private sector, civil society, communities and citizens. An inclusive, multistakeholder approach also contributes to building trust and legitimacy in the policy process, and results in more equitable and context-specific policies (WHO Regional Office for Europe, 2018). Moreover, civil society is a key player in raising awareness and promoting action to tackle air pollution challenges in many

parts of the world. The private sector, in turn, has an important role in delivering context-relevant technological solutions and services. Therefore, government authorities can nurture a favourable environment by building capacity, promoting partnerships and aligning incentives (Joint Task Force on the Health Aspects of Air Pollution, 2018; Chatterton et al., 2017; CCAC & UNEP, 2019).

To control air pollution regionally, policy instruments are in place to facilitate dialogue, cooperation, and exchange of information and experiences among countries. These include, for example, the United Nations Economic Commission for Europe (UNECE) Convention on Long-Range Transboundary Air Pollution, the Malé Declaration on Control and Prevention of Air Pollution and Its Likely Transboundary Effects for South Asia, the Acid Deposition Monitoring Network in East Asia, the Association of Southeast Asian Nations' Agreement on Transboundary Haze Pollution, and the Eastern Africa Regional Framework Agreement on Air Pollution (CCAC & UNEP, 2019; UNECE, 2011). In particular, the Joint Task Force on the Health Aspects of Air Pollution, established within the UNECE Convention on Long-range Transboundary Air Pollution, is a well-established intersectoral platform for working on air pollution and health and for helping define priorities for action (WHO Regional Office for Europe, 2021b).

On the other hand, the 2030 Agenda for Sustainable Development offers a framework to combat air pollution at global level. Within the framework, connections can be identified between approximately 10 of the SDGs and air pollution, including implicit links at target level. SDG 17 (Partnerships for the Goals) offers targets for intersectoral, multilevel and multistakeholder collaboration to address air pollution that are aligned with the Paris Agreement on climate change (Longhurst et al., 2018).

7

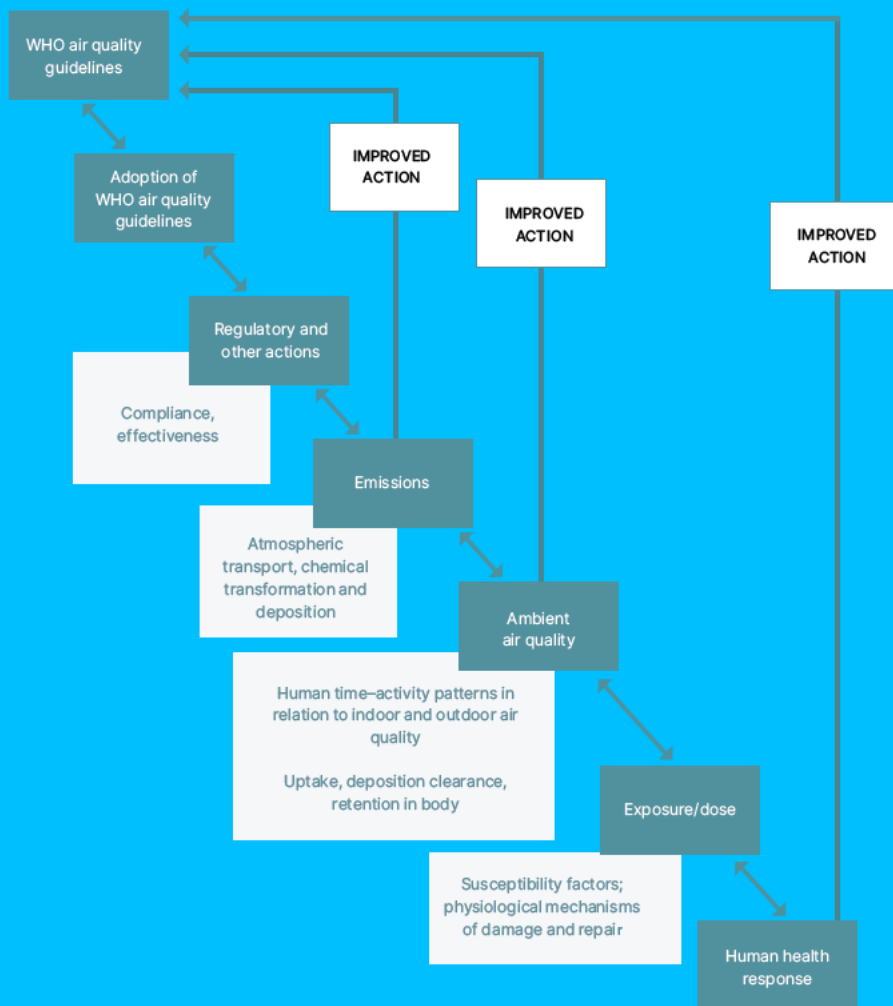
Monitoring and evaluation of the guidelines

The WHO air quality guidelines have the overall purpose of reducing the burden of disease attributable to air pollution globally, aligned with the targets set in the SDGs (UN, 2015) that offer a useful framework for considering gains made in terms of burden reduction. Targets for the SDGs have been set to ensure healthy lives and promote well-being at all ages and to make cities and human settlements inclusive, safe, resilient and sustainable. The WHO air quality guidelines are expected to effectively guide countries towards improving air quality, resulting in a beneficial impact on health risks, and moving closer to meeting several of the SDGs. Monitoring and evaluation of the consequences of implementing the updated guidelines will be key to ensuring their impacts on the reduction of disease burden from air pollution, specifically by:

- evaluating the transfer of the recommendations into local, national, regional and/or international legislation, action plans and other management actions;
- monitoring the achievement of SDG indicators that are directly affected by the recommendations;
- evaluating newly developed or revised air quality standards and other air quality management policies related to ambient air quality that are implemented in countries following publication of the guidelines, in order to determine whether WHO recommendations were used as the basis for their development; and
- surveying different stakeholders to evaluate the quality and usefulness of the guidelines.

An implicit sequence of steps to achieve health targets (such as SDG targets), summarized in [Fig. 7.1](#), follows from promulgation of a guideline or standard. Taking the actual use of the guidelines by national or other regulatory authorities as a starting point, there is a sequence of steps to achieve health benefits, some of which can be tracked (HEI Accountability Working Group, 2003). In considering monitoring and evaluation, the length of time from any action to its health benefits also needs to be acknowledged. This could be a multiyear sequence, particularly for those countries lacking air quality standards and guidelines from the start.

Fig. 7.1. Chain of events within the air pollution accountability framework^a



* HEI defines the air pollution accountability framework as a chain of events that includes the regulation of interest, air quality, exposure/dose and health outcomes, and suggests that accountability research should address the impacts of each of these linkages. Each box represents a link between regulatory action and the human health response to air pollution. Arrows connecting the links indicate possible directions of influence. Text below the arrows identifies general indices of accountability at that stage. At several stages, the knowledge gained from accountability assessments can provide valuable feedback for improving regulatory or other action.

Source: reproduced from the HEI Accountability Working Group (2003), with the permission of the publisher.

7.1 Tracking the implementation of the guidelines

As indicated in [Fig. 7.1](#), the starting point is the actual adoption of the air quality guideline (AQG) levels or interim targets. At this stage of the process, there are steps that can be monitored in a systematic manner. For example, Kutlar Joss et al. (2017) developed a potentially replicable methodology for determining what standards are in place throughout the world. This approach can be followed in maintaining the ongoing tracking of utilization of the WHO air quality guidelines in practice. With the introduction of these updated guidelines, ways to track their dissemination and implementation in countries should be put in place. As a next step, governmental actions need to be taken to incorporate the updated AQG levels or interim targets into regulations or other actions that impact air pollution sources. Such actions can also be tracked by establishing a database (that is periodically updated), as one potential model, which is illustrated in the *WHO report on the global tobacco epidemic 2019* (WHO, 2019b). Based on data compiled by Swiss TPH (Kutlar Joss et al., 2017), WHO developed an interactive tool that provides a snapshot of national air quality standards for classical pollutants for various averaging times. Presented as a map, the tool uses the WHO air quality guidelines and interim targets as references and will be updated regularly (WHO, 2021b).

7.2 Assessing population exposure to ambient pollution

The availability of appropriate population exposure monitoring is critical, as illustrated in [Fig. 7.1](#). Measurement of air pollutant concentrations at fixed-location monitoring sites is the long-standing approach used for air quality management, trend assessment and exposure estimation for epidemiological analyses. However, there is still a lack of air pollution monitoring and inadequate numbers of monitors in rural areas and locations other than major cities in many countries. Thus, monitoring metrics could be the extent of monitoring and the implementation of monitoring to cover gaps. New modelling approaches incorporating satellite and other data may also be useful. In recent decades, in addition to existing air pollution monitoring networks, advanced methods of exposure assessment have become available with the use of satellite observations and various modelling tools to support epidemiological studies, as well as health impact and risk assessment.

Global air pollution concentrations and trends and related estimates of population exposure on priority air pollutant indicators have been compiled in the WHO Global Ambient Air Quality Database, as described in [section 1.3.1](#). Additionally, this update of the WHO air quality guidelines has identified a number of advances in the global development of air pollution monitoring protocols and exposure assessment methods that can be adopted to increase result comparability across studies.

7.3 Health benefits from implementation of the guidelines

The WHO air quality guidelines have the overall purpose of benefiting the health of populations worldwide. The health benefits of the updated WHO air quality guidelines will be realized through reducing population exposures to ambient air pollution via several steps (see [Fig. 7.1](#)). Disease burden reflects both the underlying health of populations and the exposures received. Scientific evidence evaluated during the development of this update shows that health risks attributable to air pollution are large and increasing, particularly due to the increases in air pollution exposure in low- and middle-income countries and to ageing of the world population. Major health benefits are expected to be achieved when ambient air pollution levels are reduced widely, following implementation of the guidelines at a global scale. The databases described in [section 1.3.1](#) can be used to inform global estimates of disease and economic burden, and the ongoing estimates of disease burden made by WHO and sister UN agencies within the framework of the SDGs and by the research community will also be useful for tracking progress.

Furthermore, as summarized in [Chapter 3](#), the updated AQG levels and interim targets are derived with improved global CRFs and provide a set of health and exposure indicators for evidence-informed benchmarking of the health impacts of air pollution. These indicators are consistent with SDG targets and can be monitored and evaluated throughout the implementation of the WHO air quality guidelines within and across countries. By adopting the updated guidelines, progress towards achieving the SDG targets can be explicitly monitored and assessed. In particular, this is the case for indicator 3.9.1 on the mortality rate attributed to ambient air pollution and indicator 11.6.2 on the annual mean levels of fine PM, for which WHO is a custodian agency (discussed in [section 1.3.7](#)). Such measurements will assist stakeholders to assess their progress in the reduction of disease burden caused by implementation of the WHO air quality guidelines, which will likely result, in parallel, in a further reduction of air pollution.

Countries will need to incorporate the multistep process of air quality management at national level, and stakeholders could be directly and periodically surveyed to evaluate the quality and usefulness of the guidelines towards the goal of reducing disease burden and meeting the applicable SDG targets. Sustained progress in improving air quality is the goal of implementation of the guidelines; monitoring of the guidelines impact on reducing disease burden can provide a strong rationale for potential future updates of the guidelines.



Future research needs

There is extensive evidence, which was reviewed to support this update of the WHO air quality guidelines, demonstrating the health effects of exposure to major air pollutants. Evidence from toxicology and epidemiology is sufficient to justify actions to reduce population exposure. Nevertheless, uncertainties and knowledge gaps remain, and future research is needed to reduce these. Suggestions for future research that may help in this regard are listed below. These include further strengthening the policy-relevant scientific base and evidence to support decision-making worldwide, especially in low- and middle-income countries.

- **Set priorities for policy-relevant scientific questions: how, why and for whom do the health effects of air pollution exist?**
 - Assess the shape of the exposure–response relationships at both low and high air pollution concentration levels – the former are now being observed in parts of Europe, North America and Oceania, and the latter are now being observed in parts of Asia and the Eastern Mediterranean Region.
 - Study the toxicity of different sources of air pollution (e.g. tailpipe and non-tailpipe emissions, aviation and shipping emissions, specific industrial sources, wood smoke and desert dust). This includes research into the health effects of technology-driven changes in areas such as primary energy production, where mixtures of coal and biomass replace coal in places.
 - Study the health effects of particle size fractions for which there are limited data.
 - Define sensitive subgroups of the population that need to be protected (e.g. related to socioeconomic status, nutrition, pregnancy, critical windows of development, and young older age) due to the risk of immediate, delayed or lifetime effects.
 - Study multipollutant exposures to determine the relative importance of specific air pollutants (such as nitrogen dioxide, carbon monoxide) and components of PM, with an examination of additive, synergistic or antagonistic effects, including in the presence of pollens or other airborne allergens. This is an area where mechanistic research will likely play an important role.
 - Study the interaction with other environmental and behavioural factors such as traffic noise, green space and allergen exposure; physical activity and diet; and high and low temperatures and other climatic conditions.

- Undertake research into a broader range of health end-points, as the list of organ systems and conditions possibly affected by air pollution is steadily increasing.
 - Study the neurological effects, including the effects on brain morphology in young children and older people, on child development, and on cognitive decline and reduced ability to perform activities of daily life in older people.
 - Study the cardiometabolic effects – emerging evidence links diabetes to air pollution exposure (Yang B-Y et al., 2020), an association in clear need of further corroboration and characterization.
 - Study the effects on various cancer forms (excluding lung cancer, for which a relationship with air pollution has been established).
 - Study the short-term effects of exposure leading to worsening of symptoms for diseases such as allergic, cardiovascular and respiratory conditions and indicated by a wider set of (also subclinical) health status indicators, such as lung function tests or biomarkers.
- Improve the methodology in exposure assessment, study design and evidence synthesis and evaluation.
 - Study exposure assessment – inform this by integrating data from multiple sources (e.g. from large numbers of low-cost sensors) and data fusion (satellite observations, emission sources, dispersion models and ground-based monitoring).
 - Assess multiple sources of exposure in different locations (including home indoor, work indoor and transportation) and time–activity patterns.
 - Assess multiple sources of exposure in populations from different regions, living in different climates, of different socioeconomic status, etc.
 - Improve statistical methods for use in epidemiological studies, such as methods to correct for exposure measurement error in health analyses, multipollutant modelling approaches and methods to correct for confounding.
 - Expand the framework of causal inference by incorporating different study and analysis designs, including novel approaches in epidemiology such as the use of propensity scores, instrumental variables, difference-in-difference analyses and regression discontinuity.
 - Improve methodological aspects related to the evaluation of the quality of individual studies and the synthesis and overall evaluation of the scientific evidence, including determination of the certainty of the body of evidence (e.g. GRADE or other approaches).
- Undertake research into mechanisms of health effects.
 - Study the biological mechanisms explaining epidemiological associations

with all-cause and respiratory mortality of (mixtures represented by) nitrogen dioxide and ozone, especially at low concentration levels.

- Study the mechanisms of effects of (mixtures represented by) nitrogen dioxide and ozone on the cardiovascular system.
- Study the effects of mixtures containing particles of different sizes as well as gaseous pollutants to understand the underlying pathophysiology due to surface interactions between pollutants and molecular or cellular structures (e.g. proteins, lipids, DNA and RNA).
- Continue to develop burden and health impact assessment.
 - Improve methods and input data for health risk assessments, which play a key role in identifying the overall and relative importance of air pollution and its sources for population health. They provide the foundation for identifying priorities and tracking the effectiveness of solutions.
 - Improve the apportionment of population exposure to specific sources or source categories to enable source-specific health risk assessment at the local, national and regional levels.
 - Establish solid mechanisms for the regular review of evidence related to the quantification of CRFs and health burden assessments, including the integrated assessment of burdens from complex mixtures.
 - Integrate air-pollution-related health risk assessment into a comprehensive health impact assessment of actions focused on other determinants of health (such as physical activity, diet and climate).
- Improve assessment of the effectiveness of interventions (accountability research).
 - Evaluate key long-term interventions in all parts of the world, for example local traffic interventions, interventions to reduce emissions from industrial sources, changes in energy use (gas vs electricity), efforts to reduce exposure for at-risk communities and reductions in biomass burning.
 - Evaluate key short-term community (e.g. school closures) and individual (e.g. face masks) interventions during acute episodes, including studies of population exposure, health effects, and societal and economic implications. Evaluation should include conditions critical for successful intervention, for example, sensitivity to socioeconomic conditions; methods of communication; use of adequate exposure indicators; and target group knowledge, attitude and engagement.
 - Develop study methods to assess the effectiveness of interventions and which can provide direct evidence for the attribution of changes in air quality and health to an air quality improvement intervention, as well as to integrate (climate) related co-benefits and dis-benefits.

9

Updating the guidelines

The number of studies of air quality and health has significantly increased since *Global update 2005*, including new studies published after the completion of the systematic reviews conducted for this update. Taken together, the guidelines were informed by a wealth of epidemiological studies that shed light on the risks of exposure to air pollution at both the lower and upper bounds of the concentration–response relationships for the classical air pollutants, including the shapes of such relationships.

WHO will continue monitoring scientific progress in the field to assess the need for future updates. This activity will be facilitated by the Global Air Pollution and Health – Technical Advisory Group, which was established in 2021 (WHO, 2020d), and by annual meetings of the Joint Task Force on the Health Aspects of Air Pollution, established in 1998, within the UNECE Convention on Long-range Transboundary Air Pollution (WHO Regional Office for Europe, 2021b).

Moreover, participation in scientific meetings, follow-up on emerging issues, and close interaction with thematic/technical experts and stakeholders will continue so as to keep abreast of the scientific progress and gauge the need for updating the guidelines. In general, however, the recommendations made in these guidelines are expected to remain valid for a period of up to 10 years.

References

42 U.S.C. § 7409 – US Code – Unannotated Title 42. The Public Health and Welfare § 7409. National primary and secondary ambient air quality standards.

Abbey DE, Nishino N, McDonnell WF, Burchette RJ, Knutsen SF, Beeson WL et al. (1999). Long-term inhalable particles and other air pollutants related to mortality in nonsmokers. *Am J Respir Crit Care Med*. 159(2):373–82. doi: 10.1164/ajrccm.159.2.9806020.

Achakulwisut P, Brauer M, Hystad P, Anenberg SC (2019). Global, national, and urban burdens of paediatric asthma incidence attributable to ambient NO₂ pollution: estimates from global datasets. *Lancet Planet Health*. 3(4):e166–78. doi: 10.1016/S2542-5196(19)30046-4.

ACTRIS (2020). Standard operating procedures and measurement guidelines for ACTRIS in situ aerosol particle variables. Kjeller: ACTRIS Data Centre (<https://actris.nilu.noContent/?pageid=13d5615569b04814a6483f13bea96986>, accessed 7 August 2020).

Akhlaq M, Sheltami TR, Mouftah HT (2012). A review of techniques and technologies for sand and dust storm detection. *Rev Environ Sci Biotechnol*. 11:305–22. doi: <https://doi.org/10.1007/s11157-012-9282-y>.

Amann M, Holland M, Maas R, Vandyck T, Saveyn B (2017). Costs, benefits and economic impacts of the EU clean air strategy and their implications on innovation and competitiveness. Laxenburg: International Institute for Applied Systems Analysis (https://ec.europa.eu/environment/air/pdf/clean_air_outlook_economic_impact_report.pdf, accessed 2 December 2020).

ANSES (2019). Particulate matter in ambient air. Health effects according to components, sources and particle size. Impact on air pollution of the technologies and composition of the motor vehicle fleet operating in France. Paris: Agence nationale de sécurité sanitaire de l'alimentation, de l'environnement et du travail [French Agency for Food, Environmental and Occupational Health & Safety] (<https://www.anses.fr/fr/system/files/AIR2014SA0156RaEN.pdf>, accessed 7 August 2020).

Argyropoulos CD, Hassan H, Kumar P, Kakosimos KE (2020). Measurements and modelling of particulate matter building ingress during a severe dust storm event. *Build Environ*. 167:106441. doi: <https://doi.org/10.1016/j.buildenv.2019.106441>.

Ashley WS, Strader S, Dziubla DC, Haberlie A (2015). Driving blind: weather-related vision hazards and fatal motor vehicle crashes. *Bull Am Meteorol Soc*. 96(5):755–78. doi: <https://doi.org/10.1175/BAMS-D-14-00026.1>

Badaloni C, Cesaroni G, Cerza F, Davoli M, Brunekreef B, Forastiere F (2017). Effects of long-term exposure to particulate matter and metal components on mortality in the Rome longitudinal study. *Environ Int*. 109:146–54. doi: 10.1016/j.envint.2017.09.005.

Baddock MC, Strong CL, Murray PS, McTainsh GH (2013). Aeolian dust as a transport hazard. *Atmos Environ*. 71:7–14. doi: 10.1016/j.atmosenv.2013.01.042.

Ballinger A, Chowdhury T, Sherrington C, Cole G (2016). Air pollution: economic analysis. Main report. Bristol: Eunomia Research & Consulting (<https://www.nice.org.uk/guidance/ng70/evidence/economic-report-pdf-4595574493>, accessed 8 August 2020).

Beelen R, Hoek G, van den Brandt PA, Goldbohm RA, Fischer P, Schouten LJ et al. (2008). Long-term effects of traffic-related air pollution on mortality in a Dutch cohort (NLCS-AIR study). *Environ Health Perspect.* 116(2):196–202. doi: 10.1289/ehp.10767.

Beelen R, Raaschou-Nielsen O, Stafoggia M, Andersen ZJ, Weinmayr G, Hoffmann B et al. (2014). Effects of long-term exposure to air pollution on natural-cause mortality: an analysis of 22 European cohorts within the multicentre ESCAPE project. *Lancet.* 383(9919):785–95. doi: 10.1016/S0140-6736(13)62158-3.

Bell ML, Peng RD, Dominici F, Samet JM (2009). Emergency hospital admissions for cardiovascular diseases and ambient levels of carbon monoxide: results for 126 United States urban counties, 1999–2005. *Circulation.* 120(11):949–55. doi: 10.1161/CIRCULATIONAHA.109.851113.

Benmarhnia T, Rey L, Cartier Y, Clary CM, Deguen S, Brousselle A (2014). Addressing equity in interventions to reduce air pollution in urban areas: a systematic review. *Int J Public Health.* 59(6):933–44. doi: 10.1007/s00038-014-0608-0.

Bentayeb M, Wagner V, Stempfelet M, Zins M, Goldberg M, Pascal M et al. (2015). Association between long-term exposure to air pollution and mortality in France: a 25-year follow-up study. *Environ Int.* 85:5–14. doi: 10.1016/j.envint.2015.08.006.

Bice K, Eil A, Habib B, Heijmans P, Kopp R, Nogues J et al. (2009). Black carbon: a review and policy recommendations. Princeton (NJ): Woodrow Wilson School of Public & International Affairs, Princeton University.

Blair A, Burg J, Foran J, Gibb H, Greenland S, Morris R et al. (1995). Guidelines for application of meta-analysis in environmental epidemiology. ISLI Risk Science Institute. *Regul Toxicol Pharmacol.* 22(2):189–97. doi: 10.1006/rtph.1995.1084.

Bond TC, Doherty SJ, Fahey DW, Forster PM, Berntsen T, DeAngelo BJ et al. (2013). Bounding the role of black carbon in the climate system: a scientific assessment. *J Geophys Res Atmos.* 118(11):5380–552. doi: <https://doi.org/10.1002/jgrd.50171>.

Boogaard H, van Erp AM, Walker KD, Shaikh R (2017). Accountability studies on air pollution and health: the HEI experience. *Curr Environ Health Rep.* 4(4):514–22. doi: 10.1007/s40572-017-0161-0.

Bowe B, Xie Y, Li T, Yan Y, Xian H, Al-Aly Z (2018). The 2016 global and national burden of diabetes mellitus attributable to PM_{2.5} air pollution. *Lancet Planet Health.* 2(7):e301–12. doi: 10.1016/S2542-5196(18)30140-2.

Brauer M, Amann M, Burnett RT, Cohen A, Dentener F, Ezzati M et al. (2012). Exposure assessment for estimation of the global burden of disease attributable to outdoor air pollution. *Environ Sci Technol.* 46(2):652–60. doi: 10.1021/es2025752.

Brauer M, Freedman G, Frostad J, van Donkelaar A, Martin RV, Dentener F et al. (2016). Ambient air pollution exposure estimation for the global burden of disease 2013. *Environ Sci Technol.* 50(1):79–88. doi: 10.1021/acs.est.5b03709.

Brauer M, Brook JR, Christidis T, Chu Y, Crouse DL, Erickson A et al. (2019). Mortality-Air Pollution Associations in Low-Exposure Environments (MAPLE): phase 1. *Res Rep Health Eff Inst.* (203):1–87. PMID: 31909580.

Brown RJC, Beccaceci S, Butterfield DM, Quincey PG, Harris PM, Maggos T et al. (2017). Standardisation of a European measurement method for organic carbon and elemental carbon in ambient air: results of the field trial campaign and the determination of a measurement uncertainty and working range. *Environ Sci Process Impacts.* 19(10):1249–59. doi: 10.1039/c7em00261k.

Bruneekreef B, Beelen R, Hoek G, Schouten L, Bausch-Goldbohm S, Fischer P et al. (2009). Effects of long-term exposure to traffic-related air pollution on respiratory and cardiovascular mortality in the Netherlands: the NLCS-AIR study. *Res Rep Health Eff Inst.* (139):5–71. PMID: 19554969.

Burnett RT, Pope CA III, Ezzati M, Olives C, Lim SS, Mehta S et al. (2014). An integrated risk function for estimating the global burden of disease attributable to ambient fine particulate matter exposure. *Environ Health Perspect.* 122(4):397–403. doi: 10.1289/ehp.1307049.

Burnett R, Chen H, Szyszkowicz M, Fann N, Hubbell B, Pope CA III et al. (2018). Global estimates of mortality associated with long-term exposure to outdoor fine particulate matter. *Proc Natl Acad Sci U S A.* 115(38m):9592–7. doi: 10.1073/pnas.1803222115. License: CC BY-NC-ND.

Burns J, Boogaard H, Polus S, Pfadenhauer LM, Rohwer AC, van Erp AM et al. (2019). Interventions to reduce ambient particulate matter air pollution and their effect on health. *Cochrane Database Syst Rev.* 5(5):CD010919. doi: 10.1002/14651858.CD010919.pub2.

Butterfield D, Beccaceci S, Quincey P, Sweeney B, Lilley A, Bradshaw C et al. (2016). 2015 Annual Report for the UK Black Carbon Network. National Physical Laboratory (NPL Report ENV 7; <http://eprintspublications.npl.co.uk/7276/1/ENV7.pdf>, accessed 17 August 2020).

Cakmak S, Hebbern C, Pinault L, Lavigne E, Vanos J, Crouse DL et al. (2018). Associations between long-term PM_{2.5} and ozone exposure and mortality in the Canadian Census Health and Environment Cohort (CANCHEC), by spatial synoptic classification zone. *Environ Int.* 111:200–11. doi: 10.1016/j.envint.2017.11.030.

Carey IM, Atkinson RW, Kent AJ, van Staa T, Cook DG, Anderson HR (2013). Mortality associations with long-term exposure to outdoor air pollution in a national English cohort. *Am J Respir Crit Care Med.* 187(11):1226–33. doi: 10.1164/rccm.201210-1758OC.

Castro A, Götschi T, Achermann B, Baltensperger U, Buchmann B, Dietrich DF et al. (2020). Comparing the lung cancer burden of ambient particulate matter using scenarios of air quality standards versus acceptable risk levels. *Int J Public Health.* 65(2):139–48. doi: 10.1007/s00038-019-01324-y.

CCAC, UNEP (2019). Air pollution in Asia and the Pacific: science-based solutions. Nairobi: United Nations Environment Programme (<https://ccacoalition.org/en/resources/air-pollution-asia-and-pacific-science-based-solutions-summary-full-report>, accessed 9 December 2020).

CEN (2017). Ambient air: measurement of elemental carbon (EC) and organic carbon (OC) collected on filters. Brussels: Comité Européen de Normalisation [European Committee for Standardization] (BS EN 16909:2017).

Cesaroni G, Badaloni C, Gariazzo C, Stafoggia M, Sozzi R, Davoli M et al. (2013). Long-term exposure to urban air pollution and mortality in a cohort of more than a million adults in Rome. *Environ Health Perspect.* 121(3):324–31. doi: 10.1289/ehp.1205862.

Chang K-L, Cooper O, West JJ, Serre M, Schultz M, Lin M et al. (2019). A new method (M 3 Fusion-v1) for combining observations and multiple model output for an improved estimate of the global surface ozone distribution. *Geosci Model Dev.* 12(3):955–78. doi: <https://doi.org/10.5194/gmd-12-955-2019>.

Chatterton T, de Vito L, Csobod E, Szuppinger P, Heves G (2017). D3.1. Review of social science in air quality and carbon management. Final Revised April 2018. Geneva: Zenodo. doi: 10.5281/zenodo.3972106.

Chen J, Hoek G (2020). Long-term exposure to PM and all-cause and cause-specific mortality: a systematic review and meta-analysis. *Environ Int.* 143:105974. doi: 10.1016/j.envint.2020.105974. License: [CC BY-NC-ND](#).

Chen K, Breitner S, Wolf K, Stafoggia M, Sera F, Vicedo-Cabrera AM et al. (2021). Ambient carbon monoxide and daily mortality: a global time-series study in 337 cities. *Lancet Planet Health.* 5(4):e191–9. doi: 10.1016/S2542-5196(21)00026-7.

Chen R, Yin P, Meng X, Liu C, Wang L, Xu X et al. (2017). Fine particulate air pollution and daily mortality. A nationwide analysis in 272 Chinese cities. *Am J Respir Crit Care Med.* 196(1):73–81. doi: 10.1164/rccm.201609-1862OC.

Chen X, Zhang LW, Huang JJ, Song FJ, Zhang LP, Qian ZM et al. (2016). Long-term exposure to urban air pollution and lung cancer mortality: a 12-year cohort study in Northern China. *Sci Total Environ.* 571:855–61. doi: 10.1016/j.scitotenv.2016.07.064.

Cheng MF, Tsai SS, Yang CY (2009). Air pollution and hospital admissions for myocardial infarction in a tropical city: Kaohsiung, Taiwan. *J Toxicol Environ Health A.* 72(19):1135–40. doi: 10.1080/15287390903091756.

Clancy L, Goodman P, Sinclair H, Dockery DW (2002). Effect of air-pollution control on death rates in Dublin, Ireland: an intervention study. *Lancet.* 360(9341):1210–14. doi: 10.1016/S0140-6736(02)11281-5.

Cohen AJ, Brauer M, Burnett R, Anderson HR, Frostad J, Estep K et al. (2017). Estimates and 25-year trends of the global burden of disease attributable to ambient air pollution: an analysis of data from the Global Burden of Diseases Study 2015. *Lancet.* 389(10082):1907–18. doi: 10.1016/S0140-6736(17)30505-6.

Cohen P, Potchter O, Schnell I (2014). The impact of an urban park on air pollution and noise levels in the Mediterranean city of Tel-Aviv, Israel. *Environ Pollut.* 195:73–83. doi: 10.1016/j.envpol.2014.08.015.

Collaboration for Environmental Evidence (2013). Guidelines for systematic review and evidence synthesis in environmental management. Version 4.2. *Environ Evid.* (<http://www.environmentalevidence.org/Documents/Guidelines/Guidelines4.2.pdf>, accessed 5 December 2020).

Crouse DL, Peters PA, Hystad P, Brook JR, van Donkelaar A, Martin RV et al. (2015). Ambient PM_{2.5}, O₃, and NO₂ exposures and associations with mortality over 16 years of follow-up in the Canadian Census Health and Environment Cohort (CanCHEC). *Environ Health Perspect.* 123(11):1180–6. doi: 10.1289/ehp.1409276.

de Hoogh K, Gulliver J, van Donkelaar A, Martin RV, Marshall JD, Bechle MJ et al. (2016). Development of West-European PM_{2.5} and NO₂ land use regression models incorporating satellite-derived and chemical transport modelling data. *Environ Res.* 151:1–10. doi: 10.1016/j.envres.2016.07.005.

de Hoogh K, Chen J, Gulliver J, Hoffmann B, Hertel O, Ketzel M et al. (2018). Spatial PM_{2.5}, NO₂, O₃ and BC models for Western Europe: evaluation of spatiotemporal stability. *Environ Int.* 120:81–92. doi: 10.1016/j.envint.2018.07.036.

de Jesus AL, Rahman MM, Mazaheri M, Thompson H, Knibbs LD, Jeong C et al. (2019). Ultrafine particles and PM_{2.5} in the air of cities around the world: are they representative of each other? *Environ Int.* 129:118–35. doi: 10.1016/j.envint.2019.05.021.

de Longueville F, Ozer P, Doumbia S, Henry S (2013). Desert dust impacts on human health: an alarming worldwide reality and a need for studies in West Africa. *Int J Biometeorol.* 57(1):1–19. doi: 10.1007/s00484-012-0541-y.

Dehbi HM, Blangiardo M, Gulliver J, Fecht D, de Hoogh K, Al-Kanaani Z et al. (2017). Air pollution and cardiovascular mortality with over 25 years follow-up: a combined analysis of two British cohorts. *Environ Int.* 99:275–81. doi: 10.1016/j.envint.2016.12.004.

Desikan A, Crichton S, Hoang U, Barratt B, Beevers SD, Kelly FJ et al. (2016). Effect of exhaust- and nonexhaust-related components of particulate matter on long-term survival after stroke. *Stroke.* 47(12):2916–22. doi: 10.1161/STROKEAHA.116.014242.

Di Q, Wang Y, Zanobetti A, Wang Y, Koutrakis P, Choirat C et al. (2017a). Air pollution and mortality in the Medicare population. *N Engl J Med.* 376:2513–22. doi: 10.1056/NEJMoa1702747. Copyright © 2017 Massachusetts Medical Society.

Di Q, Dai L, Wang Y, Zanobetti A, Choirat C, Schwartz JD et al. (2017b). Association of short-term exposure to air pollution with mortality in older adults. *JAMA.* 318(24):2446–56. doi: 10.1001/jama.2017.17923.

Dimakopoulou K, Samoli E, Beelen R, Stafoggia M, Andersen ZJ, Hoffmann B et al. (2014). Air pollution and nonmalignant respiratory mortality in 16 cohorts within the ESCAPE project. *Am J Respir Crit Care Med.* 189(6):684–96. doi: 10.1164/rccm.201310-1777OC.

Dirgawati M, Hinwood A, Nedkoff L, Hankey GJ, Yeap BB, Flicker L et al. (2019). Long-term exposure to low air pollutant concentrations and the relationship with all-cause mortality and stroke in older men. *Epidemiology.* 30(suppl 1):S82–9. doi: 10.1097/EDE.0000000000001034.

Dockery DW, Pope CA III, Xu X, Spengler JD, Ware JH, Fay ME et al. (1993). An association between air pollution and mortality in six US cities. *N Engl J Med.* 329(24):1753–9. doi: 10.1056/NEJM199312093292401.

Dockery DW, Rich DQ, Goodman PG, Clancy L, Ohman-Strickland P, George P et al. (2013). Effect of air pollution control on mortality and hospital admissions in Ireland. *Res Rep Health Eff Inst.* (176):3–109. PMID: 24024358.

Dominici F, Schwartz J, Di Q, Braun D, Choirat C, Zanobetti A (2019). Assessing adverse health effects of long-term exposure to low levels of ambient air pollution: phase 1. *Res Rep Health Eff Inst.* (200):1–51. PMID: 31909579.

Donovan GH (2017). Including public-health benefits of trees in urban-forestry decision making. *Urban For Urban Green.* 22:120–3. doi: <https://doi.org/10.1016/j.ufug.2017.02.010>.

EEA (2019). EMEP/EEA air pollutant emission inventory guidebook 2019. Technical guidance to prepare national emission inventories 1994–2019. Copenhagen: European Environment Agency (EEA Report No. 13/2019; <https://www.eea.europa.eu/publications/emep-eea-guidebook-2019>, accessed 7 August 2020).

EEA (2020). Air quality in Europe – 2020 report. Copenhagen: European Environment Agency (EEA Report No. 09/2020; <https://www.eea.europa.eu/publications/air-quality-in-europe-2020-report>, accessed 18 February 2021).

Egorov AI, Mudu P, Braubach M, Martuzzi M (2016). Urban green spaces and health. Copenhagen: World Health Organization Regional Office for Europe (https://www.euro.who.int/__data/assets/pdf_file/0005/321971/Urban-green-spaces-and-health-review-evidence.pdf, accessed 8 August 2020).

Enstrom JE (2005). Fine particulate air pollution and total mortality among elderly Californians, 1973–2002. *Inhal Toxicol.* 17(14):803–16. doi: 10.1080/08958370500240413.

Escudero M, Querol X, Pey J, Alastuey A, Pérez N, Ferreira F et al. (2007). A methodology for the quantification of the net African dust load in air quality monitoring networks. *Atmos Environ.* 41:5516–24. doi: <https://doi.org/10.1016/j.atmosenv.2007.04.047>.

Eum KD, Kazemiparkouhi F, Wang B, Manjourides J, Pun V, Pavlu V et al. (2019). Long-term NO₂ exposures and cause-specific mortality in American older adults. *Environ Int.* 124:10–15. doi: 10.1016/j.envint.2018.12.060.

European Parliament, Council of the European Union (2008). Directive 2008/50/EC of the European Parliament and of the Council of 21 May 2008 on ambient air quality and cleaner air for Europe. OJ. L152/1 (<https://eur-lex.europa.eu/legal-content/EN/TXT/PDF/?uri=CELEX:32008L0050&from=en>, accessed 17 November 2020).

Evangelopoulos D, Perez-Velasco R, Walton H, Gumy S, Williams M, Kelly F et al. (2020). The role of burden of disease assessment in tracking progress towards achieving WHO global air quality guidelines. *Int J Public Health.* 65:1455–65. doi: 10.1007/s00038-020-01479-z.

FAO (2009). Guidelines for good forestry and range practices in arid and semi-arid zones of the Near East. Cairo: FAO Regional Office for the Near East (Working paper RNEO 1-09; <http://www.fao.org/3/al040e/al040e00.pdf>, accessed 13 July 2021).

FAO (2021). Action Against Desertification [website]. Rome: Food and Agriculture Organization of the United Nations (<http://www.fao.org/in-action/action-against-desertification/overview/en/>, accessed 2 March 2021).

Faustini A, Rapp R, Forastiere F (2014). Nitrogen dioxide and mortality: review and meta-analysis of long-term studies. *Eur Respir J.* 44(3):744–53. doi: 10.1183/09031936.00114713.

Filleul L, Rondeau V, Vandentorren S, Le Moual N, Cantagrel A, Annesi-Maesano I et al. (2005). Twenty five year mortality and air pollution: results from the French PAARC survey. *Occup Environ Med.* 62(7):453–60. doi: 10.1136/oem.2004.014746.

Fischer PH, Marra M, Ameling CB, Hoek G, Beelen R, De Hoogh K et al. (2015). Air pollution and mortality in seven million adults: the Dutch Environmental Longitudinal Study (DUELS). *Environ Health Perspect.* 123(7):697–704. doi: <http://dx.doi.org/10.1289/ehp.1408254>.

Fischer PH, Marra M, Ameling CB, Velders GJM, Hoogerbrugge R, de Vries W et al. (2020). Particulate air pollution from different sources and mortality in 7.5 million adults: the Dutch Environmental Longitudinal Study (DUELS). *Sci Total Environ.* 705:135778. doi: 10.1016/j.scitotenv.2019.135778.

Frampton MW, Balmes JR, Bromberg PA, Stark P, Arjomandi M, Hazucha MJ et al. (2017). Multicenter Ozone Study in older Subjects (MOSES): part 1. Effects of exposure to low concentrations of ozone on respiratory and cardiovascular outcomes. Boston (MA): Health Effects Institute (Research Report 192 Part 1; <https://www.healtheffects.org/publication/multicenter-ozone-study-older-subjects-moses-part-1-effects-exposure-low-concentrations>, accessed 21 February 2021).

Fussell JC, Kelly FJ. (2021). Mechanisms underlying the health effects of desert sand dust. *Environ Int.* 157:106790. doi: 10.1016/j.envint.2021.106790.

GBD 2016 Risk Factors Collaborators (2017). Global, regional, and national comparative risk assessment of 84 behavioural, environmental and occupational, and metabolic risks or clusters of risks, 1990–2016: a systematic analysis for the Global Burden of Disease Study 2016. *Lancet*. 390(10100):1345–422. doi: 10.1016/S0140-6736(17)32366-8.

GBD 2019 Risk Factors Collaborators (2020). Global burden of 87 risk factors in 204 countries and territories, 1990–2019: a systematic analysis for the Global Burden of Disease Study 2019. *Lancet*. 396(10258):1223–49. doi: 10.1016/S0140-6736(20)30752-2.

Geddes JA, Martin RV, Boys BL, van Donkelaar A (2016). Long-term trends worldwide in ambient NO₂ concentrations inferred from satellite observations. *Environ Health Perspect*. 124(3):281–9. doi: 10.1289/ehp.1409567.

Goudie AS (2014). Desert dust and human health disorders. *Environ Int*. 63:101–13. doi: 10.1016/j.envint.2013.10.011.

Government of South Australia, WHO (2017). Progressing the Sustainable Development Goals through Health in All Policies: case studies from around the world. Adelaide: Government of South Australia (https://www.who.int/social_determinants/publications/progressing-sdg-case-studies-2017.pdf?ua=1, accessed 17 November 2020).

Gryparis A, Forsberg B, Katsouyanni K, Analitis A, Touloumi G, Schwartz J et al. (2004). Acute effects of ozone on mortality from the “air pollution and health: a European approach” project. *Am J Respir Crit Care Med*. 170(10):1080–7. doi: 10.1164/rccm.200403-333OC.

Hammer MS, van Donkelaar A, Li C, Lyapustin A, Sayer AM, Hsu NC et al. (2020). Global estimates and long-term trends of fine particulate matter concentrations (1998–2018). *Environ Sci Technol*. 54(13):7879–90. doi: 10.1021/acs.est.0c01764.

Hanigan IC, Rolfe MI, Knibbs LD, Salimi F, Cowie CT, Heyworth J et al. (2019). All-cause mortality and long-term exposure to low level air pollution in the “45 and up study” cohort, Sydney, Australia, 2006–2015. *Environ Int*. 126:762–70. doi: 10.1016/j.envint.2019.02.044.

Hansell A, Ghosh RE, Blangiardo M, Perkins C, Vienneau D, Goffe K et al. (2016). Historic air pollution exposure and long-term mortality risks in England and Wales: prospective longitudinal cohort study. *Thorax*. 71(4):330–8. doi: 10.1136/thoraxjnl-2015-207111.

Hansen ADA (2005). The Aethalometer. Berkeley (CA): Magee Scientific Company (https://www.psi.ch/sites/default/files/import/lac/ProjectAddonCatcosOperationsEN/Aethalometer_book_2005.07.02.pdf, accessed 6 August 2020).

Hart JE, Garshick E, Dockery DW, Smith TJ, Ryan L, Laden F (2011). Long-term ambient multipollutant exposures and mortality. *Am J Respir Crit Care Med*. 183(1):73–8. doi: 10.1164/rccm.200912-1903OC.

Hart JE, Rimm EB, Rexrode KM, Laden F (2013). Changes in traffic exposure and the risk of incident myocardial infarction and all-cause mortality. *Epidemiology*. 24(5):734–42. doi: 10.1097/EDE.0b013e31829d5dae.

Hart JE, Liao X, Hong B, Puett RC, Yanosky JD, Suh H et al. (2015). The association of long-term exposure to PM_{2.5} on all-cause mortality in the Nurses’ Health Study and the impact of measurement-error correction. *Environ Health*. 14:38. doi: 10.1186/s12940-015-0027-6.

Hartiala J, Breton CV, Tang WHW, Lurmann F, Hazen SL, Gilliland FD et al. (2016). Ambient air pollution is associated with the severity of coronary atherosclerosis and incident myocardial infarction in patients undergoing elective cardiac evaluation. *J Am Heart Assoc*. 5(8):e003947. doi: 10.1161/JAHA.116.003947.

Hashizume M, Ueda K, Nishiwaki Y, Michikawa T, Onozuka D (2010). Health effects of Asian dust events: a review of the literature. *Nihon Eiseigaku Zasshi*. 65(3):413–21 (in Japanese). doi: 10.1265/jjh.65.413.

Health Canada (2010). Residential indoor air quality guideline: carbon monoxide. Ottawa: Health Canada (<https://www.canada.ca/en/health-canada/services/publications/healthy-living/residential-indoor-air-quality-guideline-carbon-monoxide.html>, accessed 25 June 2021).

Health Canada (2013). Canadian smog science assessment. Volume 2: health effects. Ottawa: Health Canada (<https://publications.gc.ca/site/eng/447367/publication.html>, accessed 12 July 2021).

Health Canada (2016a). Human health risk assessment for ambient nitrogen dioxide. Ottawa: Health Canada (<https://www.canada.ca/en/health-canada/services/publications/healthy-living/human-health-risk-assessment-ambient-nitrogen-dioxide.html>, accessed 12 July 2021).

Health Canada (2016b). Human health risk assessment for sulphur dioxide – analysis of ambient exposure to and health effects of sulphur dioxide in the Canadian population. Ottawa: Health Canada (https://publications.gc.ca/collections/collection_2016/sc-hc/H144-29-2016-eng.pdf, accessed 12 July 2021).

HEI (2013). Understanding the health effects of ambient ultrafine particles. HEI Review Panel on Ultrafine Particles. Boston (MA): Health Effects Institute (HEI Perspectives 3; <https://www.healtheffects.org/system/files/Perspectives3.pdf>, accessed 7 August 2020).

HEI (2021). Mortality and morbidity effects of long-term exposure to low-level PM_{2.5}, black carbon, NO₂ and O₃: an analysis of European cohorts. In: Ongoing research [website]. Boston (MA): Health Effects Institute (<https://www.healtheffects.org/research/ongoing-research/mortality-and-morbidity-effects-long-term-exposure-low-level-pm25-black>, accessed 6 July 2021).

HEI Accountability Working Group (2003). Assessing health impact of air quality regulations: concepts and methods for accountability research. Boston (MA): Health Effects Institute (Communication 11; <https://www.healtheffects.org/system/files/Comm11.pdf>, accessed 15 March 2021).

Heinrich J, Thiering E, Rzehak P, Krämer U, Hochadel M, Rauchfuss KM et al. (2013). Long-term exposure to NO₂ and PM₁₀ and all-cause and cause-specific mortality in a prospective cohort of women. *Occup Environ Med*. 70(3):179–86. doi: 10.1136/oemed-2012-100876.

Higgins JPT, Green S, editors (2011). Cochrane handbook for systematic reviews of interventions. Version 5.1.0 (updated March 2011). The Cochrane Collaboration (<https://training.cochrane.org/handbook/archive/v5.1/>, accessed 6 December 2020).

Hoek G, Beelen R, de Hoogh K, Vienneau D, Gulliver J, Fischer P et al. (2008). A review of land-use regression models to assess spatial variation of outdoor air pollution. *Atmos Environ*. 42(33):7561–78. doi: <https://doi.org/10.1016/j.atmosenv.2008.05.057>.

Hsieh YL, Yang YH, Wu TN, Yang CY (2010). Air pollution and hospital admissions for myocardial infarction in a subtropical city: Taipei, Taiwan. *J Toxicol Environ Health A*. 73(11):757–65. doi: 10.1080/15287391003684789.

Huangfu P, Atkinson R (2020). Long-term exposure to NO₂ and O₃ and all-cause and respiratory mortality: a systematic review and meta-analysis. *Environ Int*. 144:105998. doi: 10.1016/j.envint.2020.105998. License: CC BY-NC-ND.

Hultcrantz M, Rind D, Akl EA, Treweek S, Mustafa RA, Iorio A et al. (2017). The GRADE working group clarifies the construct of certainty of evidence. *J Clin Epidemiol*. 87:4–13. doi: 10.1016/j.jclinepi.2017.05.006.

Hvidtfeldt UA, Sorensen M, Geels C, Ketzel M, Khan J, Tjonneland A et al. (2019). Long-term residential exposure to PM_{2.5}, PM₁₀, black carbon, NO₂, and ozone and mortality in a Danish cohort. *Environ Int*. 123:265–72. doi: 10.1016/j.envint.2018.12.010.

Hystad P, Setton E, Cervantes A, Poplawski K, Deschenes S, Brauer M et al. (2011). Creating national air pollution models for population exposure assessment in Canada. *Environ Health Perspect*. 119(8):1123–9. doi: 10.1289/ehp.1002976.

IDAEA (2013). The scientific basis of street cleaning activities as road dust mitigation measure. Barcelona: Institute of Environmental Assessment and Water Research (http://airuse.eu/wp-content/uploads/2013/11/B7-3-ES_road-cleaning.pdf, accessed 2 December 2020).

International Programme on Chemical Safety (1997). Nitrogen oxides, 2nd edition. Environmental Health Criteria 188. Geneva: United Nations Environment Programme, International Labour Organization, World Health Organization (<http://www.inchem.org/documents/ehc/ehc/ehc188.htm>, accessed 17 June 2021).

Int'Hout J, Ioannidis JP, Rovers MM, Goeman JJ (2016). Plea for routinely presenting prediction intervals in meta-analysis. *BMJ Open*. 6(7):e010247. doi: 10.1136/bmjopen-2015-010247.

ISO (2015). ISO 27891:2015. Aerosol particle number concentration: calibration of condensation particle counters. ISO/TC 24/SC 4. Geneva: International Organization for Standardization (<https://www.iso.org/standard/44414.html>, accessed 7 August 2020).

Janjua S, Powell P, Atkinson R, Stovold E, Fortescue R (2019). Individual-level interventions to reduce personal exposure to outdoor air pollution and their effects on long-term respiratory conditions. *Cochrane Database of Syst. Rev.* (10):CD013441. doi: 10.1002/14651858.CD013441.

Janssen NAH, Gerlofs-Nijland ME, Lanki T, Salonen RO, Cassee F, Hoek G et al. (2012). Health effects of black carbon. Copenhagen: WHO Regional Office for Europe (https://www.euro.who.int/__data/assets/pdf_file/0004/162535/e96541.pdf, accessed 7 August 2020).

Jeronimo M, Stewart Q, Weakley AT, Giacomo J, Zhang X, Hyslop N et al. (2020). Analysis of black carbon on filters by image-based reflectance. *Atmos Environ*. 223:117300. doi: 10.1016/j.atmosenv.2020.117300.

Jerrett M, Finkelstein MM, Brook JR, Arain MA, Kanaroglou P, Stieb DM et al. (2009). A cohort study of traffic-related air pollution and mortality in Toronto, Ontario, Canada. *Environ Health Perspect*. 117(5):772–7. doi: 10.1289/ehp.11533.

Jindal R, Swallow B, Kerr J (2008). Forestry-based carbon sequestration projects in Africa: potential benefits and challenges. *Natural Resources Forum*. 32(2):116–30. doi: <https://doi.org/10.1111/j.1477-8947.2008.00176.x>.

Joint Task Force on the Health Aspects of Air Pollution (2018). Effects of air pollution on health. Report of the Joint Task Force on the Health Aspects of Air Pollution on its twenty-first meeting. Geneva: United Nations Economic Commission for Europe (https://unece.org/fileadmin/DAM/env/documents/2018/Air/EMEP/ECE_EB.AIR_GE.1_2018_17-1811946E.pdf, accessed 15 March 2021).

Karanasiou A, Moreno N, Moreno T, Viana M, De Leeuw F, Querol X (2012). Health effects from Sahara dust episodes in Europe: literature review and research gaps. *Environ Int.* 47:107–14. doi: 10.1016/j.envint.2012.06.012.

Karner AA, Eisinger DS, Niemeier DA (2010). Near-roadway air quality: synthesizing the findings from real-world data. *Environ Sci Technol.* 44(14):5334–44. doi: 10.1021/es100008x.

Katanoda K, Sobue T, Satoh H, Tajima K, Suzuki T, Nakatsuka H et al. (2011). An association between long-term exposure to ambient air pollution and mortality from lung cancer and respiratory diseases in Japan. *J Epidemiol.* 21(2):132–43. doi: 10.2188/jea.20100098.

Katra I, Krasnov H (2020). Exposure assessment of indoor PM levels during extreme dust episodes. *Int J Environ Res Public Health.* 17(5):1625. doi: 10.3390/ijerph17051625.

Kazemiparkouhi F, Eum KD, Wang B, Manjourides J, Suh HH (2020). Long-term ozone exposures and cause-specific mortality in a US Medicare cohort. *J Expo Sci Environ Epidemiol.* 30(4):650–8. doi: 10.1038/s41370-019-0135-4.

Khreis H, Kelly C, Tate J, Parslow R, Lucas K, Nieuwenhuijsen M (2017). Exposure to traffic-related air pollution and risk of development of childhood asthma: a systematic review and meta-analysis. *Environ Int.* 100:1–31. doi: 10.1016/j.envint.2016.11.012.

Kim O-J, Kim S-Y, Kim H (2017). Association between long-term exposure to particulate matter air pollution and mortality in a South Korean national cohort: comparison across different exposure assessment approaches. *Int J Environ Res Public Health.* 14(10):1103. doi: 10.3390/ijerph14101103.

Klompaker JO, Hoek G, Bloemsmas LD, Marra M, Wijga AH, van den Brink C et al. (2020). Surrounding green, air pollution, traffic noise exposure and non-accidental and cause-specific mortality. *Environ Int.* 134:105341. doi: 10.1016/j.envint.2019.105341.

Knibbs LD, Hewson MG, Bechle MJ, Marshall JD, Barnett AG (2014). A national satellite-based land-use regression model for air pollution exposure assessment in Australia. *Environ Res.* 135:204–11. doi: 10.1016/j.envres.2014.09.011.

Krewski D, Burnett RT, Goldberg MS, Hoover BK, Siemiatycki J, Jerrett M et al. (2003). Overview of the reanalysis of the Harvard Six Cities Study and American Cancer Society study of particulate air pollution and mortality. *J Toxicol Environ Health A.* 66(16–19):1507–51. doi: 10.1080/15287390306424.

Kutlar Joss M, Eeftens M, Gintowt E, Kappeler R, Künzli N (2017). Time to harmonize national ambient air quality standards. *Int J Public Health.* 62(4):453–62. doi: 10.1007/s00038-017-0952-y. License: [CC BY 4.0](#).

Laden F, Schwartz J, Speizer FE, Dockery DW (2006). Reduction in fine particulate air pollution and mortality: extended follow-up of the Harvard Six Cities Study. *Am J Respir Crit Care Med.* 173(6):667–72. doi: 10.1164/rccm.200503-443OC.

Larkin A, Geddes JA, Martin RV, Xiao Q, Liu Y, Marshall JD et al. (2017). Global land use regression model for nitrogen dioxide air pollution. *Environ Sci Technol.* 51(12):6957–64. doi: 10.1021/acs.est.7b01148.

Lee KK, Spath N, Miller MR, Mills NL, Shah ASV (2020). Short-term exposure to carbon monoxide and myocardial infarction: a systematic review and meta-analysis. *Environ Int.* 143:105901. doi: 10.1016/j.envint.2020.105901. License: [CC BY-NC-ND](#).

- Lefler JS, Higbee JD, Burnett RT, Ezzati M, Coleman NC, Mann DD et al. (2019). Air pollution and mortality in a large, representative US cohort: multiple-pollutant analyses, and spatial and temporal decompositions. *Environ Health*. 18(1):101. doi: 10.1186/s12940-019-0544-9.
- Lepeule J, Laden F, Dockery D, Schwartz J (2012). Chronic exposure to fine particles and mortality: an extended follow-up of the Harvard Six Cities Study from 1974 to 2009. *Environ Health Perspect*. 120(7):965–70. doi: 10.1289/ehp.1104660.
- Lewis AC, von Schneidmesser E, Peltier RE (2018). Low-cost sensors for the measurement of atmospheric composition: overview of topic and future applications. Geneva: World Meteorological Organization (<https://www.ccacoalition.org/en/resources/low-cost-sensors-measurement-atmospheric-composition-overview-topic-and-future>, accessed 2 December 2020).
- Lim CC, Hayes RB, Ahn J, Shao Y, Silverman DT, Jones RR et al. (2019). Long-term exposure to ozone and cause-specific mortality risk in the United States. *Am J Respir Crit Care Med*. 200(8):1022–31. doi: 10.1164/rccm.201806-1161OC. Copyright © 2019 American Thoracic Society. All rights reserved.
- Lim SS, Vos T, Flaxman AD, Danaei G, Shibuya K, Adair-Rohani H et al. (2012). A comparative risk assessment of burden of disease and injury attributable to 67 risk factors and risk factor clusters in 21 regions, 1990–2010: a systematic analysis for the Global Burden of Disease Study 2010. *Lancet*. 380(9859):2224–60. doi: 10.1016/S0140-6736(12)61766-8.
- Lipfert FW, Wyzga RE, Baty JD, Miller JP (2006). Traffic density as a surrogate measure of environmental exposures in studies of air pollution health effects: long-term mortality in a cohort of US veterans. *Atmos Environ*. 40:154–69. doi: <https://doi.org/10.1016/j.atmosenv.2005.09.027>.
- Lippmann M, Chen LC, Gordon T, Ito K, Thurston GD (2013). National Particle Component Toxicity (NPACT) Initiative: integrated epidemiologic and toxicologic studies of the health effects of particulate matter components. *Res Rep Health Eff Inst*. (177):5–13. PMID: 24377209.
- Lipsett MJ, Ostro BD, Reynolds P, Goldberg D, Hertz A, Jerrett M et al. (2011). Long-term exposure to air pollution and cardiorespiratory disease in the California Teachers Study cohort. *Am J Respir Crit Care Med*. 184(7):828–35. doi: 10.1164/rccm.201012-2082OC.
- Liu C, Chen R, Sera F, Vicedo-Cabrera AM, Guo Y, Tong S et al. (2019). Ambient particulate air pollution and daily mortality in 652 cities. *N Engl J Med*. 381(8):705–15. doi: 10.1056/NEJMoa1817364. Copyright © 2019 Massachusetts Medical Society.
- Longhurst J, Barnes J, Chatterton T, de Vito L, Everard M, Hayes E et al. (2018). Analysing air pollution and its management through the lens of the UN Sustainable Development Goals: a review and assessment. *WIT Trans Ecol Environ*. 230(1):3–14. doi: 10.2495/AIR180011.
- Luben TJ, Nichols JL, Dutton SJ, Kirrane E, Owens EO, Datko-Williams L et al. (2017). A systematic review of cardiovascular emergency department visits, hospital admissions and mortality associated with ambient black carbon. *Environ Int*. 107:154–62. doi: <https://doi.org/10.1016/j.envint.2017.07.005>.
- Markevych I, Schoierer J, Hartig T, Chudnovsky A, Hystad P, Dzhambov AM et al. (2017). Exploring pathways linking greenspace to health: theoretical and methodological guidance. *Environ Res*. 158:301–17. doi: 10.1016/j.envres.2017.06.028.
- Mathur MB, VanderWeele TJ (2020). Sensitivity analysis for unmeasured confounding in meta-analyses. *J Am Stat Assoc*. 115(529):163–72. doi: 10.1080/01621459.2018.1529598.

- McDonnell WF, Nishino-Ishikawa N, Petersen FF, Chen LH, Abbey DE (2000). Relationships of mortality with the fine and coarse fractions of long-term ambient PM₁₀ concentrations in nonsmokers. *J Expo Anal Environ Epidemiol*. 10(5):427–36. doi: 10.1038/sj.jea.7500095.
- Meng X, Liu C, Chen R, Sera F, Vicedo-Cabrera AM, Milojevic A et al. (2021). Short term associations of ambient nitrogen dioxide with daily total, cardiovascular, and respiratory mortality: multilocation analysis in 398 cities. *BMJ*. 372:n534. doi: 10.1136/bmj.n534.
- Middleton N, Kang U (2017). Sand and dust storms: impact mitigation. *Sustainability*. 9(6):1053. doi: <https://doi.org/10.3390/su9061053>.
- Middleton NJ (2017). Desert dust hazards: a global review. *Aeolian Res*. 24:53–63. doi: <https://doi.org/10.1016/j.aeolia.2016.12.001>.
- Miller J, Jin L (2019). Global progress toward soot-free diesel vehicles in 2019. Washington (DC): International Council on Clean Transportation (<https://theicct.org/publications/global-progress-toward-soot-free-diesel-vehicles-2019>, accessed 7 August 2020).
- Mills IC, Atkinson RW, Anderson HR, Maynard RL, Strachan DP (2016). Distinguishing the associations between daily mortality and hospital admissions and nitrogen dioxide from those of particulate matter: a systematic review and meta-analysis. *BMJ Open*. 6(7):e010751. doi: 10.1136/bmjopen-2015-010751.
- Morawska L, Ristovski ZD, Jayaratne ER, Keogh D, Ling X (2008). Ambient nano and ultrafine particles from motor vehicle emissions: characteristics, ambient processing and implications on human exposure. *Atmos Environ*. 42(35):8113–38. doi: <https://doi.org/10.1016/j.atmosenv.2008.07.050>.
- Morgan RL, Thayer KA, Bero L, Bruce N, Falck-Ytter Y, Ghersi D et al. (2016). GRADE: assessing the quality of evidence in environmental and occupational health. *Environ Int*. 92–93:611–16. doi: 10.1016/j.envint.2016.01.004.
- Morgan RL, Thayer KA, Santesso N, Holloway AC, Blain R, Eftim SE et al. (2019). A risk of bias instrument for non-randomized studies of exposures: a users' guide to its application in the context of GRADE. *Environ Int*. 122:168–84. doi: 10.1016/j.envint.2018.11.004.
- Mori I, Nishikawa M, Tanimura T, Quan H (2003). Change in size distribution and chemical composition of kosa (Asian dust) aerosol during long-range transport. *Atmos Environ*. 37(30):4253–63. doi: [https://doi.org/10.1016/S1352-2310\(03\)00535-1](https://doi.org/10.1016/S1352-2310(03)00535-1).
- Moshhammer H, Poteser M, Kundi M, Lemmerer K, Weitensfelder L, Wallner P et al. (2020). Nitrogen-dioxide remains a valid air quality indicator. *Int J Environ Res Public Health*. 17(10):3733. doi: 10.3390/ijerph17103733. License: CC BY.
- National Research Council (2012). Exposure science in the 21st century: a vision and a strategy. Washington (DC): The National Academies Press.
- National Research Council (2014). Review of EPA's Integrated Risk Information System (IRIS) process. Washington (DC): The National Academies Press.
- NHCO (2019). National Health Assembly [website]. Nonthaburi: National Health Commission Office (<https://en.nationalhealth.or.th/nha/>, accessed 17 November 2020).
- NIHR (2021). PROSPERO. International prospective register of systematic reviews [database]. York: National Institute for Health Research (<https://www.crd.york.ac.uk/PROSPERO/>, accessed 7 July 2021).

Nordstrom KF, Hotta S (2004). Wind erosion from crop-land in the USA: a review of problems, solutions, and prospects. *Geoderma*. 121(3-4):157-67. doi: <https://doi.org/10.1016/j.geoderma.2003.11.012>.

Novotny EV, Bechle MJ, Millet DB, Marshall JD (2011). National satellite-based land-use regression: NO₂ in the United States. *Environ Sci Technol*. 45(10):4407-14. doi: 10.1021/es103578x.

Nowak DJ, Heisler GM (2010). Air quality effects of urban trees and parks. Ashburn (VA): National Recreation and Parks Association Research (https://www.fs.fed.us/nrs/pubs/jrnl/2010/nrs_2010_nowak_002.pdf, accessed 8 August 2020).

Ohlwein S, Kappeler R, Kutlar Joss M, Künzli N, Hoffmann B (2019). Health effects of ultrafine particles: a systematic literature review update of epidemiological evidence. *Int J Public Health*. 64(4):547-59. doi: 10.1007/s00038-019-01202-7.

OpenAQ (2021). OpenAQ [online database]. Washington (DC): OpenAQ (<https://openaq.org>, accessed 17 February 2021).

Orellano P, Reynoso J, Quaranta N (2021). Short-term exposure to sulphur dioxide (SO₂) and all-cause and respiratory mortality: a systematic review and meta-analysis. *Environ Int*. 150:106434. doi: [10.1016/j.envint.2021.106434](https://doi.org/10.1016/j.envint.2021.106434). License: CC BY-NC-ND.

Orellano P, Reynoso J, Quaranta N, Bardach A, Ciapponi A (2020). Short-term exposure to particulate matter (PM₁₀ and PM_{2.5}), nitrogen dioxide (NO₂), and ozone (O₃) and all-cause and cause-specific mortality: systematic review and meta-analysis. *Environ Int*. 142:105876. doi: [10.1016/j.envint.2020.105876](https://doi.org/10.1016/j.envint.2020.105876). License: CC BY-NC-ND.

Ostro B, Hu J, Goldberg D, Reynolds P, Hertz A, Bernstein L et al. (2015). Associations of mortality with long-term exposures to fine and ultrafine particles, species and sources: results from the California Teachers Study cohort. *Environ Health Perspect*. 123(6):549-56. doi: 10.1289/ehp.1408565.

Owens EO, Patel MM, Kirrane E, Long TC, Brown J, Cote I et al. (2017). Framework for assessing causality of air pollution-related health effects for reviews of the national ambient air quality standards. *Regul Toxicol Pharmacol*. 88:332-7. doi: 10.1016/j.yrtph.2017.05.014.

Pappin AJ, Christidis T, Pinault LL, Crouse DL, Brook JR, Erickson A et al. (2019). Examining the shape of the association between low levels of fine particulate matter and mortality across three cycles of the Canadian Census Health and Environment Cohort. *Environ Health Perspect*. 127(10):107008. doi: 10.1289/EHP5204.

Parker JD, Kravets N, Vaidyanathan A (2018). Particulate matter air pollution exposure and heart disease mortality risks by race and ethnicity in the United States: 1997 to 2009 National Health Interview Survey with mortality follow-up through 2011. *Circulation*. 137(16):1688-97. doi: 10.1161/CIRCULATIONAHA.117.029376.

Paul KC, Haan M, Mayeda ER, Ritz BR (2019). Ambient air pollution, noise, and late-life cognitive decline and dementia risk. *Annu Rev Public Health*. 40:203-20. doi: 10.1146/annurev-publhealth-040218-044058.

Perez L, Künzli N (2011). Saharan dust: no reason to exempt from science or policy. *Occup Environ Med*. 68(6):389-90. doi: 10.1136/oem.2010.063990.

Peters R, Ee N, Peters J, Booth A, Mudway I, Anstey KJ (2019). Air pollution and dementia: a systematic review. *J Alzheimers Dis*. 70:S145-63. doi: 10.3233/JAD-180631.

Petzold A, Ogren JA, Fiebig M, Laj P, Li SM, Baltensperger U et al. (2013). Recommendations for reporting “black carbon” measurements. *Atmos Chem Phys*. 13:8365–79. doi: <https://doi.org/10.5194/acp-13-8365-2013>, 2013.

Pey J, Querol X, Alastuey A, Forastiere F, Stafoggia M (2013). African dust outbreaks over the Mediterranean Basin during 2001–2011: PM₁₀ concentrations, phenomenology and trends, and its relation with synoptic and mesoscale meteorology. *Atmos Chem Phys*. 13:1395–410. doi: <https://doi.org/10.5194/acp-13-1395-2013>.

PHAC, WHO (2008). Health equity through intersectoral action: an analysis of 18 country case studies. Ottawa: Public Health Agency of Canada (https://www.who.int/social_determinants/resources/health_equity_isa_2008_en.pdf?ua=1, accessed 17 November 2020).

PHE (2018). Associations of long-term average concentrations of nitrogen dioxide with mortality: a report by the Committee on the Medical Effects of Air Pollutants. London: Public Health England (https://assets.publishing.service.gov.uk/government/uploads/system/uploads/attachment_data/file/734799/COMEAP_NO2_Report.pdf, accessed 21 February 2021).

PHE (2020). Review of interventions to improve outdoor air quality and public health: principal interventions for local authorities. London: Public Health England (https://assets.publishing.service.gov.uk/government/uploads/system/uploads/attachment_data/file/795185/Review_of_interventions_to_improve_air_quality.pdf, accessed 5 December 2020).

Pinault L, Tjepkema M, Crouse DL, Weichenthal S, van Donkelaar A, Martin RV et al. (2016). Risk estimates of mortality attributed to low concentrations of ambient fine particulate matter in the Canadian Community Health Survey cohort. *Environ Health*. 15:18. doi: 10.1186/s12940-016-0111-6. License: [CC BY 4.0](https://creativecommons.org/licenses/by/4.0/).

Pinault LL, Weichenthal S, Crouse DL, Brauer M, Erickson A, Donkelaar AV et al. (2017). Associations between fine particulate matter and mortality in the 2001 Canadian Census Health and Environment Cohort. *Environ Res*. 159:406–15. doi: 10.1016/j.envres.2017.08.037.

Pope CA III, Ezzati M, Dockery DW (2009). Fine-particulate air pollution and life expectancy in the United States. *N Engl J Med*. 360(4):376–86. doi: 10.1056/NEJMsa0805646.

Pope CA III, Burnett RT, Thun MJ, Calle EE, Krewski D, Ito K et al. (2002). Lung cancer, cardiopulmonary mortality, and long-term exposure to fine particulate air pollution. *JAMA*. 287(9):1132–41. doi: 10.1001/jama.287.9.1132.

Puett RC, Schwartz J, Hart JE, Yanosky JD, Speizer FE, Suh H et al. (2008). Chronic particulate exposure, mortality, and coronary heart disease in the nurses’ health study. *Am J Epidemiol*. 168(10):1161–8. doi: 10.1093/aje/kwn232.

Puett RC, Hart JE, Suh H, Mittleman M, Laden F (2011). Particulate matter exposures, mortality, and cardiovascular disease in the health professionals follow-up study. *Environ Health Perspect*. 119(8):1130–5. doi: 10.1289/ehp.1002921.

Querol X, Pey J, Pandolfi M, Alastuey A, Cusack M, Perez N et al. (2009). African dust contributions to mean ambient PM₁₀ levels across the Mediterranean Basin. *Atmos Environ*. 43(28):4266–77. doi: <https://doi.org/10.1016/j.atmosenv.2009.06.013>.

Querol X, Perez N, Reche C, Ealo M, Ripoll A, Tur J et al. (2019a). African dust and air quality over Spain: is it only dust that matters? *Sci Total Environ*. 686:737–52. doi: 10.1016/j.scitotenv.2019.05.349.

Querol X, Tobías A, Pérez N, Karanasiou A, Amato F, Stafoggia M et al. (2019b). Monitoring the impact of desert dust outbreaks for air quality for health studies. *Environ Int.* 130:104867. doi: 10.1016/j.envint.2019.05.061.

Rajan D, Mathurapote N, Putthasri W, Posayanonda T, Pinprateep P, de Courcelles S et al. (2017). The triangle that moves the mountain: nine years of Thailand's National Health Assembly (2008–2016). Geneva: World Health Organization (<https://apps.who.int/iris/handle/10665/260464>, accessed 7 December 2020).

Rich DQ, Frampton MW, Balmes JR, Bromberg PA, Arjomandi M, Hazucha MJ et al. (2020). Multicenter Ozone Study in older Subjects (MOSES): part 2. Effects of personal and ambient concentrations of ozone and other pollutants on cardiovascular and pulmonary function. Boston (MA): Health Effects Institute (Research report 192, Part 2; https://www.healtheffects.org/system/files/moses-rr-192-pt2_0.pdf, accessed 12 July 2021).

Rodríguez S, Alastuey A, Alonso-Pérez S, Querol X, Cuevas E, Abreu-Afonso J et al. (2011). Transport of desert dust mixed with North African industrial pollutants in the subtropical Saharan Air Layer. *Atmos Chem Phys.* 11(13):6663–85. doi: <https://doi.org/10.5194/acp-11-6663-2011>.

Rogers D, Tsirkunov V (2010). Global assessment report on disaster risk reduction. Global assessment report on disaster risk reduction: costs and benefits of early warning systems. Washington (DC): International Strategy for Disaster Reduction and the World Bank (<https://documents1.worldbank.org/curated/en/609951468330279598/pdf/693580ESW0P1230aster0Risk0Reduction.pdf>, accessed 12 July 2020).

Rojas-Rueda D, Nieuwenhuijsen MJ, Gascon M, Perez-Leon D, Mudu P (2019). Green spaces and mortality: a systematic review and meta-analysis of cohort studies. *Lancet Planet Health.* 3(11):e469–77. doi: 10.1016/S2542-5196(19)30215-3.

Romieu I, Gouveia N, Cifuentes LA, de Leon AP, Junger W, Vera J et al. (2012). Multicity study of air pollution and mortality in Latin America (the ESCALA study). *Res Rep Health Eff Inst.* (171):5–86. PMID: 23311234.

Rosenlund M, Picciotto S, Forastiere F, Stafoggia M, Perucci CA (2008). Traffic-related air pollution in relation to incidence and prognosis of coronary heart disease. *Epidemiology.* 19(1):121–8. doi: 10.1097/EDE.0b013e31815c1921.

Rothman KJ, Greenland S (2018). Planning study size based on precision rather than power. *Epidemiology.* 29(5):599–603. doi: 10.1097/EDE.0000000000000876.

Sacks JD, Fann N, Gump S, Kim I, Ruggeri G, Mudu P (2020). Quantifying the public health benefits of reducing air pollution: critically assessing the features and capabilities of WHO's AirQ+ and US EPA's Environmental Benefits Mapping and Analysis Program—community edition (BenMAP-CE). *Atmosphere (Basel).* 11(5):1–15. doi: 10.3390/atmos11050516.

Samet JM (2010). Urban air quality. In: Vlahov D, Boufford JL, Pearson CE, Norris L, editors. *Urban health: global perspectives*. Hoboken (NJ): John Wiley & Sons: 317–38.

Samoli E, Peng R, Ramsay T, Pipikou M, Touloumi G, Dominici F et al. (2008). Acute effects of ambient particulate matter on mortality in Europe and North America: results from the APHENA study. *Environ Health Perspect.* 116(11):1480–6. doi: 10.1289/ehp.11345.

Saracci R (2017). The hazards of hazard identification in environmental epidemiology. *Environ Health.* 16(1):85. doi: 10.1186/s12940-017-0296-3.

Schneider A, Cyrus J, Breitner S, Kraus U, Peters A (2018). Quantifizierung von umweltbedingten Krankheitslasten aufgrund der Stickstoffdioxid-Exposition in Deutschland [Quantification of environmental burdens of disease due to nitrogen dioxide exposure in Germany]. Dessau-Roßlau: Umweltbundesamt [German Environment Agency] (https://www.umweltbundesamt.de/sites/default/files/medien/421/publikationen/abschlussbericht_no2_krankheitslast_final_2018_03_05.pdf, accessed 15 March 2021).

Schünemann H, Brożek J, Guyatt G, Oxman A, editors (2013). Handbook for grading the quality of evidence and the strength of recommendations using the GRADE approach (updated October 2013). Hamilton: GRADE Working Group (gdt.guidelinedevelopment.org/app/handbook/handbook.html, accessed 4 November 2020).

Selmi W, Weber C, Rivi re E, Blond N, Mehdi L, Nowak D (2016). Air pollution removal by trees in public green spaces in Strasbourg city, France. *Urban For Urban Green*. 17:192–201. doi: <https://doi.org/10.1016/j.ufug.2016.04.010>.

Shaddick G, Thomas ML, Amini H, Broday D, Cohen A, Frostad J et al. (2018). Data integration for the assessment of population exposure to ambient air pollution for global burden of disease assessment. *Environ Sci Technol*. 52(16):9069–78. doi: 10.1021/acs.est.8b02864.

Sheehan MC, Lam J, Navas-Acien A, Chang HH (2016). Ambient air pollution epidemiology systematic review and meta-analysis: a review of reporting and methods practice. *Environ Int*. 92–93:647–56. doi: 10.1016/j.envint.2016.02.016.

Steenland K, Schubauer-Berigan MK, Vermeulen R, Lunn RM, Straif K, Zahm S et al. (2020). Risk of bias assessments and evidence syntheses for observational epidemiologic studies of environmental and occupational exposures: strengths and limitations. *Environ Health Perspect*. 128(9):095002. doi: 10.1289/EHP6980.

Straif K, Cohen A, Samet J, editors (2013). Air pollution and cancer. Lyon: International Agency for Research on Cancer (IARC Scientific Publications; 161; <https://publications.iarc.fr/Book-And-Report-Series/Iarc-Scientific-Publications/Air-Pollution-And-Cancer-2013>, accessed July).

Tanaka TY, Chiba M (2006). A numerical study of the contributions of dust source regions to the global dust budget. *Glob Planet Change*. 52(1–4):88–104. doi: <https://doi.org/10.1016/j.gloplacha.2006.02.002>.

Thinking Outside the Box team (2019). Ambient ultrafine particles: evidence for policy makers. Pfnztal: European Federation of Clean Air and Environmental Protection Associations (White paper; [https://efca.net/files/WHITE%20PAPER-UFP%20evidence%20for%20policy%20makers%20\(25%20OCT\).pdf](https://efca.net/files/WHITE%20PAPER-UFP%20evidence%20for%20policy%20makers%20(25%20OCT).pdf), accessed 7 August 2020).

Thurston GD, Ahn J, Cromar KR, Shao Y, Reynolds HR, Jerrett M et al. (2016a). Ambient particulate matter air pollution exposure and mortality in the NIH-AARP diet and health cohort. *Environ Health Perspect*. 124(4):484–90. doi: 10.1289/ehp.1509676.

Thurston GD, Burnett RT, Turner MC, Shi Y, Krewski D, Lall R et al. (2016b). Ischemic heart disease mortality and long-term exposure to source-related components of US fine particle air pollution. *Environ Health Perspect*. 124(6):785–94. doi: 10.1289/ehp.1509777.

Thurston GD, Kipen H, Annesi-Maesano I, Balmes J, Brook RD, Cromar K et al. (2017). A joint ERS/ATS policy statement: what constitutes an adverse health effect of air pollution? An analytical framework. *Eur Respir J*. 49(1):1600419. doi: 10.1183/13993003.00419-2016.

Tobías A, Stafoggia M (2020). Modeling desert dust exposures in epidemiologic short-term health effects studies. *Epidemiology*. 31:788–95. doi: 10.1097/EDE.0000000000001255.

Tobias A, Karanasiou A, Amato F, Roqué M, Querol X (2019a). Health effects of desert dust and sand storms: a systematic review and meta-analysis protocol. *BMJ Open*. 9(7):e029876. doi: 10.1136/bmjopen-2019-029876.

Tobias A, Karanasiou A, Amato F, Querol X (2019b). Health effects of desert dust and sand storms: a systematic review and meta-analysis. *Environ Epidemiol*. 3:396. doi: 10.1097/01.EE9.0000610424.75648.58.

Tonne C, Wilkinson P (2013). Long-term exposure to air pollution is associated with survival following acute coronary syndrome. *Eur Heart J*. 34(17):1306–11. doi: 10.1093/eurheartj/ehs480.

Tsai SS, Chen PS, Yang YH, Liou SH, Wu TN, Sung FC et al. (2012). Air pollution and hospital admissions for myocardial infarction: are there potentially sensitive groups? *J Toxicol Environ Health A*. 75(4):242–51. doi: 10.1080/15287394.2012.641202.

Tseng E, Ho WC, Lin MH, Cheng TJ, Chen PC, Lin HH (2015). Chronic exposure to particulate matter and risk of cardiovascular mortality: cohort study from Taiwan. *BMC Public Health*. 15:936. doi: 10.1186/s12889-015-2272-6.

Turner MC, Cohen A, Jerrett M, Gapstur SM, Diver WR, Pope CA III et al. (2014). Interactions between cigarette smoking and fine particulate matter in the risk of lung cancer mortality in Cancer Prevention Study II. *Am J Epidemiol*. 180(12):1145–9. doi: 10.1093/aje/kwu275.

Turner MC, Jerrett M, Pope CA III, Krewski D, Gapstur SM, Diver WR et al. (2016). Long-term ozone exposure and mortality in a large prospective study. *Am J Respir Crit Care Med*. 193(10):1134–42. doi: 10.1164/rccm.201508-1633OC.

Ueda K, Nagasawa S-Y, Nitta H, Miura K, Ueshima H (2012). Exposure to particulate matter and long-term risk of cardiovascular mortality in Japan: NIPPON DATA80. *J Atheroscler Thromb*. 19(3):246–54. doi: 10.5551/jat.9506.

UN (1948). Universal declaration of human rights. New York (NY): United Nations (<https://www.un.org/en/about-us/universal-declaration-of-human-rights>, accessed 9 December 2020).

UN (2015). Draft outcome document of the United Nations summit for the adoption of the post-2015 development agenda. New York (NY): United Nations (United Nations General Assembly draft resolution A/69/L.85; <https://digitallibrary.un.org/record/800852>, accessed 9 December 2020).

UN (2016). Combating sand and dust storms. New York (NY): United Nations (United Nations General Assembly resolution A/RES/70/195; <https://undocs.org/en/A/RES/70/195>, accessed 9 December 2020).

UN (2017). Combating sand and dust storms. New York (NY): United Nations (United Nations General Assembly resolution A/RES/71/219 <https://undocs.org/pdf?symbol=en/a/res/71/219>, accessed 9 December 2020).

UN (2018a). Political declaration of the third high-level meeting of the General Assembly on the prevention and control of non-communicable diseases. New York (NY): United Nations (United Nations General Assembly resolution A/RES/73/2; https://www.un.org/en/ga/search/view_doc.asp?symbol=A/RES/73/2, accessed 10 July 2021).

UN (2018b). Combating sand and dust storms. New York (NY): United Nations (United Nations General Assembly resolution A/RES/72/225; <https://undocs.org/pdf?symbol=en/A/RES/72/225>, accessed 9 December 2020).

UN (2019a) Issue of human rights obligations relating to the enjoyment of a safe, clean, healthy and sustainable environment. New York (NY): United Nations (A/HRC/40/55; <https://undocs.org/en/A/HRC/40/55>, accessed 20 February 2021).

UN (2019b). Combating sand and dust storms. New York (NY): United Nations (United Nations General Assembly resolution A/RES/73/237; <https://undocs.org/en/A/RES/73/237>, accessed 9 December 2020).

UN Statistics Division (2020). Global indicator framework for the Sustainable Development Goals and targets of the 2030 Agenda for Sustainable Development. New York (NY): UN Statistics Division; 2020 (A/RES/71/313, E/CN.3/2018/2, E/CN.3/2019/2, E/CN.3/2020/2; https://unstats.un.org/sdgs/indicators/Global%20Indicator%20Framework%20after%202020%20review_Eng.pdf, accessed 12 July 2021).

UNECE (2011). Strengthening cooperation with regional air pollution networks and initiatives outside the convention. Geneva: United Nations Economic Commission for Europe (Informal document No. 12; https://www.unece.org/fileadmin/DAM/env/documents/2011/eb/eb_n_12.pdf, accessed 9 December 2020).

UNEP (2014). Strengthening the role of the United Nations Environment Programme in promoting air quality. Nairobi: United Nations Environment Programme (United Nations Environment Assembly resolution 1/7; <https://www.informea.org/en/decision/strengthening-role-united-nations-environment-programme-promoting-air-quality>, accessed 9 December 2020).

UNEP (2016a). Sand and dust storms. Nairobi: United Nations Environment Programme (United Nations Environment Assembly resolution UNEP/EA.2/Res.21; <https://wedocs.unep.org/handle/20.500.11822/11194?show=full>, accessed 9 December 2020).

UNEP (2016b). Sand and dust storms. Nairobi: United Nations Environment Programme (UNEA-2 Factsheet; <https://wedocs.unep.org/bitstream/handle/20.500.11822/7608/sand.pdf?sequence=3&amp%3BisAllowed=>, accessed 9 December 2020).

UNEP (2018). Preventing and reducing air pollution to improve air quality globally. Nairobi: United Nations Environment Programme (United Nations Environment Assembly resolution UNEP/EA.3/Res.8; <https://wedocs.unep.org/handle/20.500.11822/31023>, accessed 9 December 2020).

UNEP (2020). Used vehicles and the environment. A global overview of used light duty vehicles: flow, scale and regulation. Nairobi: United Nations Environment Programme (<https://www.unep.org/resources/report/global-trade-used-vehicles-report>, accessed 18 November 2020).

UNEP, WMO, UNCCD (2016). Global assessment of sand and dust storms. Nairobi: United Nations Environment Programme (https://uneplive.unep.org/redesign/media/docs/assessments/global_assessment_of_sand_and_dust_storms.pdf, accessed 9 December 2020).

US EPA (1993). Air quality criteria for oxides of nitrogen. Research Triangle Park (NC): United States Environmental Protection Agency (EPA/600/8-91/049aF-cF. 3v).

US EPA (1995). Air quality criteria for ozone and related photochemical oxidants. Research Triangle Park (NC): United States Environmental Protection Agency (EPA/600/P-93/004aF-cF.3v).

US EPA (2009). Integrated Science Assessment (ISA) for particulate matter (final report, Dec 2009). Washington (DC): United States Environmental Protection Agency (EPA/600/R-08/139F; <https://cfpub.epa.gov/ncea/isa/recordisplay.cfm?deid=216546>, accessed 9 December 2020).

US EPA (2010). Integrated Science Assessment (ISA) for carbon monoxide. Washington (DC): United States Environmental Protection Agency (EPA/600/R-09/019F; <https://cfpub.epa.gov/ncea/risk/recordisplay.cfm?deid=218686>, accessed 5 December 2020).

US EPA (2013). Integrated Science Assessment (ISA) for ozone and related photochemical oxidants (final report, February 2013). Washington (DC): United States Environmental Protection Agency (EPA/600/R-10/076F; <https://cfpub.epa.gov/ncea/isa/recordisplay.cfm?deid=247492>, accessed 9 December 2020).

US EPA (2015). Integrated Science Assessment (ISA) for sulfur oxides – health criteria (first external review draft, November 2015). Washington (DC): United States Environmental Protection Agency (EPA/600/R-15/066; <https://cfpub.epa.gov/ncea/isa/recordisplay.cfm?deid=310044>, accessed 9 December 2020).

US EPA (2016). Integrated Science Assessment (ISA) for oxides of nitrogen – health criteria (final report, January 2016). Washington (DC): United States Environmental Protection Agency (EPA/600/R-15/068; <https://cfpub.epa.gov/ncea/isa/recordisplay.cfm?deid=310879>, accessed 9 December 2020).

US EPA (2017). Integrated Science Assessment (ISA) for sulfur oxides – health criteria. Washington (DC): United States Environmental Protection Agency (EPA/600/R-17/451; <https://cfpub.epa.gov/ncea/isa/recordisplay.cfm?deid=338596>, accessed 5 December 2020).

US EPA (2019a). Integrated Science Assessment (ISA) for particulate matter (final report, December 2019). Washington (DC): United States Environment Protection Agency (EPA/600/R-19/188; <https://www.epa.gov/isa/integrated-science-assessment-isa-particulate-matter>, accessed 21 February 2021).

US EPA (2019b). AP-42: Compilation of air emissions factors. In: Air emissions factors and quantification [website]. Washington (DC): United States Environment Protection Agency (<https://www.epa.gov/air-emissions-factors-and-quantification/ap-42-compilation-air-emissions-factors>, accessed 7 August 2020).

US EPA (2020). Integrated Science Assessment (ISA) for ozone and related photochemical oxidants. Washington (DC): United States Environmental Protection Agency (<https://www.epa.gov/isa/integrated-science-assessment-isa-ozone-and-related-photochemical-oxidants>, accessed 28 October 2020).

US EPA (2021). Environmental Benefits Mapping and Analysis Program – Community Edition (BenMAP-CE) [website]. Washington (DC): United States Environmental Protection Agency (<https://www.epa.gov/benmap>, accessed 20 February 2021).

Vahlsing C, Smith KR (2012). Global review of national ambient air quality standards for PM(10) and SO(2) (24 h). *Air Qual Atmos Health*. 5(4):393–9. doi: 10.1007/s11869-010-0131-2.

van Erp AM, O’Keefe R, Cohen AJ, Warren J (2008). Evaluating the effectiveness of air quality interventions. *J of Toxicol Environ Health. Part A.* 71(9–10):583–7. doi: 10.1080/15287390801997708.

Vedal S, Campen MJ, McDonald JD, Larson TV, Sampson PD, Sheppard L et al. (2013). National Particle Component Toxicity (NPACT) initiative report on cardiovascular effects. *Res Rep Health Eff Inst.* (178):5–8. PMID: 24377210.

Vicedo-Cabrera AM, Sera F, Liu C, Armstrong B, Milojevic A, Guo Y et al. (2020). Short term association between ozone and mortality: global two stage time series study in 406 locations in 20 countries. *BMJ.* 368:m108. doi: 10.1136/bmj.m108. License: [CC BY 4.0](#).

Villeneuve PJ, Weichenthal SA, Crouse D, Miller AB, To T, Martin RV et al. (2015). Long-term exposure to fine particulate matter air pollution and mortality among Canadian women. *Epidemiology.* 26(4):536–45. doi: 10.1097/EDE.0000000000000294. Copyright © 2015 Wolters Kluwer Health, Inc. All rights reserved (https://journals.lww.com/epidem/Fulltext/2015/07000/Long_term_Exposure_to_Fine_Part particulate_Matter_Air.14.aspx, accessed 11 July 2021).

Vohra K, Vodonos A, Schwartz J, Marais EA, Sulprizio MP, Mickley LJ (2021). Global mortality from outdoor fine particle pollution generated by fossil fuel combustion: results from GEOS-Chem. *Environ Res.* 195:110754. doi: 10.1016/j.envres.2021.110754.

Vrijheid M, Fossati S, Maitre L, Marquez S, Roumeliotaki T, Agier L et al. (2020). Early-life environmental exposures and childhood obesity: an exposome-wide approach. *Environ Health Perspect.* 128(6):67009. doi: 10.1289/EHP5975.

Weichenthal S, Pinault LL, Burnett RT (2017). Impact of oxidant gases on the relationship between outdoor fine particulate air pollution and nonaccidental, cardiovascular, and respiratory mortality. *Sci Rep.* 7(1):16401. doi: 10.1038/s41598-017-16770-y.

Weichenthal S, Villeneuve PJ, Burnett RT, van Donkelaar A, Martin RV, Jones RR et al. (2014). Long-term exposure to fine particulate matter: association with nonaccidental and cardiovascular mortality in the agricultural health study cohort. *Environ Health Perspect.* 122(6):609–15. doi: 10.1289/ehp.1307277.

Whaley P, Nieuwenhuijsen M, Burns J, editors (2021). Update of the WHO global air quality guidelines: systematic reviews. *Environ Int.* 142(Special issue) (<https://www.sciencedirect.com/journal/environment-international/special-issue/10MTC4W8FXJ>, accessed 17 June 2021).

WHO (2014a). WHO handbook for guideline development, 2nd edition. Geneva: World Health Organization (<https://apps.who.int/iris/handle/10665/145714>, accessed 4 November 2020).

WHO (2014b). WHO guidelines for indoor air quality: household fuel combustion. Geneva: World Health Organization (<https://apps.who.int/iris/handle/10665/141496>, accessed 9 December 2020).

WHO (2014c). Strong recommendations when the evidence is low quality. In: WHO handbook for guideline development, 2nd edition. Geneva: World Health Organization:169–82 (https://apps.who.int/iris/bitstream/handle/10665/145714/9789241548960_chap14_eng.pdf?sequence=5&isAllowed=y, accessed 6 August 2020).

WHO (2014d). Health in All Policies: Helsinki statement. Framework for country action. Geneva: World Health Organization; 2014 (<https://apps.who.int/iris/handle/10665/112636>, accessed 5 March 2021).

WHO (2015). Resolution WHA68.8. Health and the environment: addressing the health impact of air pollution. Geneva: World Health Organization (<https://apps.who.int/iris/handle/10665/253237>, accessed 19 November 2020).

WHO (2016a). Health and the environment: road map for an enhanced global response to the adverse health effects of air pollution. In: Sixty-ninth World Health Assembly, Geneva, 23–28 May 2016. Resolutions and decisions, annexes. Geneva: World Health Organization (<https://apps.who.int/iris/handle/10665/259134>, accessed 6 March 2021).

WHO (2016b). International Statistical Classification of Diseases and Related Health Problems, tenth revision, 2016 version [website]. Geneva: World Health Organization; 2016 (<https://icd.who.int/browse10/2016/en>, accessed 7 March 2021).

WHO (2017). Communicating risk in public health emergencies: a WHO guideline for emergency risk communication (ERC) policy and practice. Geneva: World Health Organization (<https://apps.who.int/iris/handle/10665/259807>, accessed 3 March 2021).

WHO (2018). Burden of disease from ambient air pollution for 2016, version 2. April 2018. Geneva: World Health Organization (https://www.who.int/airpollution/data/AAP_BoD_results_May2018_final.pdf?ua=1, accessed 17 March 2021).

WHO (2019a). Compendium of WHO malaria guidance: prevention, diagnosis, treatment, surveillance and elimination. Geneva: World Health Organization (<https://apps.who.int/iris/handle/10665/312082>, accessed 10 December 2020).

WHO (2019b). WHO report on the global tobacco epidemic 2019: offer help to quit tobacco use. Geneva: World Health Organization (<https://apps.who.int/iris/handle/10665/326043>, accessed 17 November 2020).

WHO (2020a). Personal interventions and risk communication on air pollution. Geneva: World Health Organization (<https://apps.who.int/iris/handle/10665/333781>, accessed 21 June 2021).

WHO (2020b). Health equity. In: Social determinants of health [website]. Geneva: World Health Organization (https://www.who.int/health-topics/social-determinants-of-health#tab=tab_3, accessed 17 June 2021).

WHO (2020c). Urban Health Initiative [website]. Geneva: World Health Organization; 2020 (<https://www.who.int/initiatives/urban-health-initiative>, accessed 28 June 2021).

WHO (2020d). Call for experts: Global Air Pollution and Health – Technical Advisory Group [website]. Geneva: World Health Organization (<https://www.who.int/news-room/articles-detail/call-for-experts-global-air-pollution-and-health-technical-advisory-group>, accessed 11 July 2021).

WHO (2021a). WHO Global Ambient Air Quality Database [online database]. Geneva: World Health Organization (<https://www.who.int/data/gho/data/themes/air-pollution/who-air-quality-database>, accessed 10 September 2021).

WHO (2021b). National air quality standards [website]. Geneva: World Health Organization (<https://www.who.int/tools/air-quality-standards>, accessed 10 July 2021).

WHO, CCAC, UNEP (2018). Breathelife campaign [website]. Geneva: World Health Organization (<https://breathelife2030.org/>, accessed 27 June 2021).

WHO Global Air Quality Guidelines Working Group on Certainty of Evidence Assessment (2020). Approach to assessing the certainty of evidence from systematic reviews informing WHO global air quality guidelines. Amsterdam: Elsevier (<https://ars.els-cdn.com/content/image/1-s2.0-S0160412020318316-mmc4.pdf>, accessed 21 July 2020).

WHO Regional Office for Europe (1987). Air quality guidelines for Europe. Copenhagen: WHO Regional Office for Europe (<https://apps.who.int/iris/handle/10665/107364>, accessed 25 June 2021).

WHO Regional Office for Europe (1998). Guidance for setting air quality standards. Report on a WHO Working Group. Copenhagen: WHO Regional Office for Europe (EUR/ICP/EHPM 02 01 02; <https://apps.who.int/iris/handle/10665/108084>, accessed 24 June 2021).

WHO Regional Office for Europe (1999). Implementing national environmental health action plans in partnership: Third Ministerial Conference on Environment and Health, London, 16–18 June 1999. Copenhagen: WHO Regional Office for Europe (<https://apps.who.int/iris/handle/10665/108214>, accessed 7 December 2020).

WHO Regional Office for Europe (2000a). Air quality guidelines for Europe, 2nd edition. Copenhagen: WHO Regional Office for Europe (<https://apps.who.int/iris/handle/10665/107335>, accessed 24 June 2021).

WHO Regional Office for Europe (2000b). Use of the guidelines in protecting public health. In: Air quality guidelines, 2nd edition. Copenhagen: WHO Regional Office for Europe:41–55. (<https://apps.who.int/iris/handle/10665/107335>, accessed 9 December 2020).

WHO Regional Office for Europe (2006). Air quality guidelines – global update 2005. Particulate matter, ozone, nitrogen dioxide and sulfur dioxide. Copenhagen: WHO Regional Office for Europe (<https://apps.who.int/iris/handle/10665/107823>, accessed 9 December 2020).

WHO Regional Office for Europe (2009). WHO guidelines for indoor air quality: dampness and mould. Copenhagen: WHO Regional Office for Europe (<https://apps.who.int/iris/handle/10665/164348>, accessed 17 June 2021).

WHO Regional Office for Europe (2010). WHO guidelines for indoor air quality: selected pollutants. Copenhagen: WHO Regional Office for Europe (<https://apps.who.int/iris/handle/10665/260127>, accessed 24 June 2021).

WHO Regional Office for Europe (2013a). Review of evidence on health aspects of air pollution – REVIHAAP project. Copenhagen: WHO Regional Office for Europe (Technical report; <https://apps.who.int/iris/handle/10665/341712>, accessed 17 June 2021).

WHO Regional Office for Europe (2013b). Health and environment: communicating the risks. Copenhagen: WHO Regional Office for Europe (<https://apps.who.int/iris/handle/10665/108629>, accessed 18 February 2021).

WHO Regional Office for Europe (2016a). WHO expert consultation: available evidence for the future update of the WHO global air quality guidelines (AQGs). Copenhagen: WHO Regional Office for Europe (<https://apps.who.int/iris/handle/10665/341714>, accessed 17 June 2021).

WHO Regional Office for Europe (2016b). Health risk assessment of air pollution: general principles. Copenhagen: WHO Regional Office for Europe (<https://apps.who.int/iris/handle/10665/329677>, accessed 9 December 2020).

WHO Regional Office for Europe (2017). Evolution of WHO air quality guidelines: past, present and future. Copenhagen: WHO Regional Office for Europe (<https://apps.who.int/iris/handle/10665/341912>, accessed 10 July 2021).

WHO Regional Office for Europe (2018). Multisectoral and intersectoral action for improved health and well-being for all: mapping of the WHO European Region. Governance for a sustainable future: improving health and well-being for all. Final report. Copenhagen: WHO Regional Office for Europe (<https://apps.who.int/iris/handle/10665/341715>, accessed 17 June 2021).

WHO Regional Office for Europe (2020). Risk of bias assessment instrument for systematic reviews informing WHO global air quality guidelines. Copenhagen: WHO Regional Office for Europe (<https://apps.who.int/iris/handle/10665/341717>, accessed 17 June 2021).

WHO Regional Office for Europe (2021a). AirQ+: software tool for health risk assessment of air pollution [website]. Copenhagen: WHO Regional Office for Europe (<https://www.euro.who.int/en/health-topics/environment-and-health/air-quality/activities/airq-software-tool-for-health-risk-assessment-of-air-pollution>, accessed 20 February 2021).

WHO Regional Office for Europe (2021b). Health aspects of long-range transboundary air pollution [website]. Copenhagen: WHO Regional Office for Europe (<https://www.euro.who.int/en/health-topics/environment-and-health/air-quality/activities/health-aspects-of-long-range-transboundary-air-pollution>, accessed 10 July 2021).

WMO (2020a). Sand and dust storms [website]. Geneva: World Meteorological Organization (<https://public.wmo.int/en/our-mandate/focus-areas/environment/SDS>, accessed 10 December 2020).

WMO (2020b). Dust storm or sandstorm [website]. Geneva: World Meteorological Organization (<https://cloudatlas.wmo.int/en/dust-storm-or-sandstorm.html>, accessed 10 December 2020).

WMO (2020c). WMO Sand and Dust Storm Warning Advisory and Assessment System (SDS-WAS) [website]. Geneva: World Meteorological Organization (<https://community.wmo.int/wwrp-gaw-sds>, accessed 10 December 2020).

Wong CM, Vichit-Vadakan N, Kan H, Qian Z (2008). Public Health and Air Pollution in Asia (PAPA): a multicity study of short-term effects of air pollution on mortality. *Environ Health Perspect.* 116(9):1195–202. doi: 10.1289/ehp.11257.

Woodruff TJ, Sutton P (2011). An evidence-based medicine methodology to bridge the gap between clinical and environmental health sciences. *Health Aff (Millwood)*. 30(5):931–7. doi: 10.1377/hlthaff.2010.1219.

World Bank (2014). Reducing black carbon emissions from diesel vehicles: impacts, control strategies, and cost benefit analysis. Washington (DC): World Bank (<http://documents1.worldbank.org/curated/en/329901468151500078/pdf/864850WP00PUBLOI0report002April2014.pdf>, accessed 18 February 2021).

World Bank (2016). The cost of air pollution: strengthening the economic case for action. Washington (DC): World Bank (<https://documents.worldbank.org/en/publication/documents-reports/documentdetail/781521473177013155/the-cost-of-air-pollution-strengthening-the-economic-case-for-action>, accessed 17 March 2021).

World Bank (2019). Sand and dust storms in the Middle East and North Africa Region: sources, costs, and solutions. Washington (DC): World Bank (<http://documents1.worldbank.org/curated/en/483941576489819272/pdf/Sand-and-Dust-Storms-in-the-Middle-East-and-North-Africa-MENA-Region-Sources-Costs-and-Solutions.pdf>, accessed 13 March 2021).

World Bank (2021). The world by income and region [website]. Washington (DC): World Bank (<https://datatopics.worldbank.org/world-development-indicators/the-world-by-income-and-region.html>, accessed 7 July 2021).

Yang B-Y, Fan S, Thiering E, Seissler J, Nowak D, Dong G-H et al. (2020). Ambient air pollution and diabetes: a systematic review and meta-analysis. *Environ Res.* 180:108817. doi: 10.1016/j.envres.2019.108817.

Yang J, McBride J, Zhou J, Sun Z (2005). The urban forest in Beijing and its role in air pollution reduction. *Urban For Urban Green.* 3(2):65–78. doi: <https://doi.org/10.1016/j.ufug.2004.09.001>.

Yang J, Zhou M, Li M, Yin P, Hu J, Zhang C et al. (2020). Fine particulate matter constituents and cause-specific mortality in China: a nationwide modelling study. *Environ Int.* 143:105927. doi: 10.1016/j.envint.2020.105927.

Yang X, Liang F, Li J, Chen J, Liu F, Huang K et al. (2020). Associations of long-term exposure to ambient PM_{2.5} with mortality in Chinese adults: a pooled analysis of cohorts in the China-PAR project. *Environ Int.* 138:105589. doi: 10.1016/j.envint.2020.105589.

Yang Y, Tang R, Qiu H, Lai P-C, Wong P, Thach T-Q et al. (2018). Long term exposure to air pollution and mortality in an elderly cohort in Hong Kong. *Environ Int.* 117:99–106. doi: 10.1016/j.envint.2018.04.034.

Yin P, Brauer M, Cohen A, Burnett RT, Liu J, Liu Y et al. (2017). Long-term fine particulate matter exposure and nonaccidental and cause-specific mortality in a large national cohort of Chinese men. *Environ Health Perspect.* 125(11):117002. doi: 10.1289/EHP1673.

Yorifuji T, Kashima S, Tsuda T, Ishikawa-Takata K, Ohta T, Tsuruta K et al. (2013). Long-term exposure to traffic-related air pollution and the risk of death from hemorrhagic stroke and lung cancer in Shizuoka, Japan. *Sci Total Environ.* 443:397–402. doi: 10.1016/j.scitotenv.2012.10.088.

Zhang Q, Jiang X, Tong D, Davis SJ, Zhao H, Geng G et al. (2017). Transboundary health impacts of transported global air pollution and international trade. *Nature.* 543(7647):705–9. doi: 10.1038/nature21712.

Zhang X, Zhao L, Tong DQ, Wu G, Dan M, Teng B (2016). A systematic review of global desert dust and associated human health effects. *Atmosphere.* 7(12):158. doi: <https://doi.org/10.3390/atmos7120158>.

Zheng X-y, Orellano P, Lin H-I, Jiang M, Guan W-j (2021). Short-term exposure to ozone, nitrogen dioxide, and sulphur dioxide and emergency room visits and hospital admissions due to asthma: a systematic review and meta-analysis. *Environ Int.* 150:106435. doi: 10.1016/j.envint.2021.106435. License: [CC BY-NC-ND](https://creativecommons.org/licenses/by-nc-nd/4.0/).

Zhou M, Liu Y, Wang L, Kuang X, Xu X, Kan H (2014). Particulate air pollution and mortality in a cohort of Chinese men. *Environ Pollut.* 186:1–6. doi: 10.1016/j.envpol.2013.11.010.

Annex 1. Groups engaged during the development of the guidelines

Tables A1.1–A1.7 give details of the various teams involved in the development of the guidelines at various stages.

Table A1.1. WHO steering group

Name (membership period)	Position	Department
	Technical Officer	WHO headquarters, Geneva, Switzerland
	Scientist, Protection of the Human Environment	WHO Regional Office for Africa, Brazzaville, Congo
	Coordinator, Interventions for Healthy Environments	WHO headquarters, Geneva, Switzerland
	Technical Officer	WHO headquarters, Geneva, Switzerland
	Team Leader, Environmental Health	WHO Regional Office for the Western Pacific, Manila, Philippines
	Technical Officer, Air Quality And Noise	WHO Regional Office for Europe, European Centre for Environment and Health, Bonn, Germany
	Programme Manager, Living and Working Environments	WHO Regional Office for Europe, European Centre for Environment and Health, Bonn, Germany
	Coordinator, Health and the Environment	WHO Regional Office for the Western Pacific, Manila, Philippines
	Head, Monographs Group	International Agency for Research on Cancer, Lyon, France

Table A1.1 contd

Name (membership period)	Position	Department
	Regional Adviser, Environmental Health Exposures	WHO Regional Office for the Eastern Mediterranean, Regional Centre for Environmental Health Action, Amman, Jordan
	Technical Officer, Climate Change, Health and Environment	WHO Regional Office for Africa, Brazzaville, Congo
	Technical Officer	WHO Regional Office for Europe, European Centre for Environment and Health, Bonn, Germany
	Regional Adviser, Occupational and Environmental Epidemiology	WHO Regional Office for South-East Asia, New Delhi, India
	Head of Office	WHO Regional Office for Europe, European Centre for Environment and Health, Bonn, Germany
	Technical Officer, Environment and Health	WHO Regional Office for Europe, European Centre for Environment and Health, Bonn, Germany
	Coordinator, Health and the Environment	WHO Regional Office for the Western Pacific, Manila, Philippines
	Unit Head, Air Quality and Health	WHO headquarters, Geneva, Switzerland
	Adviser, Environmental Epidemiology	WHO Regional Office for the Americas, Washington, DC, the United States
	Consultant; Scientist	WHO Regional Office for Europe, European Centre for Environment and Health, Bonn, Germany (2016); International Agency for Research on Cancer, Lyon, France (2018)
	Technical Officer, Air Quality	WHO Regional Office for Europe, European Centre for Environment and Health, Bonn, Germany

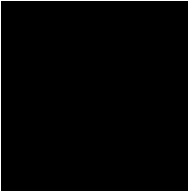
Table A1.2. Guideline development group

Name (membership period)	Position and affiliation	Sex	Area of expertise specifically sought for guidelines^a
	Chief Environmental Consultant, Kuwait Environment Public Authority, Safat, Kuwait	M	3, 6
	Head, Environmental Quality Department, Environmental Protection Agency, Accra, Ghana	M	5, 6
	Associate Dean (Research) and Director, WHO Collaborating Centre for Occupational and Environmental Health, Sri Ramachandra University, Chennai, India	F	5, 6
	Professor, School of Population and Public Health, University of British Columbia, Vancouver, BC, Canada	M	1, 3
	Professor Emeritus, Institute for Risk Assessment Sciences, Utrecht University, Utrecht, the Netherlands	M	1, 7
	Consulting Principal Scientist, Health Effects Institute, Boston, MA, the United States	M	1, 7
	Senior Researcher, Institute for Biomedical Research and Innovation, National Research Council (CNR-IRIB), Palermo, Italy	M	1, 2
	China Director, Clean Air Asia, Beijing, China	F	4–6
	Director, Urban Emissions, Goa, India	M	1, 3
	Associate Professor, Institute for Environment Research, Tehran University of Medical Sciences, Tehran, Iran	M	1, 3
	Head, Air Quality Assessment Section, Health Canada, Ottawa, ON, Canada	F	1, 6, 7
	Professor, School of Public Health, Peking University, Beijing, China	F	2, 6

Table A1.2 contd

Name (membership period)	Position and affiliation	Sex	Area of expertise specifically sought for guidelines^a
	Professor and Director, School of Public Health, Fudan University, Shanghai, China	M	1, 5
	Professor, Environmental Engineering and Management, Asian Institute of Technology, Pathumthani, Thailand	F	3
	Visiting Professor, School of Public Health, Imperial College London, London, England, the United Kingdom	M	1, 6, 7
	Professor and Unit Head, Education and Training, Swiss Tropical and Public Health Institute (Swiss TPH) and University of Basel, Basel, Switzerland	M	1
	Senior Epidemiologist, United States Environmental Protection Agency, Research Triangle Park, NC, the United States	M	1, 7
	Distinguished Professor and Director, International Laboratory for Air Quality and Health, Queensland University of Technology, Brisbane, QLD, Australia	F	3, 7
	Head of Programs, Clean Air Asia, Manila, Philippines	F	5, 6
	Director, European Lung Foundation, Sheffield, England, the United Kingdom	F	5
	Environmental Health Director, National Institute of Public Health (INSP), Cuernavaca, Mexico	M	1, 3–5
	Dean and Professor, Colorado School of Public Health, Aurora, CO, the United States	M	1, 6
	Professor, School of Public Health, Imperial College London, London, England, the United Kingdom	M	3, 6
	Senior Specialist Scientist, Environment and Health Research Unit, South African Medical Research Council (SAMRC), Pretoria, South Africa	F	3

Table A1.2 contd

Name (membership period)	Position and affiliation	Sex	Area of expertise specifically sought for guidelines ^a
	Professor, Peking Union Medical College, School of Basic Medicine, Beijing, China	F	1
	Policy Officer, Industrial Emissions, Air Quality & Noise Unit, Directorate-General for Environment, European Commission, Brussels, Belgium	M	6

^a Area of expertise/interest: 1. Health effects of air pollution – epidemiological evidence and/or risk assessment; 2. Health effects of air pollution – toxicological and clinical evidence; 3. Air pollution emissions and atmospheric chemistry/exposure assessment; 4. Best practices, interventions and/or health economics; 5. Vulnerable groups, equity, human rights, gender and/or developing country perspective; 6. End-user perspective, policy implications, implementation of the guidelines; 7. Methodology and guideline development.

^b Deceased 21 September 2020.

Table A1.3. Systematic review team^a

Systematic review topic	Experts involved	Affiliation
Long-term exposure to PM _{2.5} and PM ₁₀		Institute for Risk Assessment Sciences, Utrecht University, Utrecht, the Netherlands
All-cause and cause-specific mortality		Institute for Risk Assessment Sciences, Utrecht University, Utrecht, the Netherlands
Long-term exposure to O ₃ and NO ₂		Population Health Research Institute, St George's, University of London, London, England, the United Kingdom
All-cause and respiratory mortality		Population Health Research Institute, St George's, University of London, London, England, the United Kingdom
Short-term exposure to PM _{2.5} , PM ₁₀ , O ₃ and NO ₂		Institute for Clinical Effectiveness and Health Policy (IECS-CONICET), Buenos Aires, Argentina
All-cause and cause-specific mortality		Institute for Clinical Effectiveness and Health Policy (IECS-CONICET), Buenos Aires, Argentina
		Centre for Research and Technology Transfer San Nicolás, National Technology University (UTN-CONICET), San Nicolás, Argentina
		San Nicolás Regional Faculty, National Technology University (UTN), San Nicolás, Argentina
		Scientific Research Commission of the Province of Buenos Aires, La Plata, Argentina
		San Felipe General Hospital, San Nicolás, Argentina
Short-term exposure to SO ₂		Centre for Research and Technology Transfer San Nicolás, National Technology University (UTN-CONICET), San Nicolás, Argentina
All-cause and respiratory mortality		San Nicolás Regional Faculty, National Technology University (UTN), San Nicolás, Argentina
		Scientific Research Commission of the Province of Buenos Aires, La Plata, Argentina
		San Felipe General Hospital, San Nicolás, Argentina

Table A1.3 contd

Systematic review topic	Experts involved	Affiliation
Short-term exposure to O ₃ , NO ₂ and SO ₂ Emergency department visits and hospital admissions due to asthma		State Key Laboratory of Respiratory Disease, National Clinical Research Center for Respiratory Disease, Guangzhou Institute of Respiratory Health, First Affiliated Hospital of Guangzhou Medical University, Guangzhou Medical University, Guangzhou, China
		State Key Laboratory of Respiratory Disease, National Clinical Research Center for Respiratory Disease, Guangzhou Institute of Respiratory Health, First Affiliated Hospital of Guangzhou Medical University, Guangzhou Medical University, Guangzhou, China
		Sun Yat-sen University, Guangzhou, China
		Centre for Research and Technology Transfer San Nicolás, National Technology University (UTN-CONICET), San Nicolás, Argentina
		Institute of Non-communicable Disease Control and Prevention, Guangdong Provincial Center for Disease Control And Prevention, Guangdong, China
Short-term exposure to CO Myocardial infarction		British Heart Foundation Centre for Cardiovascular Science, University of Edinburgh, Edinburgh, Scotland, the United Kingdom
		British Heart Foundation Centre for Cardiovascular Science, University of Edinburgh, Edinburgh, Scotland, the United Kingdom
		British Heart Foundation Centre for Cardiovascular Science and the Usher Institute of Population Health Sciences and Informatics, University of Edinburgh, Edinburgh, Scotland, the United Kingdom
		British Heart Foundation Centre for Cardiovascular Science and Usher Institute of Population Health Sciences and Informatics, University of Edinburgh, Edinburgh, Scotland, the United Kingdom
		British Heart Foundation Centre for Cardiovascular Science, University of Edinburgh, Edinburgh, Scotland, the United Kingdom

^a Specific contributions are reported in the articles published in the special issue of *Environment International*: Update of the WHO global air quality guidelines: systematic reviews (Whaley et al., 2021; see main reference list).

Table A1.4. External methodologists

Methodological topic	Methodologist (period of service)	Affiliation
Systematic review and guideline development (guideline methodology)		Coordinating Editor, Cochrane Work Review Group, Kuopio, Finland
RoB assessment		Assistant Professor, McMaster University, Hamilton, ON, Canada

Table A1.5. External review group – individual experts

Name	Affiliation	Sex	Area of expertise specifically sought for guidelines ^a
	Public Health and Reforms Centre, Baku, Azerbaijan	M	2, 3, 6
	College of Life Sciences, Kuwait University, Kuwait City, Kuwait	M	3, 6
	Federal Office of the Environment, Bern, Switzerland	M	6
	Institute for Health and Social Policy, McGill University, Montreal, QC, Canada	F	1, 5
	Health Effects Institute, Boston, MA, the United States	F	1, 3
	Faculty of Civil and Environmental Engineering, Technion – Israel Institute of Technology, Haifa, Israel	M	3
	Population Studies Division, Health Canada, Ottawa, ON, Canada	M	1, 6
	Institute for Medical Informatics, Biometry and Epidemiology, Pettenkofer School of Public Health, Ludwig-Maximilians-University, Munich, Germany	M	7
	National Institute for Public Health and the Environment (RIVM), Bilthoven, the Netherlands	M	2

Table A1.5 contd

Name	Affiliation	Sex	Area of expertise specifically sought for guidelines^a
	United States Environmental Protection Agency, Research Triangle Park, NC, the United States	M	1, 7
	School of Public Health (EHESP), Rennes, France	F	1, 5
	Centre for Atmospheric Sciences, Indian Institute of Technology, New Delhi, India	M	3, 5, 6
	School of Public Health, Imperial College London, London, England, the United Kingdom	M	1
	Faculty of Medicine, Pharmacy and Dentistry, Cheikh Anta Diop University (UCAD), Dakar, Senegal	M	1–3
	United States Environmental Protection Agency, Research Triangle Park, NC, the United States	M	1, 4
	School of Public Health, Imperial College London, London, England, the United Kingdom	F	3, 5
	School of Public Health, Imperial College London, London, England, the United Kingdom	F	2
	National Health and Medical Research Council, Canberra, ACT, Australia	F	6, 7
	Finnish Institute for Health and Welfare (THL), Helsinki, Finland	M	1, 2, 6
	Institute for Occupational, Social and Environmental Medicine, Heinrich Heine University of Düsseldorf, Düsseldorf, Germany	F	1, 2
	Econometrics Research and Consulting, Reading, England, the United Kingdom	M	4, 6
	Institute of Environmental Medicine, College of Medicine, Seoul National University, Seoul, Republic of Korea	M	1, 2, 6
	School of Public Health and Community Medicine, University of New South Wales, Kensington, NSW, Australia	M	1, 5

Table A1.5 contd

Name	Affiliation	Sex	Area of expertise specifically sought for guidelines ^a
	Swiss TPH, University of Basel, Basel, Switzerland	F	1, 6
	Department of Health Sciences, California State University, East Bay, Hayward, CA, the United States	F	1, 7
	Nuffield Department of Population Health, University of Oxford, Oxford, England, the United Kingdom	M	1, 4
	Institute of Technology Bandung, Bandung, Indonesia	F	3, 6
	NYU School of Medicine, New York University, New York, NY, the United States	M	1–3
	Public Health France, Saint-Maurice, France	F	1, 6
	School of Nursing and Public Health, University of Kwazulu Natal, Durban, South Africa	M	1, 5, 6
	Barcelona Institute for Global Health (ISGlobal), Barcelona, Spain	M	1, 3
	Department of Environmental Health Science, Soonchunhyang University, Asan, Republic of Korea	F	1, 3
	Cardiology Centre, University of Ferrara, Ferrara, Italy	F	2
	Helmholtz Zentrum München – German Research Center for Environmental Health, Institute of Epidemiology II, Neuherberg, Germany	F	1, 2
	Copernicus Atmosphere Monitoring Service, European Centre for Medium-Range Weather Forecasts, Reading, England, the United Kingdom	M	3, 6
	College of Family, Home, and Social Sciences, Brigham Young University, Provo, UT, the United States	M	1, 4
	School of Public Health, College of Health Sciences, University of Ghana, Legon, Ghana	M	5–7

Table A1.5 contd

Name	Affiliation	Sex	Area of expertise specifically sought for guidelines ^a
	Institute of Environmental Assessment and Water Research (IDAEA), Spanish National Research Council (CSIC), Barcelona, Spain	M	3, 4
	Agency for Food, Environmental and Occupational Health & Safety (ANSES), Maisons-Alfort, France	M	1, 7
	Institute for Medical Informatics, Biometry and Epidemiology, Pettenkofer School of Public Health, Ludwig-Maximilians-University Munich, Munich, Germany	F	6, 7
	Scientific Research Institute for Atmospheric Air Protection (SRI Atmosphere), Saint Petersburg, Russian Federation	M	3, 6
	Centre for Science and Environment (CSE), New Delhi, India	F	4–6
	United States Environmental Protection Agency, Research Triangle Park, NC, the United States	M	1, 7
	Faculty of Medicine, University of São Paulo, São Paulo, Brazil	M	2
	Faculty of Arts and Science, American University of Beirut, Beirut, Lebanon	F	3
	London School of Hygiene and Tropical Medicine, University of London, London, England, the United Kingdom	F	4
	Rollins School of Public Health, Emory University, Atlanta, GA, the United States	M	1, 3
	Radboud Institute for Health Sciences, Nijmegen, the Netherlands	M	2, 7
	Central Department of Statistics, Tribhuvan University, Kirtipur, Kathmandu, Nepal	M	1, 3, 5

Table A1.5 contd

Name	Affiliation	Sex	Area of expertise specifically sought for guidelines ^a
	Proklima International, Latin America and the Caribbean, German Corporation for International Cooperation (GIZ), Eschborn, Germany	F	5, 6
	School of Public Health, University of California Berkeley, Berkeley, CA, the United States	M	1, 4, 5
	Department of Epidemiology, Lazio Region Health Service, Rome, Italy	M	1, 4
	Air Quality Health Effects Research Section, Health Canada, Vancouver BC, Canada	M	1, 2, 7
	Barcelona Institute for Global Health (ISGlobal), Barcelona, Spain	M	1
	Keck School of Medicine, University of Southern California, Los Angeles, CA, the United States	M	1, 7
	NYU School of Medicine, New York University, New York, NY, the United States	M	1
	School of Public Health, The University of Hong Kong, China, Hong Kong SAR	M	1, 2
	Institute of Environmental Assessment and Water Research (IDAEA), Spanish National Research Council (CSIC), Barcelona, Spain	M	1, 4, 7
	European Commission Joint Research Centre, Ispra, Italy	F	3
	National Centre for Epidemiology and Population Health, Australian National University, Canberra ACT, Australia	M	1, 4
	Institute of Biomedicine and Molecular Immunology "Alberto Monroy", National Research Council (CNR-IBIM), Palermo, Italy	M	1, 2

Table A1.5 contd

Name	Affiliation	Sex	Area of expertise specifically sought for guidelines ^a
	School of Health Systems and Public Health, University of Pretoria, Hatfield, South Africa	F	1, 2, 5
	School of Public Health, Imperial College London, London, England, the United Kingdom	F	1, 6
	Lancaster Environment Centre, Lancaster University, Lancaster, England, the United Kingdom	M	7
	Graduate School of Environmental and Life Science, Okayama University, Okayama, Japan	M	1, 2

^a Area of expertise/interest: 1. Health effects of air pollution – epidemiological evidence and/or risk assessment; 2. Health effects of air pollution – toxicological and clinical evidence; 3. Air pollution emissions and atmospheric chemistry/exposure assessment; 4. Best practices, interventions and/or health economics; 5. Vulnerable groups, equity, human rights, gender and/or developing country perspective; 6. End-user perspective, policy implications, implementation of the guidelines; 7. Methodology and guideline development.

^b Deceased 15 June 2020.

Table A1.6. External review group – stakeholder organizations

Organization	Location	Area of expertise specifically sought for guidelines ^a
Abu Dhabi Global Environmental Data Initiative (AGEDI)	Abu Dhabi, United Arab Emirates	2, 3
African Centre for Clean Air (ACCA)	Kampala, Uganda	1, 4
Association for Emissions Control by Catalyst (AECC)	Schaerbeek, Belgium	1, 2, 5
Clean Air Asia (CAA)	Manila, the Philippines	1, 2
ClientEarth	London, England, the United Kingdom	3
Concawe	Brussels, Belgium	2, 4, 6

Table A1.6 contd

Organization	Location	Area of expertise specifically sought for guidelines ^a
European Environment Agency (EEA)	Copenhagen, Denmark	3
European Environmental Bureau (EEB)	Brussels, Belgium	3
European Federation of Allergy and Airways Diseases Patients' Associations (EFA)	Brussels, Belgium	4
European Respiratory Society (ERS)	Lausanne, Switzerland	4
Health and Environment Alliance (HEAL)	Brussels, Belgium	3, 4
International Society for Environmental Epidemiology (ISEE)	Herndon, VA, the United States	3, 4
International Transport Forum (ITF)	Paris, France	5
South Asia Co-operative Environment Programme (SACEP)	Colombo, Sri Lanka	3

^a Area of expertise/interest: 1. Air quality; 2. Climate change; 3. Environment in general; 4. Health; 5. Transport; 6. Energy.

Table A1.7. Working groups^a

Working group title	Experts involved	Group membership in the process
Risk of Bias Assessment		Guideline development group
		Guideline development group
		Guideline development group
		External methodologists
		External methodologists
Working group title	Experts involved	Group membership in the process
Certainty of Evidence Assessment		Guideline development group
		Guideline development group
		Guideline development group
		Guideline development group
		External methodologists
		External methodologists
Derivation of Air Quality Guideline Levels and Interim Targets		Guideline development group
		Guideline development group
		Guideline development group
		Systematic review team
		Guideline development group
		Guideline development group
		Guideline development group
		External methodologists
		Guideline development group
		Guideline development group

Table A1.7 contd

Working group title	Experts involved	Group membership in the process
Good Practice Statements		Guideline development group
		Guideline development group
		Guideline development group
		Guideline development group
		External review group
		External review group
		External review group

^a The working groups were coordinated by the members of the WHO Secretariat, Román Pérez Velasco and Dorota Jarosińska, with general assistance from Hanna Yang and specific support from Pierpaolo Mudu on the good practice statements related to SDS. The work produced by the working groups was reviewed by the GDG and by members of the systematic review team and the ERG, where needed.

^b Technical consultant who supported the work conducted by the working group.

^c Deceased 21 September 2020.

Annex 2. Assessment of conflict of interest

All external contributors to the guidelines, including members of the GDG, systematic review team, external methodologists and ERG, completed WHO declaration of interest forms in accordance with WHO's policy for experts. Further, WHO technical staff reviewed curricula vitae of candidates for the groups. At the beginning of the GDG meetings, participants declared or updated their competing interests ([Table A2.1](#)).

The conflict-of-interest assessment was done according to WHO procedures. If a conflict was declared, an initial review was undertaken by the WHO Secretariat to assess its relevance and significance. A declared conflict of interest is insignificant or minimal if it is unlikely to affect or to be reasonably perceived to affect the expert's judgement. Insignificant or minimal interests are those unrelated or only tangentially related to the subject of the activity or work and its outcome; nominal in amount or inconsequential in importance; or expired and unlikely to affect current behaviour.

The WHO Secretariat reviewed and assessed the declarations, which were cleared through the Office of Compliance, Risk Management and Ethics when required. WHO was of the opinion that these declarations did not constitute conflicts of interest and that the considered experts could participate in the process subject to disclosure of their interests.

The relevant declared interests of members of the GDG are summarized below. Other participants in the process, such as the systematic review team (see the special issue of *Environment International: Update of the WHO global air quality guidelines: systematic reviews* (Whaley et al., 2021; see main reference list)) and external methodologists, did not declare relevant interests. Some individual members of the large ERG declared non-significant, relevant interests. However, these interests – as well as those of the stakeholder organizations – were carefully considered in assessing their inputs and comments.

Table A2.1. Summary of relevant interests declared by members of the GDG

Name	Details of interests
	Employed by the Environment Public Authority of Kuwait
	Consultant for HEI and the British Columbia Lung Association; research support from HEI; travel expenses to meetings of the European Respiratory Society; expert opinions for the Ministry of Justice, Province of Ontario, for the Greater Vancouver Regional District and, on behalf of EcoJustice, on a lawsuit against the Province of Ontario; chair of the Air Pollution Expert Group of the World Heart Federation (2019–present); honorarium paid for by the Electric Power Research Institute to present at its annual meeting regarding the International Agency for Research on Cancer’s air pollution monograph
	Research support from the HEI; Chairman of the European Respiratory Society Environment and Health Committee (2014–2017)
	Formerly employed by HEI; consulting for HEI and Vital Strategies
	Consultant for HEI, Health Canada, World Bank and WHO; member of the European Respiratory Society Environment and Health Committee
	Employed by Clean Air Asia
	Research support from the Tehran University of Medical Sciences
	Employed by Health Canada and formerly employed by WHO (until 2017)
	Consultant for WHO
	Consultant for the Frank Bold Society, Health and Environment Alliance, Health Canada, UN Environment, Vital Strategies and WHO; chair of the Policy Committee at the International Society for Environmental Epidemiology (until 2018); member of the Board of the International Joint Policy Committee of the Societies of Epidemiology (until 2018)

Table A2.1 contd

Name	Details of interests
	President of the Swiss Federal Commission on Air Hygiene; member of the European Respiratory Society Environment and Health Committee (until 2018)
	Travel expenses to meeting paid for by the American Petroleum Institute; expert opinion for the United States Department of Justice on the lawsuit <i>United States vs Mountain State Carbon, LLC</i>
	Consultant for WHO (March–July 2019)
	Employed by Clean Air Asia (until 2017); expert opinion on behalf of Clean Air Asia in the development of PM _{2.5} standards in the Philippines (2013)
	Employed by the European Lung Foundation
	Chair of the Oversight Committee of Long-Term Epidemiological Studies of Air Pollution, HEI
	Consultant for the World Bank; research support from the European Commission
	Employed by South African Medical Research Council; research support from National Department of Environmental Affairs of South Africa; vice-President of the National Association for Clean Air (until 2018); founder of the Environmental Health Research Network; member of the Public Health Association of South Africa
	Employed by the European Commission (until 2017)

^a Deceased 21 September 2020.

Annex 3. Summaries of systematic reviews of evidence informing the air quality guideline levels

This annex contains the abstracts and certainty of evidence tables from the systematic reviews published in the special issue of *Environment International: Update of the WHO global air quality guidelines: systematic reviews* (Whaley et al., 2021; see main reference list), where additional information can be found. The abstracts and tables are provided in this annex courtesy of *Environment International*.

A3.1 Long-term exposure to PM and all-cause and cause-specific mortality: a systematic review and meta-analysis (Chen & Hoek, 2020)

Abstract

As new scientific evidence on health effects of air pollution is generated, air quality guidelines need to be periodically updated. The objective of this review is to support the derivation of updated guidelines by the World Health Organization (WHO) by performing a systematic review of evidence of associations between long-term exposure to particulate matter with diameter under 2.5 μm (PM_{2.5}) and particulate matter with diameter under 10 μm (PM₁₀), in relation to all-cause and cause-specific mortality. As there is especially uncertainty about the relationship at the low and high end of the exposure range, the review needed to provide an indication of the shape of the concentration-response function (CRF).

We systematically searched MEDLINE and EMBASE from database inception to 9 October 2018. Articles were checked for eligibility by two reviewers. We included cohort and case-control studies on outdoor air pollution in human populations using individual level data. In addition to natural-cause mortality, we evaluated mortality from circulatory diseases (ischemic heart disease (IHD) and cerebrovascular disease (stroke) also specifically), respiratory diseases (Chronic Obstructive Pulmonary Disease (COPD) and acute lower respiratory

illness (ALRI) also specifically) and lung cancer. A random-effect meta-analysis was performed when at least three studies were available for a specific exposure-outcome pair. Risk of bias was assessed for all included articles using a specifically developed tool coordinated by WHO. Additional analyses were performed to assess consistency across geographic region, explain heterogeneity and explore the shape of the CRF. A GRADE (Grading of Recommendations Assessment, Development and Evaluation) assessment of the body of evidence was made using a specifically developed tool coordinated by WHO.

A large number (N=107) of predominantly cohort studies (N=104) were included after screening more than 3000 abstracts. Studies were conducted globally with the majority of studies from North America (N=62) and Europe (N=25). More studies used PM_{2.5} (N=71) as the exposure metric than PM₁₀ (N=42). PM_{2.5} was significantly associated with all causes of death evaluated. The combined Risk Ratio (RR) for PM_{2.5} and natural-cause mortality was 1.08 (95%CI 1.06, 1.09) per 10 µg/m³. Meta analyses of studies conducted at the low mean PM_{2.5} levels (<25, 20, 15, 12, 10 µg/m³) yielded RRs that were similar or higher compared to the overall RR, consistent with the finding of generally linear or supralinear CRFs in individual studies. Pooled RRs were almost identical for studies conducted in North America, Europe and Western Pacific region. PM₁₀ was significantly associated with natural cause and most but not all causes of death. Application of the risk of bias tool showed that few studies were at a high risk of bias in any domain. Application of the GRADE tool resulted in an assessment of “high certainty of evidence” for PM_{2.5} with all assessed endpoints except for respiratory mortality (moderate). The evidence was rated as less certain for PM₁₀ and cause-specific mortality (“moderate” for circulatory, IHD, COPD and “low” for stroke mortality).

Compared to the previous global WHO evaluation, the evidence base has increased substantially. However studies conducted in low and middle income countries (LMICs) are still limited. There is clear evidence that both PM_{2.5} and PM₁₀ were associated with increased mortality from all causes, cardiovascular disease, respiratory disease and lung cancer. Associations remained below the current WHO guideline value of 10 µg/m³ for PM_{2.5}.

Systematic review registration number (PROSPERO ID): CRD42018082577.

Table A3.1. Certainty of evidence profile for each exposure–outcome combination

	Reasons for downgrading							Reasons for upgrading							Overall	Final certainty assessment		
	A1	Rationale	A2	Rationale	A3	Rationale	A4	Rationale	A5	Rationale	B1	Rationale	B2	Rationale			B3	Rationale
PM _{2.5} and natural cause	0	Little influence on the overall effect	0	No evidence of indirectness	0	Prediction interval does not include unity	0	Sample size large enough to assess RR with sufficient precision	0	No evidence of publication bias	0	Insufficient basis for upgrading	0	Confounders would shift the RR in both directions	+1	Evidence of increase in risk with increasing exposure	+1	High
PM ₁₀ and natural cause	0	Little influence on the overall effect	0	No evidence of indirectness	0	Prediction interval does not include unity	0	Sample size large enough to assess RR with sufficient precision	0	No evidence of publication bias	0	Insufficient basis for upgrading	0	Confounders would shift the RR in both directions	+1	Evidence of increase in risk with increasing exposure	+1	High
PM _{2.5} and circulatory	0	Little influence on the overall effect	0	No evidence of indirectness	0	Prediction interval does not include unity	0	Sample size large enough to assess RR with sufficient precision	0	No evidence of publication bias	0	Insufficient basis for upgrading	0	Confounders would shift the RR in both directions	+1	Evidence of increase in risk with increasing exposure	+1	High
PM ₁₀ and circulatory	0	Little influence on the overall effect	0	No evidence of indirectness	-1	Prediction interval includes unity and larger than twice the CI	0	Sample size large enough to assess RR with sufficient precision	0	No evidence of publication bias	0	Insufficient basis for upgrading	0	Confounders would shift the RR in both directions	+1	Evidence of increase in risk with increasing exposure	0	Moderate

A1 = limitations in studies (risk of bias); A2 = indirectness; A3 = inconsistency; A4 = imprecision; A5 = publication bias.
B1 = large RR; B2 = all confounding decreases observed RR; B3 = concentration–response gradient.
Note: “+” indicates an increase in the confidence level; “-” indicates a decrease in the confidence level; “0” indicates no change in the confidence level.

A3.2 Long-term exposure to NO₂ and O₃ and all-cause and respiratory mortality: a systematic review and meta-analysis (Huangfu & Atkinson, 2020)

Abstract

Background: WHO has published several volumes of Global Air Quality Guidelines to provide guidance on the health risks associated with exposure to outdoor air pollution. As new scientific evidence is generated, air quality guidelines need to be periodically revised and, where necessary, updated.

Objectives: The aims of the study were 1) to summarise the available evidence on the effect of long-term exposure to ozone (O₃) and nitrogen dioxide (NO₂) on mortality; 2) and to assess concentration response functions (CRF), their shape and the minimum level of exposures measured in studies to support WHO's update of the air quality guidelines.

Data sources: We conducted a systematic literature search of the Medline, Embase and Web of Science databases following a protocol proposed by WHO and applied Preferred Reporting Items for Systematic Review and Meta-Analyses (PRISMA) guidelines for reporting our results.

Study eligibility criteria: Cohort studies in human populations (including sub-groups at risk) exposed to long-term concentrations of NO₂ and O₃. Outcomes assessed were all-cause, respiratory, Chronic Obstructive Pulmonary Disease (COPD) and Acute Lower Respiratory Infection (ALRI) mortality.

Study appraisal and synthesis methods: Studies included in the meta-analyses were assessed using a new risk of bias instrument developed by a group of experts convened by WHO. Study results are presented in forest plots and quantitative meta-analyses were conducted using random effects models. The certainty of evidence was assessed using a newly developed adaptation of GRADE.

Results: The review identified 2068 studies of which 95 were subject to review with 45 meeting the inclusion criteria. An update in September 2018 identified 159 studies with 1 meeting the inclusion criteria. Of the 46 included studies, 41 reported results for NO₂ and 20 for O₃. The majority of studies were from the USA and Europe with the remainder from Canada, China and Japan. Forty-two studies reported results for all-cause mortality and 22 for respiratory mortality.

Associations for NO₂ and mortality were positive; random-effects summary relative risks (RR) were 1.02 (95% CI: 1.01, 1.04), 1.03 (1.00, 1.05), 1.03 (1.01, 1.04) and 1.06 (1.02, 1.10) per 10 µg/m³ for all-cause (24 cohorts), respiratory (15 cohorts), COPD (9 cohorts) and ALRI (5 cohorts) mortality respectively. The review identified high levels of heterogeneity for all causes of death except COPD. A small number of studies investigated the shape of the concentration–response relationship and generally found little evidence to reject the assumption of linearity across the concentration range.

Studies of O₃ using annual metrics showed the associations with all-cause and respiratory mortality were 0.97 (0.93, 1.02) and 0.99 (0.89, 1.11) per 10 µg/m³ respectively. For studies using peak O₃ metrics, the association with all-cause mortality was 1.01 (1.00, 1.02) and for respiratory mortality 1.02 (0.99, 1.05), each per 10 µg/m³. The review identified high levels of heterogeneity. Few studies investigated the shape of the concentration–response relationship.

Certainty in the associations (adapted GRADE) with mortality was rated low to moderate for each exposure–outcome pair, except for NO₂ and COPD mortality which was rated high.

Limitations: The substantial heterogeneity for most outcomes in the review requires explanation. The evidence base is limited in terms of the geographical spread of the study populations and, for some outcomes, the small number of independent cohorts for meta-analysis precludes meaningful meta-regression to explore causes of heterogeneity. Relatively few studies assessed specifically the shape of the CRF or multi-pollutant models.

Conclusions: The short-comings in the existing literature base makes determining the precise nature (magnitude and linearity) of the associations challenging. Grade assessments were moderate or low for both NO₂ and O₃ for all causes of mortality except for NO₂ and COPD mortality where the certainty of the evidence was judged as high.

Table A3.2. Certainty of evidence profile for NO₂ and all-cause mortality

Domain	Judgement	Down/up grade
Limitations in studies	Twenty-four included studies. Risk of bias moderate because although not all studies adjusted for all confounders, exclusion of high risk of bias studies did not reduce the summary RR	No downgrading
Indirectness	All studies included the desired population, exposures and outcomes	No downgrading
Inconsistency	The 80% prediction interval included 1 & > twice CI. High level of heterogeneity in general population studies. Studies controlling for individual measures of BMI, smoking, SES gave slightly higher, less precise summary RR. Exclusion of patient cohorts (6) did not change summary RR & CI	Downgrade one level
Imprecision	The number of person years in the included studies was greater than 940 000	No downgrading
Publication bias	According to the funnel plot and Egger's test ($P < 0.1$), there were sign of publication bias/funnel plot asymmetry	No downgrading
Large effect size	Summary RR = 1.02. Precision reduced for cohorts with all individual confounder adjustment but not summary estimate. Insufficient information on unmeasured potential confounders available	No upgrading
Plausible confounding towards null	Confounding direction unknown but precision may be affected	No upgrading
Dose-response relation	A linear dose-response relationship was assumed in all studies. 5 studies investigated the shape of the dose-response relationship with no evidence to suggest non-linear. 95% CI for linear RR excluded 1	Upgrade one level
GRADE conclusion	Downgrade one level and upgrade one level	MODERATE CERTAINTY EVIDENCE MEAN RR UNADJUSTED FOR CO-POLLUTANTS EQUALS 1.02 PER 10 µg/m ³

Table A3.3. Certainty of evidence profile for NO₂ and respiratory mortality

Domain	Judgement	Down/up grade
Limitations in studies	Fifteen included studies. Risk of bias moderate because although not all studies adjusted for all confounders, exclusion of high risk of bias studies did not alter summary RR	No downgrading
Indirectness	All studies included the desired population, exposures and outcomes	No downgrading
Inconsistency	The 80% prediction interval included 1; PI = 2 x CI. Studies controlling for individual measures of BMI, smoking, SES gave lower summary RR and CI included 1. Exclusion of single patient cohort did not change summary RR & CI. High level of heterogeneity in general population studies	Downgrade one level
Imprecision	The number of person years in the included studies was greater than 940 000	No downgrading
Publication bias	According to the funnel plot little evidence of publication bias	No downgrading
Large effect size	Summary RR = 1.03. Insufficient information on unmeasured potential confounders available	No upgrading
Plausible confounding towards null	Confounding direction unknown but precision may be affected	No upgrading
Dose–response relation	A linear dose–response relationship was assumed in all studies, 95% CI for linear RR excluded 1. No evidence to confirm shape of the dose–response relationship	Upgrade one level
GRADE conclusion	No downgrade and no upgrade	MODERATE CERTAINTY EVIDENCE MEAN RR UNADJUSTED FOR CO-POLLUTANTS EQUALS 1.03 PER 10 µg/m ³

Table A3.4. Certainty of evidence profile for NO₂ and COPD mortality

Domain	Judgement	Down/up grade
Limitations in studies	Nine included studies. Risk of bias moderate because although not all studies adjusted for all confounders, exclusion of 2 high risk of bias studies did not alter summary RR	No downgrading
Indirectness	All studies included the desired population, exposures and outcomes	No downgrading
Inconsistency	The 80% prediction interval did not include 1	No downgrading
Imprecision	The number of person years in the included studies was greater than 940 000	No downgrading
Publication bias	No analysis of publication bias – too few studies (n=9)	No downgrading
Large effect size	Summary RR = 1.02. Insufficient information on unmeasured potential confounders available	No upgrading
Plausible confounding towards null	Confounding direction unknown but precision may be affected	No upgrading
Dose–response relation	A linear dose–response relationship was assumed in all studies, 95% CI for linear RR excluded 1. 2 studies investigated the shape of the dose–response relationship with no evidence to suggest non-linear	Upgrade one level
GRADE conclusion	No downgrade and upgrade one level	HIGH CERTAINTY EVIDENCE MEAN RR UNADJUSTED FOR CO-POLLUTANTS EQUALS 1.03 PER 10 µg/m ³

Table A3.5. Certainty of evidence profile for NO₂ and ALRI mortality

Domain	Judgement	Down/up grade
Limitations in studies	Five included studies. Risk of bias moderate for all studies, not all studies adjusted for all confounders	No downgrading
Indirectness	All studies included the desired population, exposures and outcomes	No downgrading
Inconsistency	The 80% prediction interval included 1 but the PI was not > 2 x CI. Substantial heterogeneity amongst small number of studies	Downgrade one level
Imprecision	The number of person years in the included studies was greater than 940 000	No downgrading
Publication bias	No analysis of publication bias – too few studies	No downgrading
Large effect size	Summary RR = 1.02. Insufficient information on unmeasured potential confounders available	No upgrading
Plausible confounding towards null	Confounding direction unknown but precision may be affected	No upgrading
Dose–response relation	No information on shape. 95% CI for linear RR excluded 1	Upgrade one level
GRADE conclusion	No downgrade and no upgrade	MODERATE CERTAINTY EVIDENCE MEAN RR UNADJUSTED FOR CO-POLLUTANTS EQUALS 1.06 PER 10 µg/m ³

Table A3.6. Certainty of evidence profile for O₃ annual exposure and all-cause mortality

Domain	Judgement	Down/up grade
Limitations in studies	Nine included studies. Three studies with a total weight of 28% in the meta-analysis had high risk of bias. Excluding these studies did not change significantly the summary RR	No downgrading
Indirectness	One study with study sample of stroke patients based in London. However, it was a small study and only carried 1% weight	No downgrading
Inconsistency	The 80% prediction interval included 1 & PI > 2 x CI	Downgrade one level
Imprecision	The number of person years in the included studies was greater than 940 000	No downgrading
Publication bias	No analysis of publication bias – too few studies (n=9)	No downgrading
Large effect size	Summary RR=0.97	No upgrading
Plausible confounding towards null	Confounding direction unknown but precision may be affected	No upgrading
Dose–response relation	A linear dose–response relationship was assumed in all studies. 95% CI for linear RR included 1. None of the studies reported the dose–response relationship	No upgrading
GRADE conclusion	Downgrade one level and no upgrade	LOW CERTAINTY EVIDENCE MEAN RR UNADJUSTED FOR CO-POLLUTANTS EQUALS 0.97 PER 10 µg/m ³

Table A3.7. Certainty of evidence profile for O₃ annual exposure and respiratory mortality

Domain	Judgement	Down/up grade
Limitations in studies	Only 4 studies; all rated low or moderate risk of bias	No downgrading
Indirectness	All studies included the desired population, exposures and outcomes	No downgrading
Inconsistency	The 80% prediction interval included 1 & PI > 2 x CI. Substantial heterogeneity amongst small number of studies	Downgrade one level
Imprecision	The number of person years in the included studies was greater than 940 000	No downgrading
Publication bias	No analysis of publication bias – too few studies (n=4)	No downgrading
Large effect size	Summary RR=0.99	No upgrading
Plausible confounding towards null	Confounding direction unknown but precision may be affected	No upgrading
Dose–response relation	A linear dose–response relationship was assumed in all studies. 95% CI for linear RR included 1. None of the studies reported dose–response relationship	No upgrading
GRADE conclusion	Downgrade one level and no upgrade	LOW CERTAINTY EVIDENCE MEAN RR UNADJUSTED FOR CO-POLLUTANTS EQUALS 0.99 PER 10 µg/m ³

Table A3.8. Certainty of evidence profile for O₃ peak exposure and all-cause mortality

Domain	Judgement	Down/up grade
Limitations in studies	Seven included studies. One study with high risk of bias – exclusion did not change summary RR	No downgrading
Indirectness	One study might have introduced some selection bias due to the volunteering sample chosen. However, it was only weighted at less than 2% among all studies	No downgrading
Inconsistency	The 80% prediction interval included 1; PI = 2 x CI	No downgrading
Imprecision	The number of person years in the included studies was greater than 940 000	No downgrading
Publication bias	No analysis of publication bias – too few studies (<i>n</i> =6)	No downgrading
Large effect size	Summary RR = 1.01. All critical confounders were adjusted for. Insufficient information on unmeasured potential confounders available	No upgrading
Plausible confounding towards null	Confounding direction unknown but precision may be affected	No upgrading
Dose–response relation	A linear dose–response relationship was assumed in all studies. 95% CI for linear RR included 1. One study investigated the shape of the dose–response relationship with no evidence to suggest non-linear	No upgrading
GRADE conclusion	No downgrade and no upgrade	MODERATE CERTAINTY EVIDENCE MEAN RR UNADJUSTED FOR CO-POLLUTANTS EQUALS 1.01 PER 10 µg/m ³

Table A3.9. Certainty of evidence profile for O₃ peak exposure and respiratory mortality

Domain	Judgement	Down/up grade
Limitations in studies	Four included studies. One study high risk of bias. Exclusion did not alter significantly the RR and CI	No downgrading
Indirectness	All studies included the desired population, exposures and outcomes	No downgrading
Inconsistency	The 80% prediction interval included 1; PI = 2 x CI. Substantial heterogeneity amongst small number of studies	Downgrade one level
Imprecision	The number of person years in the included studies was greater than 940 000	No downgrading
Publication bias	No analysis of publication bias – too few studies (n=3)	No downgrading
Large effect size	Summary RR = 1.02. Insufficient information on unmeasured potential confounders available	No upgrading
Plausible confounding towards null	Confounding direction unknown but precision may be affected	No upgrading
Dose–response relation	A linear dose–response relationship was assumed in all studies. 95% CI for linear RR included 1. One study investigated the dose–response relationship. No evidence to confirm shape of the dose–response relationship for ozone exposure	No upgrading
GRADE conclusion	No downgrade and no upgrade	LOW CERTAINTY EVIDENCE MEAN RR UNADJUSTED FOR CO-POLLUTANTS EQUALS 1.02 PER 10 µg/m ³

A3.3 Short-term exposure to particulate matter (PM₁₀ and PM_{2.5}), nitrogen dioxide (NO₂), and ozone (O₃) and all-cause and cause-specific mortality: systematic review and meta-analysis (Orellano et al., 2020)

Abstract

Background: Air pollution is a leading cause of mortality and morbidity worldwide. Short-term exposure (from one hour to days) to selected air pollutants has been associated with human mortality. This systematic review was conducted to analyse the evidence on the effects of short-term exposure to particulate matter with aerodynamic diameters less or equal than 10 and 2.5 µm (PM₁₀, PM_{2.5}), nitrogen dioxide (NO₂), and ozone (O₃), on all-cause mortality, and PM₁₀ and PM_{2.5} on cardiovascular, respiratory, and cerebrovascular mortality.

Methods: We included studies on human populations exposed to outdoor air pollution from any source, excluding occupational exposures. Relative risks (RRs) per 10 µg/m³ increase in air pollutants concentrations were used as the effect estimates. Heterogeneity between studies was assessed using 80% prediction intervals. Risk of bias (RoB) in individual studies was analysed using a new domain-based assessment tool, developed by a working group convened by the World Health Organization and designed specifically to evaluate RoB within eligible air pollution studies included in systematic reviews. We conducted subgroup and sensitivity analyses by age, sex, continent, study design, single or multicity studies, time lag, and RoB. The certainty of evidence was assessed for each exposure-outcome combination. The protocol for this review was registered with PROSPERO (CRD42018087749).

Results: We included 196 articles in quantitative analysis. All combinations of pollutants and all-cause and cause-specific mortality were positively associated in the main analysis, and in a wide range of sensitivity analyses. The only exception was NO₂, but when considering a 1-hour maximum exposure. We found positive associations between pollutants and all-cause mortality for PM₁₀ (RR: 1.0041; 95% CI: 1.0034-1.0049), PM_{2.5} (RR: 1.0065; 95% CI: 1.0044-1.0086), NO₂ (24-hour average) (RR: 1.0072; 95% CI: 1.0059-1.0085), and O₃ (RR: 1.0043; 95% CI: 1.0034-1.0052). PM₁₀ and PM_{2.5} were also positively associated with cardiovascular, respiratory, and cerebrovascular mortality. We found some degree of heterogeneity between studies in three exposure-outcome combinations, and this heterogeneity could not be explained after subgroup analysis. RoB was low or moderate in the majority of articles.

The certainty of evidence was judged as high in 10 out of 11 combinations, and moderate in one combination.

Conclusions: This study found evidence of a positive association between short-term exposure to PM₁₀, PM_{2.5}, NO₂, and O₃ and all-cause mortality, and between PM₁₀ and PM_{2.5} and cardiovascular, respiratory and cerebrovascular mortality. These results were robust through several sensitivity analyses. In general, the level of evidence was high, meaning that we can be confident in the associations found in this study.

Table A3.10 Certainty of evidence profile for each exposure–outcome combination

Exposure – outcome	Limitations in studies	Indirectness	Inconsistency	Imprecision	Publication bias	Large effect size	Confounding	Concentration–response gradient	Certainty of evidence
PM ₁₀ – all-cause mortality	(0) No differences between studies with low/moderate versus high RoB	(0) The research question in the studies reflects the original question.	(0) 80% prediction interval did not include unity.	(0) Number of mortality cases higher than 100,000.	(0) Publication bias detected, but no difference between multicity and single-city studies was observed.	(+1) Unmeasured confounding would not suffice to explain away the effect estimate.	(0) Several potential confounders that would shift the RR in both directions.	(+1) Significant positive association detected in the main analysis.	High ⊠⊠⊠⊠
PM ₁₀ – cardiovascular mortality	(0) No differences between studies with low/moderate versus high RoB	(0) The research question in the studies reflects the original question.	(0) 80% prediction interval did not include unity.	(0) Number of mortality cases higher than 100,000.	(0) Publication bias detected, but no difference between multicity and single-city studies was observed.	(+1) Unmeasured confounding would not suffice to explain away the effect estimate.	(0) Several potential confounders that would shift the RR in both directions.	(+1) Significant positive association detected in the main analysis.	High ⊠⊠⊠⊠
PM ₁₀ – respiratory mortality	(0) No differences between studies with low/moderate versus high RoB	(0) The research question in the studies reflects the original question.	(0) 80% prediction interval did not include unity.	(0) Number of mortality cases higher than 100,000.	(0) Publication bias was not detected.	(+1) Unmeasured confounding would not suffice to explain away the effect estimate.	(0) Several potential confounders that would shift the RR in both directions.	(+1) Significant positive association detected in the main analysis.	High ⊠⊠⊠⊠

Certainty of evidence, starting from moderate certainty (⊠⊠⊠⊠); CRFs, concentration–response functions; (), between brackets is the downgrading of levels in that domain; RoB, risk of bias in individual studies; RR, relative risk.

ANNEX 3. SUMMARIES OF SYSTEMATIC REVIEWS OF EVIDENCE INFORMING THE AIR QUALITY GUIDELINE LEVELS

Exposure – outcome	Limitations in studies	Indirectness	Inconsistency	Imprecision	Publication bias	Large effect size	Confounding	Concentration-response gradient	Certainty of evidence
PM _{10-2.5} – cerebrovascular mortality	(0) No differences between studies with low/moderate and high RoB	(0) The research question in the studies reflects the original question.	(0) 80% prediction interval did not include unity.	(0) Number of mortality cases higher than 100,000.	(0) Publication bias detected, but no difference between multicity and single-city studies was observed.	(+1) Unmeasured confounding would suffice to explain away the effect estimate.	(0) Several potential confounders that would shift the RR in both directions.	(+1) Significant positive association detected in the main analysis.	High ⊠⊠⊠⊠⊠
PM _{2.5} – all-cause mortality	(0) Statistical differences between studies with low/moderate versus high RoB, but studies showing high RoB had small weight on the results	(0) The research question in the studies reflects the original question.	(0) 80% prediction interval did not include unity.	(0) Number of mortality cases higher than 100,000.	(0) Publication bias detected, but no difference between multicity and single-city studies was observed.	(+1) Unmeasured confounding would suffice to explain away the effect estimate.	(0) Several potential confounders that would shift the RR in both directions.	(+1) Significant positive association detected in the main analysis.	High ⊠⊠⊠⊠⊠
PM _{2.5} – cardiovascular mortality	(0) No differences between studies with low/moderate versus high RoB	(0) The research question in the studies reflects the original question.	(0) 80% prediction interval did not include unity.	(0) Number of mortality cases higher than 100,000.	(0) Publication bias was not detected.	(+1) Unmeasured confounding would suffice to explain away the effect estimate.	(0) Several potential confounders that would shift the RR in both directions.	(+1) Significant positive association detected in the main analysis.	High ⊠⊠⊠⊠⊠

Certainty of evidence, starting from moderate certainty (moderate); CRFs, concentration–response functions; I, between brackets is the downgrading of levels in that domain; RoB, risk of bias in individual studies; RR, relative risk.

Table A3.10 contd

Exposure – outcome	Limitations in studies	Indirectness	Inconsistency	Imprecision	Publication bias	Large effect size	Confounding	Concentration-response gradient	Certainty of evidence
Downgrade				Upgrade					
PM _{2.5} – respiratory mortality	(0)	(0)	(0)	(0)	(0)	(0)	(0)	(+1)	High ⊠⊠⊠⊠
	No differences between studies with low/moderate versus high RoB	The research question in the studies reflects the original question.	80% prediction interval included unity, but is not twice the confidence interval.	Number of mortality cases higher than 100,000.	Publication bias was not detected.	Unmeasured confounding could influence the effect estimate.	Several potential confounders that would shift the RR in both directions.	Significant positive association detected in the main analysis.	
PM _{2.5} – cerebrovascular mortality	(-1)	(0)	(0)	(0)	(0)	(+1)	(0)	(+1)	High ⊠⊠⊠⊠
	Statistical differences between studies with low/moderate versus high RoB	The research question in the studies reflects the original question.	80% prediction interval included unity, but is not twice the confidence interval.	Number of mortality cases higher than 100,000.	Publication bias was not detected.	Unmeasured confounding would not suffice to explain away the effect estimate.	Several potential confounders that would shift the RR in both directions.	Significant positive association detected in the main analysis.	
NO ₂ (24-hour average) – all-cause mortality	(-1)	(0)	(0)	(0)	(0)	(+1)	(0)	(+1)	High ⊠⊠⊠⊠
	Statistical differences between studies with low/moderate versus high RoB	The research question in the studies reflects the original question.	80% prediction interval did not include unity.	Number of mortality cases higher than 100,000.	Publication bias detected, but no difference between multicity and single-city studies was observed.	Unmeasured confounding would not suffice to explain away the effect estimate.	Several potential confounders that would shift the RR in both directions.	Significant positive association detected in the main analysis.	

Certainty of evidence, starting from moderate certainty (⊠⊠⊠⊠); CRFs; concentration–response functions; (); between brackets is the downgrading of levels in that domain; RoB, risk of bias in individual studies; RR, relative risk.

Table A3.10 contd

Exposure – outcome	Limitations in studies	Indirectness	Inconsistency	Imprecision	Publication bias	Large effect size	Confounding	Concentration-response gradient	Certainty of evidence
Downgrade					Upgrade				
NO ₂ – (1-hour max.) – all-cause mortality	(0)	(0)	(0)	(0)	(0)	(0)	(0)	(0)	Moderate ⊗⊗⊗⊗
	No differences between studies with low/moderate versus high RoB	The research question in the studies reflects the original question.	80% prediction interval did not include unity.	Number of mortality cases higher than 100,000.	Publication bias was not detected.	Unmeasured confounding could influence the effect estimate.	Several potential confounders that would shift the RR in both directions.	No significant association detected in the main analysis.	
O ₃ – all-cause mortality	(0)	(0)	(0)	(0)	(0)	(+1)	(0)	(+1)	High ⊗⊗⊗⊗
	No differences between studies with low/moderate versus high RoB	The research question in the studies reflects the original question.	80% prediction interval did not include unity.	Number of mortality cases higher than 100,000.	Publication bias detected, but no difference between multicity and single-city studies was observed.	Unmeasured confounding would not suffice to explain away the effect estimate.	Several potential confounders that would shift the RR in both directions.	Significant positive association detected in the main analysis.	

Certainty of evidence, starting from moderate certainty (⊗⊗⊗⊗); CRFs, concentration–response functions; (), between brackets is the downgrading of levels in that domain; RoB, risk of bias in individual studies; RR, relative risk.

A3.4 Short-term exposure to sulfur dioxide (SO₂) and all-cause and respiratory mortality: a systematic review and meta-analysis (Orellano, Reynoso & Quaranta, 2021)

Abstract

Background: Many studies have assessed the harmful effects of ambient air pollution on human mortality, but the evidence needs further exploration, analysis, and refinement, given the large number of studies that have been published in recent years. The objective of this study was to evaluate all the available evidence of the effect of short-term exposure to ambient sulphur dioxide (SO₂) on all-cause and respiratory mortality.

Methods: Articles reporting observational epidemiological studies were included, comprising time-series and case-crossover designs. A broad search and wide inclusion criteria were considered, encompassing international and regional databases, with no geographical or language restrictions. A random effect meta-analysis was conducted, and pooled relative risk for an increment of 10 µg/m³ in SO₂ concentrations were calculated for each outcome. We analysed the risk of bias (RoB) in individual studies for specific domains using a new domain-based RoB assessment tool, and the certainty of evidence across studies with an adaptation of the Grading of Recommendations Assessment, Development and Evaluation approach. The certainty of evidence was judged separately for each exposure-outcome combination. A number of subgroup and sensitivity analyses were carried out, as well as assessments of heterogeneity and potential publication bias. The protocol for this review was registered with PROSPERO (CRD42019120738).

Results: Our search retrieved 1,128 articles, from which 67 were included in quantitative analysis. The RoB was low or moderate in the majority of articles and domains. An increment of 10 µg/m³ in SO₂ (24-hour average) was associated with all-cause mortality (RR: 1.0059; 95% CI: 1.0046–1.0071; p-value: <0.01), and respiratory mortality (RR: 1.0067; 95% CI: 1.0025–1.0109; p-value: <0.01), while the same increment in SO₂ (1-hour max.) was associated with respiratory mortality (RR: 1.0052; 95% CI: 1.0013–1.0091; p-value: 0.03). Similarly, the association was positive but non-significant for SO₂ (1-hour max.) and all-cause mortality (RR: 1.0016; 95% CI: 0.9930–1.0102; p-value: 0.60). These associations were still significant after the adjustment for particulate matter, but not for other pollutants, according to the results from 13 articles that evaluated co-pollutant models. In general, linear concentration-response functions with no thresholds were found for the two outcomes, although this was only evaluated in a small number of studies. We found signs of heterogeneity for SO₂ (24-hour average) – respiratory mortality and SO₂ (1-hour max.) – all-cause mortality, and funnel plot asymmetry

for SO₂ (24-hour average) – all-cause mortality. The certainty of evidence was high in two combinations, i.e. SO₂ (24-hour average) – all-cause mortality and SO₂ (1-hour max.) – respiratory mortality, moderate in one combination, i.e. SO₂ (24-hour average) – respiratory mortality, and low in the remaining one combination.

Conclusions: Positive associations were found between short-term exposure to ambient SO₂ and all-cause and respiratory mortality. These associations were robust against several sensitivity analyses, and were judged to be of moderate or high certainty in three of the four exposure-outcome combinations.

Table A3.11. Certainty of evidence profile for each exposure–outcome combination

Exposure – outcome	Limitations in studies in studies	Indirectness	Inconsistency	Imprecision	Publication bias	Downgrade			Upgrade			Certainty of evidence
SO ₂ (24-hour average) – all-cause mortality	(0) No differences between studies with low/moderate versus high RoB	(0) The research question in the studies reflects the PECO question.	(0) 80% prediction interval did not include unity.	(0) At least one multic study showed a clinically meaningful association.	(0) Publication bias possibly detected, but a positive significant association was estimated when using multic studies only.	(+1) Unmeasured confounding would not suffice to explain away the effect estimate.	(0) Several potential confounders that would shift the RR in both directions.	(+1) Significant association detected in the main analysis.	High ⊠⊠⊠⊠			
SO ₂ (24-hour average) – respiratory mortality	(0) No differences between studies with low/moderate versus high RoB	(0) The research question in the studies reflects the PECO question.	(-1) 80% prediction interval included unity, and was twice the size of the 95% CI.	(0) At least one multic study showed a clinically meaningful association.	(0) Publication bias was not detected.	(0) According to the analysis of the E-value, the presence of unmeasured confounders is plausible.	(0) Several potential confounders that would shift the RR in both directions.	(+1) Significant association detected in the main analysis.	Moderate ⊠⊠⊠⊠			

Certainty of evidence, starting from moderate certainty (⊠⊠⊠⊠); (0), between brackets is the downgrading of levels in that domain; RoB, risk of bias in individual studies; RR, relative risk; CI, 95% confidence interval; PECO, population, exposure, comparator, and outcomes.

Table A3.11 contd

Exposure – outcome	Limitations in studies in studies	Indirectness	Inconsistency	Imprecision	Publication bias	Large effect size	Confounding	Concentration-response gradient	Certainty of evidence
Upgrade									
SO ₂ (1-hour max.) – all-cause mortality	(0) Not enough studies to detect differences in the RoB.	(0) The research question in the studies reflects the PECO question..	(-1) 80% prediction interval included unity, and was twice the size of the 95% CI	(0) At least one multicity study showed a clinically meaningful association..	(0) Not enough studies to analyse publication bias.	(0) Unmeasured confounding was not analysed, because the association was RR in both not significant.	(0) Several potential confounders that would shift the RR in both directions.	(0) The association was not significant.	Low ⊗⊗⊗⊗
SO ₂ (1-hour max.) – respiratory mortality	(0) Not enough studies to detect differences in the RoB.	(0) The research question in the studies reflects the PECO question.	(0) 80% prediction interval did not include unity.	(0) At least one multicity study showed a clinically meaningful association..	(0) Not enough studies to analyse publication bias.	(0) According to the analysis of the E-value, the presence of unmeasured confounders is plausible.	(0) Several potential confounders that would shift the RR in both directions.	(+1) Significant association detected in the main analysis.	High ⊗⊗⊗⊗

Certainty of evidence, starting from moderate certainty (⊗⊗⊗⊗); ⊕, between brackets is the downgrading of levels in that domain; RoB, risk of bias in individual studies; RR, relative risk; CI, 95% confidence interval; PECO, population, exposure, comparator, and outcomes.

A3.5 Short-term exposure to ozone, nitrogen dioxide, and sulphur dioxide and emergency department visits and hospital admissions due to asthma: a systematic review and meta-analysis (Zheng et al., 2021)

Abstract

Background: Air pollution is a major environmental hazard to human health and a leading cause of morbidity for asthma worldwide.

Objectives: To assess the current evidence on short-term effects (from several hours to 7 days) of exposure to ozone (O₃), nitrogen dioxide (NO₂), and sulphur dioxide (SO₂) on asthma exacerbations, defined as emergency room visits (ERVs) and hospital admissions (HAs).

Methods: We searched PubMed/MEDLINE, EMBASE and other electronic databases to retrieve studies that investigated the risk of asthma-related ERVs and HAs associated with short-term exposure to O₃, NO₂, or SO₂. We evaluated the risks of bias (RoB) for individual studies and the certainty of evidence for each pollutant in the overall analysis. A subgroup analysis was performed, stratified by sex, age, and type of asthma exacerbation. We conducted sensitivity analysis by excluding the studies with high RoB and based on the E-value. Publication bias was examined with the Egger's test and with funnel plots.

Results: Our literature search retrieved 9,059 articles, and finally 67 studies were included, from which 48 studies included the data on children, 21 on adults, 14 on the elderly, and 31 on the general population. Forty-three studies included data on asthma ERVs, and 25 on asthma HAs. The pooled relative risk (RR) per 10 µg/m³ increase of ambient concentrations was 1.008 (95%CI: 1.005, 1.011) for maximum 8-hour daily or average 24-hour O₃, 1.014 (95%CI: 1.008, 1.020) for average 24-hour NO₂, 1.010 (95%CI: 1.001, 1.020) for 24-hour SO₂, 1.017 (95%CI: 0.973, 1.063) for maximum 1-hour daily O₃, 0.999 (95%CI: 0.966, 1.033) for 1-hour NO₂, and 1.003 (95%CI: 0.992, 1.014) for 1-hour SO₂. Heterogeneity was observed in all pollutants except for 8-hour or 24-hour O₃ and 24-hour NO₂. In general, we found no significant differences between subgroups that can explain this heterogeneity. Sensitivity analysis based on the RoB showed certain differences in NO₂ and SO₂ when considering the outcome or confounding domains, but the analysis using the E-value showed that no unmeasured confounders were expected. There was no major evidence of publication bias.

Based on the adaptation of the Grading of Recommendations Assessment, Development and Evaluation, the certainty of evidence was high for 8-hour or 24-hour O₃ and 24-hour NO₂, moderate for 24-hour SO₂, 1-hour O₃, and 1-hour SO₂, and low for 1-hour NO₂.

Conclusion: Short-term exposure to daily O₃, NO₂, and SO₂ was associated with an increased risk of asthma exacerbation in terms of asthma-associated ERVs and HAs.

Table A3.12. Certainty of evidence profile for each exposure–outcome combination

Exposure – outcome	Downgrade					Upgrade			Certainty of evidence
	Limitations in studies	Indirectness	Inconsistency	Imprecision	Publication bias	Large effect size	Confounding	Concentration–response gradient	
O ₃ (8-h or 24-h) – ERV or HA	(0) Statistical differences between studies with low/moderate versus high RoB, but the pooled effect in low/moderate RoB articles is significant	(0) The research question in the studies reflects the original question	(0) 80% prediction interval did not include unity	(0) Number of asthma exacerbations higher than 150,000.	(0) Publication bias was not detected	(+1) Unmeasured confounding would not suffice to explain away the effect estimate	(0) Several potential confounders that would shift the RR in both directions.	(+1) Significant positive association detected in the main analysis.	High ⊠⊠⊠⊠
NO ₂ (24-h) – ERV or HA	(0) Statistical differences between studies with low/moderate versus high RoB, but the pooled effect in low/moderate RoB articles is significant	(0) The research question in the studies reflects the original question	(0) 80% prediction interval did not include unity	(0) Number of asthma exacerbations higher than 150,000.	(0) Publication bias was not detected	(+1) Unmeasured confounding would not suffice to explain away the effect estimate	(0) Several potential confounders that would shift the RR in both directions.	(+1) Significant positive association detected in the main analysis.	High ⊠⊠⊠⊠
SO ₂ (24-h) z–ERV or HA	(–1) Statistical differences between studies with low/moderate versus high RoB	(0) The research question in the studies reflects the original question	(–1) 80% prediction interval included unity, and was twice the width of the 95%CI	(0) Number of asthma exacerbations higher than 150,000.	(0) Publication bias was not detected	(+1) Unmeasured confounding would not suffice to explain away the effect estimate	(0) Several potential confounders that would shift the RR in both directions.	(+1) Significant positive association detected in the main analysis.	Moderate ⊠⊠⊠⊠

Certainty of evidence, starting from moderate certainty (⊠⊠⊠⊠); (–), between brackets is the downgrading of levels in that domain; RoB, risk of bias in individual studies; RR, relative risk.

Table A3.12 contd

Exposure – outcome	Limitations in studies	Indirectness	Inconsistency	Imprecision	Publication bias	Large effect size	Confounding	Concentration-response gradient	Certainty of evidence
Downgrade									
O ₃ (1-h) – ERV or HA	(0)	(0)	(0)	(0)	(0)	(0)	(0)	(0)	Moderate ⊗⊗⊗⊗
	Differences between studies with low/moderate versus high RoB were not evaluated.	The research question in the studies reflects the original question	80% prediction interval included unity, but is not twice the 95%CI	Number of asthma exacerbations higher than 150,000.	Publication bias was not evaluated	E-value was not calculated, because the pooled RR is not significant.	Several potential confounders that would shift the RR in both directions.	No significant association detected in the main analysis.	
Upgrade									
NO ₂ (1-h) – ERV or HA	(0)	(0)	(0)	(-1)	(0)	(0)	(0)	(0)	Low ⊗⊗⊗⊗
	No differences between studies with low/moderate versus high RoB.	The research question in the studies reflects the original question	80% prediction interval included unity, but is not twice the 95%CI	Number of asthma exacerbations lower than 150,000.	Publication bias was not evaluated	E-value was not calculated, because the pooled RR is not significant.	Several potential confounders that would shift the RR in both directions.	No significant association detected in the main analysis.	
SO ₂ (1-h) – ERV or HA	(0)	(0)	(0)	(0)	(0)	(0)	(0)	(0)	Moderate ⊗⊗⊗⊗
	Differences between studies with low/moderate versus high RoB were not evaluated.	The research question in the studies reflects the original question	80% prediction interval included unity, but is not twice the 95%CI	Number of asthma exacerbations higher than 150,000.	Publication bias was not evaluated	E-value was not calculated, because the pooled RR is not significant.	Several potential confounders that would shift the RR in both directions.	No significant association detected in the main analysis.	

Certainty of evidence, starting from moderate certainty (⊗⊗⊗⊗); ⊕, between brackets is the downgrading of levels in that domain; RoB, risk of bias in individual studies; RR, relative risk.

A3.6 Short-term exposure to carbon monoxide and myocardial infarction: a systematic review and meta-analysis (Lee et al., 2020)

Abstract

Background: Previous studies suggest an association between short-term exposure to carbon monoxide and myocardial infarction. We performed a systematic review and meta-analysis to assess current evidence on this association to support the update of the World Health Organization (WHO) Global Air Quality Guidelines.

Methods: We searched Medline, Embase and Cochrane Central Register of Controlled Trials to update the evidence published in a previous systematic review up to 30th September 2018 for studies investigating the association between short-term exposure to ambient carbon monoxide (up to lag of seven days) and emergency department visits or hospital admissions and mortality due to myocardial infarction. Two reviewers assessed potentially eligible studies and performed data extraction independently. Random-effects meta-analysis was used to derive the pooled risk estimate per 1mg/m³ increase in ambient carbon monoxide concentration. Risk of bias in individual studies was assessed using a domain-based assessment tool. The overall certainty of the body of evidence was evaluated using an adapted certainty of evidence assessment framework.

Results: We evaluated 1,038 articles from the previous review and our updated literature search, of which, 26 satisfied our inclusion criteria. Overall, myocardial infarction was associated with exposure to ambient carbon monoxide concentration (risk ratio of 1.052, 95% confidence interval 1.017 – 1.089 per 1mg/m³ increase). A third of studies were assessed to be at high risk of bias (RoB) due to inadequate adjustment for confounding. Using an adaptation of the Grading of Recommendations Assessment, Development and Evaluation (GRADE) framework, the overall evidence was assessed to be of moderate certainty.

Conclusions: This review demonstrated that the pooled risk ratio for myocardial infarction was 1.052 (95% CI 1.017–1.089) per 1mg/m³ increase in ambient carbon monoxide concentration. However, very few studies originated from low- and middle-income countries.

Table A3.13. Certainty of evidence profile for CO and myocardial infarction

Domain	Judgement	Down/up grade
Limitations in studies	Ten studies were assessed to be high risk of bias due to inadequate adjustment for confounding. Subgroup analysis did not demonstrate a statistically significant difference in risk estimates between studies at low/moderate risk of bias versus those at high risk of bias	No downgrading
Indirectness	All included studies were consistent with the prespecified PECOS	No downgrading
Inconsistency	The 80% prediction interval was 0.871–1.271. However, most of this is driven by 3 studies that reported outlying results. Sensitivity analysis excluding these studies had a 80% prediction interval of 1.002–1.030	No downgrading
Imprecision	Although the number of participants included in the review (1.5 million) was significantly lower than the estimated sample size required (12.1 million), risk estimates reported by the studies are sufficiently precise	No downgrading
Publication bias	Visual inspection of the funnel plot does not indicate significant asymmetry	No downgrading
Large effect size	Overall relative risk was 1.052. Based on this, an E-value of 1.29 was calculated. However there is insufficient information to determine strength of unmeasured confounders	No upgrading
Plausible confounding towards null	Direction of effect of other confounding is unknown	No upgrading
Dose–response relation	None of the studies reported the dose–response relationship	No upgrading
GRADE conclusion		MODERATE CERTAINTY OF EVIDENCE

The main objective of these updated global guidelines is to offer health-based air quality guideline levels, expressed as long- or short-term concentrations, for six key air pollutants: PM_{2.5}, PM₁₀, ozone, nitrogen dioxide, sulfur dioxide and carbon monoxide. In addition, the guidelines provide interim targets to guide reduction efforts for these pollutants, as well as good practice statements for the management of certain types of PM (i.e. black carbon/elemental carbon, ultrafine particles, and particles originating from sand and dust storms). These guidelines are not legally binding standards; however, they provide WHO Member States with an evidence-informed tool they can use to inform legislation and policy. Ultimately, the goal of these guidelines is to help reduce levels of air pollutants in order to decrease the enormous health burden resulting from exposure to air pollution worldwide.

Compared with previous WHO guidelines, these guidelines:

- use new methods for evidence synthesis and guideline development;
- reinforce previous evidence on the adverse health effects of air pollution; and
- provide evidence of adverse health effects from air pollution at lower levels than previously known.

The guidelines are a critical tool for the following three main groups of users:

- policy-makers, lawmakers and technical experts at the local, national and international levels who are responsible for developing and implementing regulations and standards for air quality, air pollution control, urban planning and other policy areas;
- national and local authorities and nongovernmental organizations, civil society organizations and advocacy groups such as patients, citizen groups, industrial stakeholders and environmental organizations; and
- academics, health and environmental impact assessment practitioners, and researchers in the broad field of air pollution.

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Article

Characterization of Lithium-Ion Battery Fire Emissions—Part 2: Particle Size Distributions and Emission Factors

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Abstract: The lithium-ion battery (LIB) thermal runaway (TR) emits a wide size range of particles with diverse chemical compositions. When inhaled, these particles can cause serious adverse health effects. This study measured the size distributions of particles with diameters less than 10 μm released throughout the TR-driven combustion of cylindrical lithium iron phosphate (LFP) and pouch-style lithium cobalt oxide (LCO) LIB cells. The chemical composition of fine particles ($\text{PM}_{2.5}$) and some acidic gases were also characterized from filter samples. The emission factors of particle number and mass as well as chemical components were calculated. Particle number concentrations were dominated by those smaller than 500 nm with geometric number mean diameters below 130 nm. Mass concentrations were also dominated by smaller particles, with PM_1 particles making up 81–95% of the measured PM_{10} mass. A significant amount of organic and elemental carbon, phosphate, and fluoride was released as $\text{PM}_{2.5}$ constituents. The emission factor of gaseous hydrogen fluoride was 10–81 mg/Wh, posing the most immediate danger to human health. The tested LFP cells had higher emission factors of particles and HF than the LCO cells.

Keywords: Li-ion battery; fire; smoke; particulate matter; thermal runaway; ultrafine particles; HF; $\text{PM}_{2.5}$; emission factor; particle size distribution



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1. Introduction

Lithium-ion batteries (LIB) can generate significant gaseous and particulate emissions when they experience thermal failure, through venting, thermal runaway (TR), fire, and explosion [1,2]. The detailed characterization of particle size distribution (PSD), chemical composition, emission factor, temporal evolution, and thermal stability is important for LIB safety, including understanding the health effects of inhaled smoke particles, proper personal protection, the mechanisms of fire origination and propagation, fire detection and suppression, post-fire cleanup, and material recycling. For example, recent studies of soot particles from LIB fires show higher toxicity to human cells than wood smoke [3,4]. However, as shown in a recent review [2], few studies have examined particle emissions from LIB fires.

Size distribution is an important parameter that describes the transport behavior, atmospheric residence time, and inhalation risk of particles. LIB TR generates a wide range of particle sizes, varying from several nanometers to several millimeters. Most previous studies have focused on larger particles that settled in the combustion chamber after experiments. Zhang et al. [5] collected particles after burning a nickel manganese cobalt (NMC) LIB cell and separated them by sieving into four size fractions. The mass percentages were

44% for 0–0.85 mm, 36% for 0.85–1.70 mm, 9% for 1.70–2.00 mm, and 11% for 2.00–8.00 mm. Laser diffraction particle sizing showed that the volume distribution had a median diameter of 198 μm for the 0–0.85 mm size fraction. Several other studies focusing on larger particles also showed volume distribution median diameters $>100 \mu\text{m}$ [6–8]. Due to their high temperature and large thermal mass, the larger sparking particles may contribute to igniting flammable gases and TR propagation [9]. As they dominate the emitted particle mass, the chemical composition of these larger particles provides useful information about particle origin and TR reactions [5,8,10]. When released into the environment, they may cause air, water, and soil contamination [11–13]. However, these particles represent a low inhalation risk to humans as their large size causes them to settle quickly by gravity and makes them unlikely to be inhaled while suspended in air [14].

Particles with aerodynamic diameters less than 10 μm (PM_{10}), 2.5 μm ($\text{PM}_{2.5}$), and even smaller (e.g., ultrafine particles with diameters $<100 \text{ nm}$ [$\text{PM}_{0.1}$]) can remain suspended in air for longer than larger particles. When inhaled, they will deposit at different locations in the human respiratory track depending on the particle size, causing respiratory, cardiovascular, and other diseases [15]. Very few studies have measured PM_{10} size distributions from LIB fires. Premnath et al. [16] used an Engine Exhaust Particle Sizer (EEPS) [17,18] to measure particle number concentrations in the size range of 5.6–560 nm in real time from lithium iron phosphate (LFP) and NMC cells undergoing TR. The geometric number mean diameters (GNMD) of the PSDs [19] ranged from 54 to 69 nm with either monomodal or bimodal distributions. The emission rate varied from 1.6×10^{15} to 2.1×10^{17} particles/hour, 5–6 orders of magnitude higher than modern heavy-duty diesel engines. While the tests evaluated the effects of the TR triggering mechanism (nail penetration vs. overcharging) and LIB chemistry (LFP vs. NMC), the number of tests were small, and the influence of the cell's state of charge (SOC) was not examined. Goupil et al. [20] measured PSDs from an NMC cell fire using a scanning mobility particle sizer (SMPS; 17.5–532.8 nm) and an optical particle spectrometer (250 nm–32 μm). The number distributions were bimodal, with one peak below 50 nm and one centered around 125 nm, while the mass distribution had a peak around 10 μm . The slow SMPS scan time (76 s) may cause inaccuracy in the PSD, and the conversion from optical to aerodynamic diameter is affected by the particle's optical properties, morphology, and density [21,22]. Several other studies have estimated PSDs from image analysis [2,11]. However, this method is prone to multiple artifacts, such as particle loss during collection and sample preparation, a compromise between field of view and size resolution, and potential overlaps between particles [23].

Emission factors (EFs) are commonly used in air quality management to estimate pollutant emissions based on emission activities [24]. Because the total emission amount depends on LIB SOC and the number of LIBs burned, the proper activity metric for estimating LIB fire emissions is the total energy capacity of the LIB cells burned [25]. Therefore, the particle number EFs are expressed in particles/Wh (watt-hours), and the size-segregated PM mass and $\text{PM}_{2.5}$ constituent EFs are expressed in mg/Wh. EFs in these units are easier to use for estimating the total emissions than the emission rates in particles/hour reported by Premnath et al. [16]. To our knowledge, no studies have reported EFs of LIB fire particle chemical constituents.

In companion with the particle chemical characteristics presented by Claassen et al. [26], the objectives of this paper are to: (1) characterize the size distribution of particles in the diameter range of 6 nm–10 μm ; (2) quantify the EFs of particle number, particle mass, and $\text{PM}_{2.5}$ constituents emitted from LIB combustion; and (3) evaluate the dependence of PSDs and EFs on cell type and SOC.

2. Materials and Methods

The experiments and data analysis have been described in the companion paper [26] and only a brief description is provided here. Two commonly used LIB types were tested: a cylindrical, 18650-style, LFP cell and a pouch-style lithium cobalt oxide (LCO) cell [27,28]. Each cell type was tested at five SOC levels with three to six tests per SOC. The cells used

for these tests were purchased shortly before the experimental campaign and were received at proper storage voltages. It was found that the LFP cells only vented at 0% and 30% SOC, flamed at 50% and 75% SOC, and had either venting or flaming at 100% SOC. In contrast, LCO tests flamed at all SOC [26]. While these two cell types will be referred to by their cell chemistry (LFP vs. LCO), they also differ in form-factor and casing, both of which can affect emissions. However, this study did not attempt to isolate the effects of these factors. The LIB combustion tests were conducted inside an $\sim 8 \text{ m}^3$ burn chamber [29]. The exhaust flow rate (Q) was set to $\sim 3300 \text{ L/min}$, as determined by thermal anemometer measurements in the exhaust duct [30]. Before each test, the chosen LIB cell was charged to the desired SOC by a programmable charger. The cell was then placed in a ceramic crucible and heated to failure by an electric hot plate. A type K thermocouple was used to measure external cell and flame temperatures.

A sample of LIB combustion emissions was extracted from the chamber exhaust vent and directed to a suite of particle analyzers and the filter sampling system. Real-time particle mass concentrations were measured by two DustTrak DRX (TSI Inc., Shoreview, MN, USA) aerosol monitors in five size fractions (i.e., PM_{10} , $\text{PM}_{2.5}$, PM_4 , PM_{10} , and PM_{15}) [21,22]. The DustTraks were placed before and after sample line dilution to determine the dilution factors.

Finer resolution PSDs in the range of 6 nm – $10 \text{ }\mu\text{m}$ were measured every second by an electrical low-pressure impactor (ELPI+; Dekati Ltd., Kangasala, Finland) [31–34] which was placed on a diluted sample line to prevent impactor overloading. In an ELPI+, particles are first introduced into a unipolar diffusion charger to achieve a stable charge distribution, and then are collected on 13 stages of electrically conducting cascade impactors and a final filter, depending on their aerodynamic diameters. The electrical charges carried by the deposited particles are measured as current by sensitive electrometers. An inversion algorithm is used to convert the 14-channel electric current data to particle number distributions with up to 500 bins. In this study, the particle number distributions were converted to mass distributions by assuming spherical particle shape and a density of 1 g/cm^3 . The high-resolution data were integrated over the size fractions of $<0.1 \text{ }\mu\text{m}$ ($\text{PM}_{0.1}$), 0.1 – $1 \text{ }\mu\text{m}$ ($\text{PM}_{0.1-1}$), 1 – $2.5 \text{ }\mu\text{m}$ ($\text{PM}_{1-2.5}$), and 2.5 – $10 \text{ }\mu\text{m}$ ($\text{PM}_{2.5-10}$). There are multiple potential artifacts (e.g., deposition of smaller particles at upper stages, size-dependent particle density, and image current) that could lead to errors in the mass concentration calculation [35]. Therefore, the ELPI $\text{PM}_{2.5}$ mass was normalized by gravimetric $\text{PM}_{2.5}$ mass to account for these artifacts.

Two filter channels were used to collect $\text{PM}_{2.5}$ and acidic gases for analysis of mass by gravimetry, ions and acidic gases by ion chromatography (IC), organic and elemental carbon (OC and EC) by thermal/optical analysis, and elements by x-ray fluorescence (XRF) and inductively coupled plasma mass spectrometry (ICP-MS) [24,36]. The backup potassium hydroxide (KOH)-impregnated cellulose-fiber filters behind the Teflon-membrane front filters were analyzed for acidic gases, including hydrogen fluoride (HF), hydrochloric acid (HCl), nitric acid (HNO_3), and sulfuric acid (H_2SO_4) as their corresponding ions (i.e., F^- , Cl^- , NO_3^- , and SO_4^{2-}) by IC [37,38].

Emission factors (EF_i in particles/Wh for particle number and mg/Wh for mass) were calculated for each species i by Equation (1):

$$\text{EF}_i = C_i \times Q \times \Delta t / E \quad (1)$$

where C_i (in particles/ m^3 for number and mg/ m^3 for mass) is the mean stack concentration for species i , Q (in m^3/s) is the mean stack exhaust flow rate, Δt (in s) is the sampling duration, and E (in Wh) is the nominal LIB cell energy capacity.

3. Results

3.1. Particle Size Distributions

Figure 1 shows the heat maps of PSD evolution for the representative tests for each cell type and SOC. The y-axis represents particle aerodynamic diameter (D_p), and the coloring

represents particle number concentration normalized by the size bin width ($d\log(D_p)$). While most particles had diameters below 500 nm, significant variations in PSD evolution were seen. Low SOC (0 and 30%) LFP tests in this study, which had no flaming combustion, emitted high concentrations of particles over a wide size range, but concentrations never exceeded 10^7 particles/cm³. These emissions occurred for a long period of time, with maximum concentrations above 10^5 particles/cm³ for over 30 min for both low SOC.

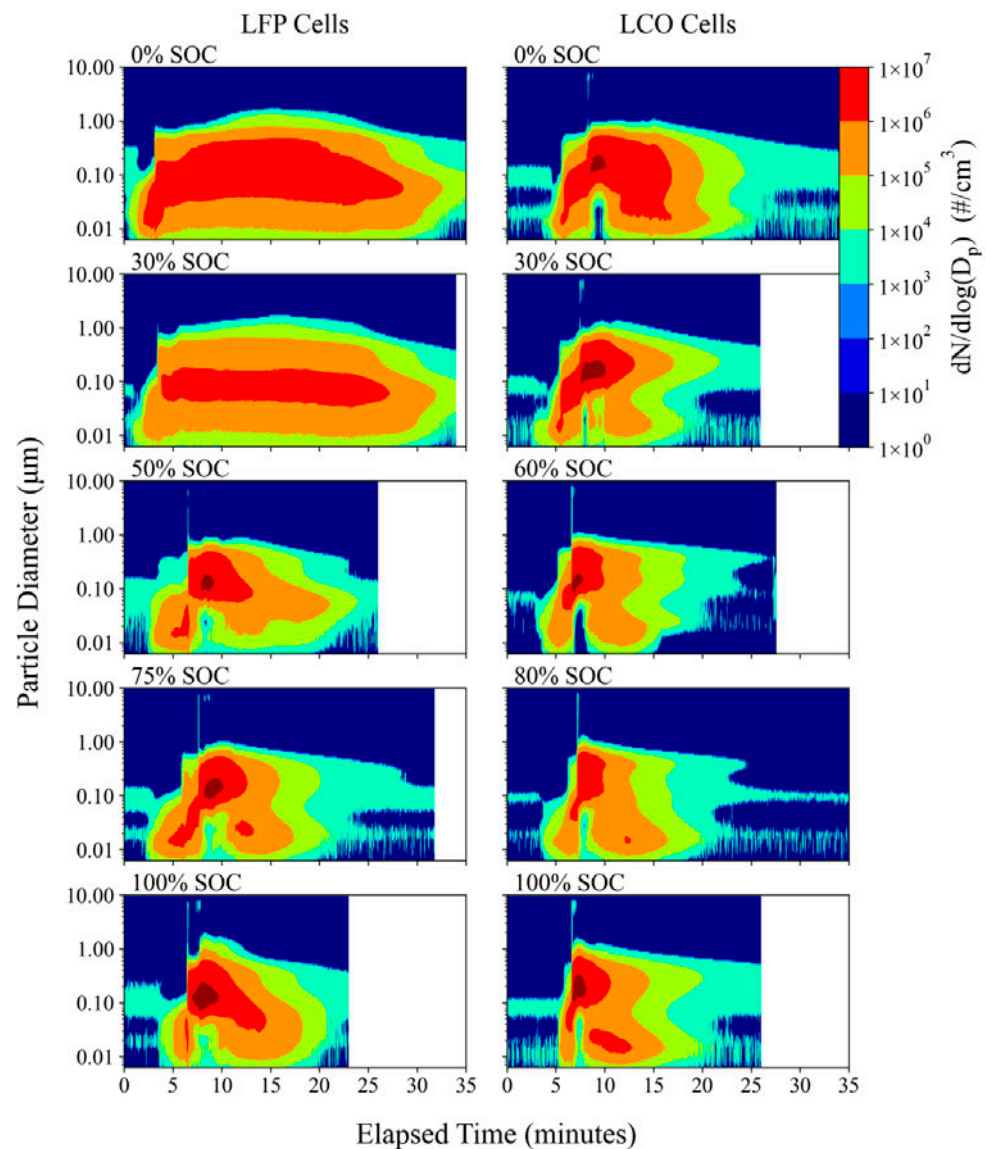


Figure 1. Particle number distribution heatmaps for representative LFP and LCO tests at each SOC.

Particle emissions for other LFP SOC and LCO tests reached peak concentrations above 10^7 particles/cm³, but concentrations above 10^5 particles/cm³ only lasted for 10–18 min. A PSD discontinuity is seen in Figure 1 for these tests, occurring between five and ten minutes of elapsed time and representing abrupt concentration increases and particle size changes. This event coincided with visual and temperature-based TR indicators, suggesting it was caused by the onset of TR. A short 5–10 s long emission of coarse particles (up to 10 μm) can be seen to occur simultaneously, likely corresponding to the ejection of spark particles. Maximum particle concentrations occurred just after TR began. This is where most of the particle mass was emitted as particle sizes peaked here as well, with a notable decrease in particles below 50 nm in diameter. After this, peak

concentrations and particle sizes decreased gradually, often forming a bimodal pattern of nanoparticles (<100 nm) and larger particles between 100 nm and 1 μm .

Figures 2 and 3 show individual PSD “snapshots” for a 0% SOC LFP test and a 60% SOC LCO test, each at four time intervals. The 0% SOC LFP test only vented, without flaming combustion, while the 60% LCO test flamed vigorously. A detailed discussion of the observed combustion behavior for all tests can be found in [26]. The intervals include background concentrations (t_1), when the test cell is outgassing (t_2), after TR (t_3), and after combustion is complete and the test cell is smoldering (t_4). These time intervals were selected to represent similar combustion stages despite the differences in combustion duration between tests. The GNMDs and geometric number standard deviations (GNSD) were calculated to characterize the PSD at each time interval [19].

Representative LFP Test (0% SOC)

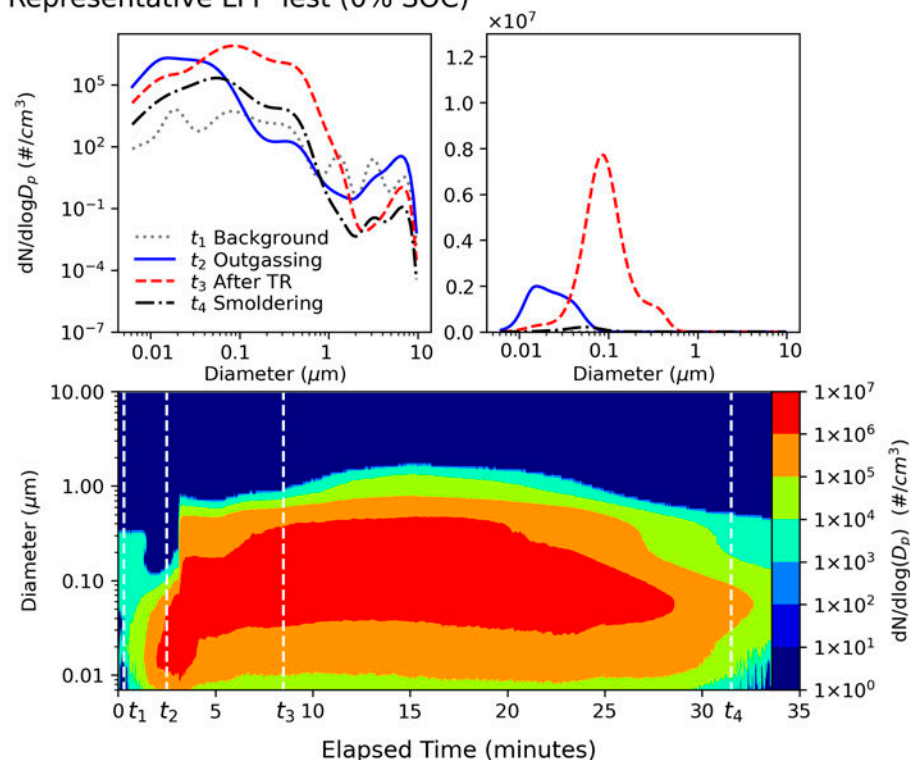


Figure 2. Particle number distribution snapshots (**left**: log scale and **right**: linear scale) and heatmap (**bottom**) for a representative 0% SOC LFP test. GNMDs for t_{1-4} are 59 nm, 22 nm, 91 nm, and 49 nm, respectively.

The 0% SOC LFP “venting” test and the 60% SOC LCO “flaming” test had similar peak outgassing (t_2) concentrations at $\sim 2 \times 10^6$ particles/ cm^3 but the particle diameters were smaller for the venting test, with a GNMD of 22 nm and a GNSD of 1.7. The flaming outgassing concentrations peaked at 59 nm with a GNSD of 1.9. While different cell types are compared here, the intent is to contrast the PSD differences due to the varied combustion behavior of the tests, not to infer differences due to the cell type itself. Concentrations after TR (t_3) peaked at four and six times the outgassing concentration for the venting and flaming tests, respectively, with the flaming test PSD being generally larger-sized and narrower. GNMDs at this time were 91 nm and 195 nm for the venting and flaming tests, with GNSDs of 1.9 and 1.6, respectively.

Figure 4a,b show the evolution of PM_{10} , $\text{PM}_{2.5}$, and PM_{10} for the tests shown in Figures 2 and 3, respectively. PM_{10} accounted for most of the emission mass. Large spikes of PM_{10} were observed at the onset of TR, corresponding to the emission of coarse particles ($\text{PM}_{2.5-10}$), as was discussed for Figure 1. Interestingly, small spikes of coarse particles also occurred just before TR and before $\text{PM}_{2.5}$ emissions began. This behavior may be

useful for early TR detection by LIB pack monitoring systems. The venting test (Figure 4a) shows $PM_{2.5}$ mass concentrations rising above PM_1 starting when emissions peaked around 15 min of elapsed time and lasting until combustion ended. This behavior occurred for all venting-only tests (LFP tests with 0% and 30% SOC as well as half of the 100% SOC tests [26]). For tests where flaming occurred, PM_1 dominated PM mass emissions outside of TR onset.

Representative LCO Test (60% SOC)

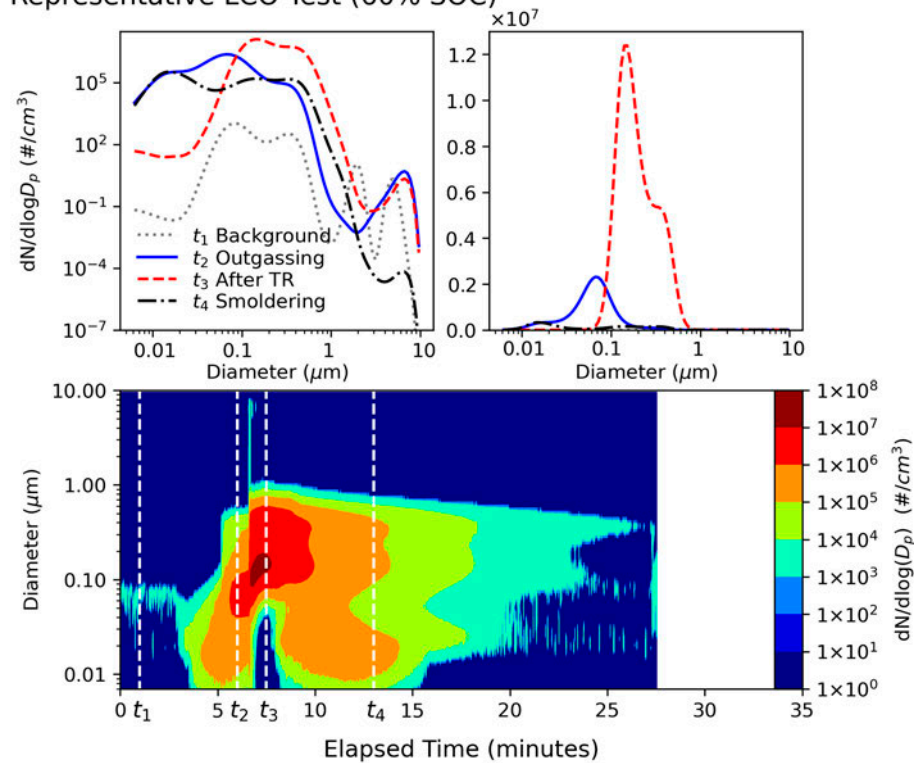


Figure 3. Particle number distribution snapshots (**left**: log scale and **right**: linear scale) and heatmap (**bottom**) for a representative 60% SOC LCO test. GNMDs for t_{1-4} are 117 nm, 59 nm, 195 nm, and 52 nm, respectively.

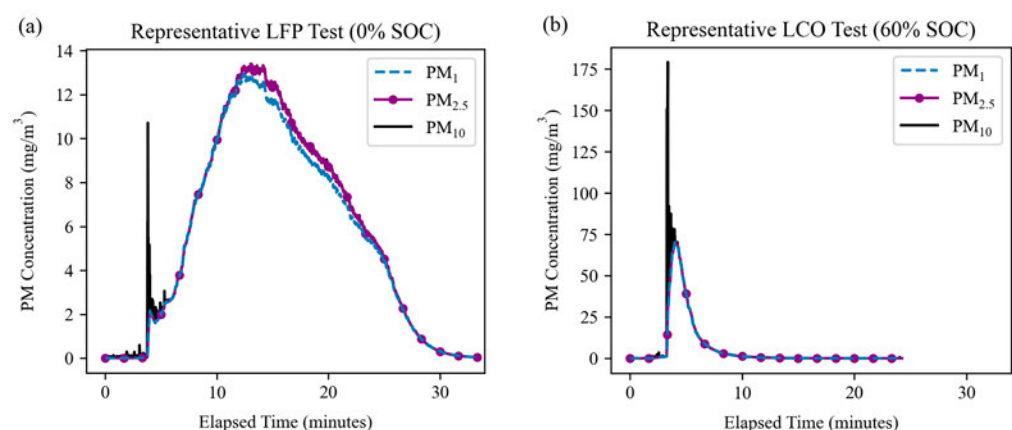


Figure 4. Mass concentrations of PM_1 , $PM_{2.5}$, and PM_{10} for the representative tests: (a) LFP at 0% SOC and (b) LCO at 60% SOC (same as those in Figures 2 and 3). The coarse particles ($PM_{2.5-10}$) are released predominantly during TR.

Figure 5 shows the average number-based (a and b) and mass-based (c and d) PSDs for each SOC and cell type. The number distributions show a dominant mode centered around 70–140 nm, with additional modes centered around 20 nm and 300 nm. These

distributions are similar to those reported by Goupil et al. [20]. For LFP tests, particles were smaller and were released in lower concentrations at low SOC, increasing in size and concentration at mid-SOCs. Tests with 100% SOC fell in the middle due to having disparate combustion behavior. This is consistent with the combustion behavior of LFP tests: they only vented at 0% and 30% SOC, intensely flamed at 50% and 75% SOC, and had more variable combustion at 100% SOC [26]. Particle sizes also increased with SOC for LCO tests, but there was no consistent dependence of concentrations on SOC, likely because flaming combustion was observed at all SOC. The mass distributions show that LFP tests with 0% and 30% SOC only have one mode centered around 460 nm, with elevated concentrations between 1 and 2.5 μm . All other tests had bimodal distributions, with one fine-particle mode centered around 400 nm and a coarse-particle mode centered around 8 μm . This indicates that the coarse particles were generated from flaming rather than cell venting. The coarse particle mode mass increased with SOC for both cell types, being insignificant for low SOC LFP tests. The fine particle mode mass also increased with SOC for LFP tests as the PSD shifted to larger particle sizes. A coarse mode centered around 10 μm was also observed by Goupil et al. [20].

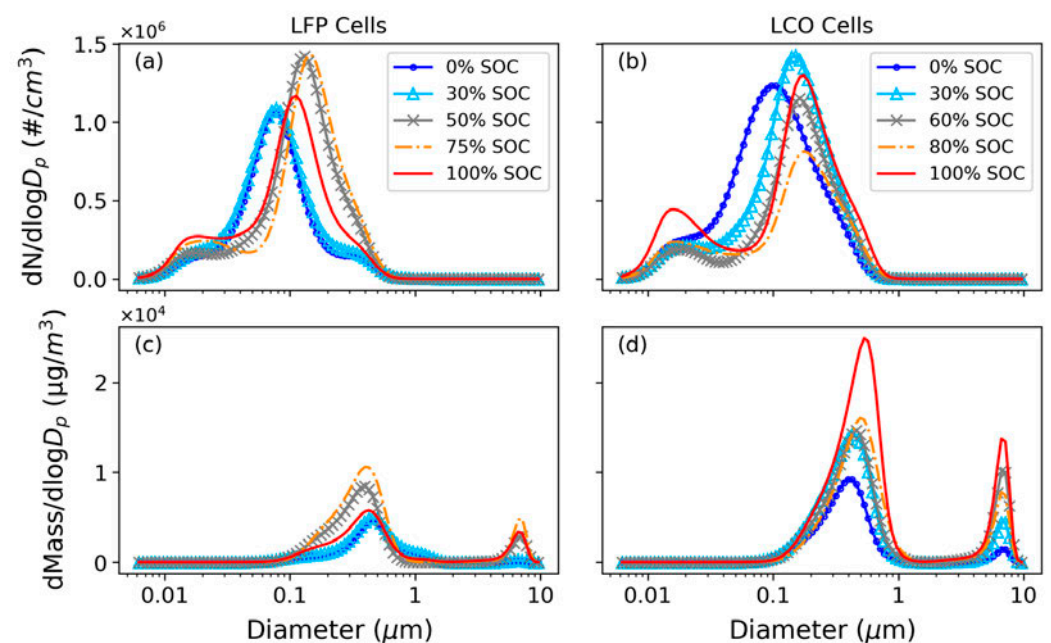


Figure 5. Particle number (top panels) and mass (bottom panels) distributions for LFP (a,c) and LCO (b,d) tests. An outlier was removed from some SOC groups to better show the prevailing trends.

Table 1 shows the GNMD and geometric mass mean diameters (GMMD) [19] for each cell type and SOC. The GNMD varied from 74 to 114 nm for LFP tests and 90 to 130 nm for LCO tests, showing that particles from LCO tests tended to be larger. GMMDs were larger than GNMDs due to mass being proportional to the cube of particle diameter. Interestingly, GNMDs peaked at mid-range SOC while GMMDs increased with SOC, peaking at 100%. This increase at high SOC, along with the previously discussed emission of large particles at TR, confirms that $\text{PM}_{2.5-10}$ is generated by the energetic combustion and ejection of material that occurs in a highly charged LIB cell during TR. Premnath et al. [16] found that GNMD and GMMD ranges were 54–69 nm and 97–204 nm for LFP and NMC cells, respectively, at 100% SOC. These diameters are smaller than those in Table 1, especially for GMMD, which is likely caused by the lower upper size limit of the EEPS (560 nm) [17,18] than the ELPI+ (10 μm).

Table 1. Average and range (in parentheses) of geometric number mean diameters (GNMD) and geometric mass mean diameters (GMMD) for each SOC and cell type.

Cell Type	SOC (%)	GNMD (nm)	GMMD (nm)
LFP	0	74 (72–77)	415 (409–424)
	30	74 (72–77)	426 (403–453)
	50	110 (109–111)	425 (415–435)
	75	114 (106–129)	479 (434–530)
	100	84 (71–93)	568 (536–611)
LCO	0	90 (78–102)	387 (371–405)
	30	120 (111–137)	474 (470–477)
	60	130 (128–132)	663 (553–751)
	80	118 (109–128)	673 (651–704)
	100	116 (96–129)	668 (602–778)

3.2. Emission Factors (EFs)

EFs for particle number, mass, PM_{2.5} constituents, and acidic gases are listed in Tables S1–S5. The following sections describe EFs for particle numbers, mass, carbon, PO₄^{3−}, selected metals and ions, and acidic gases, as well as the relationship between EF and combustion temperature.

3.2.1. EFs for Particle Number and Mass

Figure 6a shows particle number EFs by cell SOC for each size fraction. The EFs for ultrafine particles (PM_{0.1}) ranged between 1.5×10^{13} and 3.6×10^{13} particles/Wh for LFP tests and 3.2×10^{12} and 9.6×10^{12} particles/Wh for LCO tests (Table S1). Total particle numbers decreased with SOC, and LFP tests released several times more particles than LCO tests. PM_{0.1} and PM_{0.1–1} together accounted for >99.9% of PM₁₀ numbers, at 42–84% and 16–58% of PM₁₀, respectively. Particles larger than 1 μm had negligible contributions to particle number. PM_{0.1} tended to make up a higher proportion of PM₁₀ when particle emissions were high. On the other hand, Figure 6b shows PM_{0.1} made up a much smaller proportion of the PM₁₀ mass due to having very small particle volumes, at 1–8%, while PM_{0.1–1} dominated the PM₁₀ mass at 77–89% due to their high concentrations. PM₁ particles made up 81–95% of the measured PM₁₀ mass, as also shown in Figure 4. Larger particles made up significant portions of PM₁₀ mass despite their low number concentrations, with PM_{1–2.5} and PM_{2.5–10} accounting for up to 4% and 18% of PM₁₀ mass, respectively. PM_{1–2.5} EFs were high for low SOC LFP tests and venting-only 100% SOC LFP tests while all other tests had higher PM_{2.5–10} EFs instead.

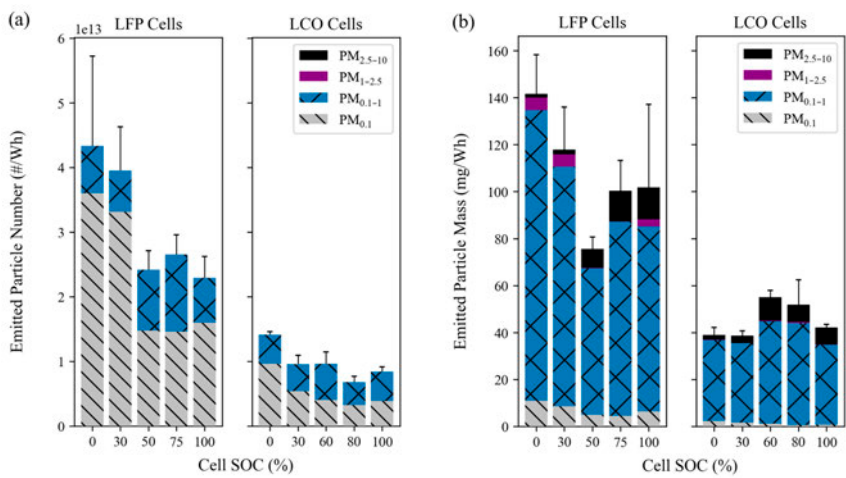


Figure 6. Emission factors for (a) particle number and (b) particle mass by size fraction for LFP and LCO tests. Error bars represent the total PM₁₀ standard error (including all smaller particle sizes) and are symmetric.

3.2.2. EFs for $\text{PM}_{2.5}$, OC, EC, PO_4^{3-} , and Toxic Metals

EFs for $\text{PM}_{2.5}$, OC, EC, and PO_4^{3-} are shown in Figure 7a. EFs for $\text{PM}_{2.5}$ mass were between 35 and 140 mg/Wh. This equates to the emission of 1.4–5.6 g of $\text{PM}_{2.5}$ from the combustion of a standard laptop-sized LIB (40 Wh), or 2.1–8.4 kg of $\text{PM}_{2.5}$ from the combustion of an electric vehicle battery pack (60 kWh) if all cells are burned in a similar way as those in this study.

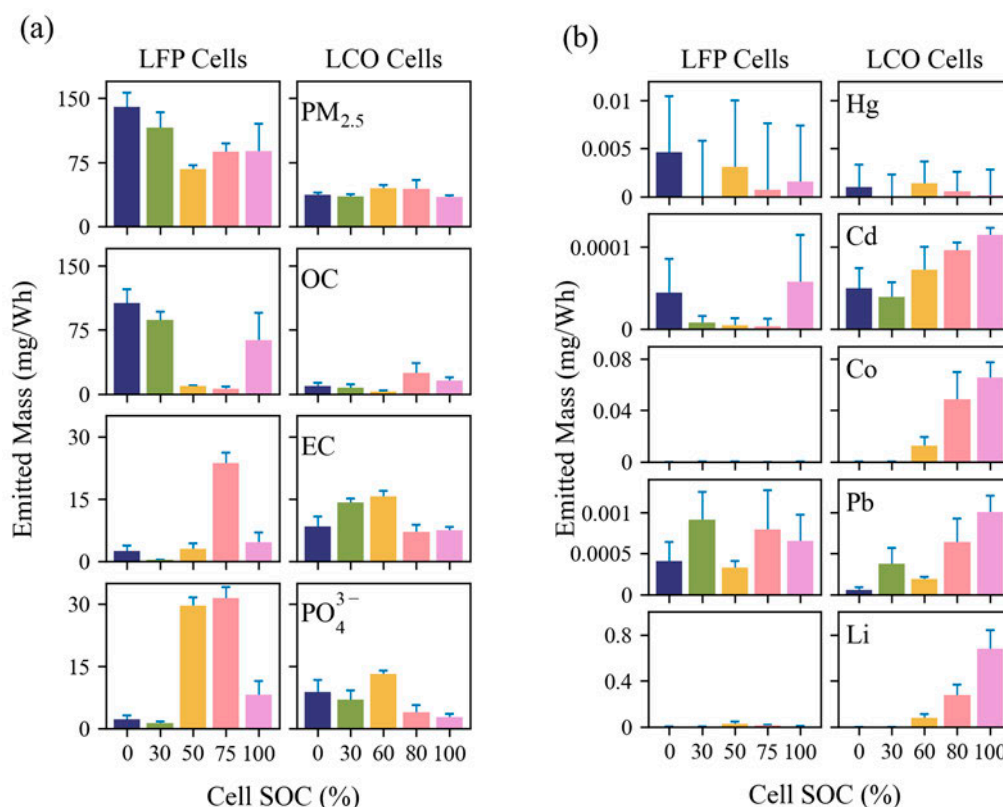


Figure 7. Emission factors for (a) $\text{PM}_{2.5}$ mass, OC, EC, and PO_4^{3-} and (b) selected metals. The error bars represent the larger of the propagated analytical uncertainty or the standard error within each SOC and are symmetric.

LFP tests had $\text{PM}_{2.5}$ EFs of 67–140 mg/Wh, with the least mass emitted at 50% SOC and increased variability at 100% SOC. OC EFs were 6–9 mg/Wh for 50–75% SOC and increased to 63–106 mg/Wh otherwise. OC variabilities at 100% SOC were similarly high to $\text{PM}_{2.5}$. Low OC emissions at mid-range SOC corresponded to high EC emissions for 75% SOC only, where EC EFs spiked from <5 mg/Wh for all other SOC to 24 mg/Wh. This resulted in total carbon EFs dropping from 109 mg/Wh (0% SOC) to 12 mg/Wh (50% SOC), before increasing back to 68 mg/Wh (100% SOC). PO_4^{3-} emissions were also significant for mid-range SOC, at approximately 30 mg/Wh, but only 1–8 mg/Wh otherwise.

LCO tests had more consistent $\text{PM}_{2.5}$ mass EFs of 35–45 mg/Wh, lower than LFP tests by 1.5–3.8 times. EFs for OC, EC, and PO_4^{3-} had much less dependence on SOC for LCO tests than LFP tests. OC EFs were 3–24 mg/Wh and peaked at 75% SOC, where LFP OC was lowest. EC EFs were 7–16 mg/Wh, higher than those of LFP tests except for at 75%/80% SOC. PO_4^{3-} EFs were 3–13 mg/Wh with a similar SOC dependence to EC.

Figure 7b shows EFs for four hazardous air pollutant metals [39] and Li. Variability within SOC was high but several features are notable. LCO tests had Cd, Co, Pb, and Li emissions that increase with SOC, reaching $(10 \pm 1) \times 10^{-5}$ mg/Wh for Cd, 0.07 ± 0.01 mg/Wh for Co, $(10 \pm 2) \times 10^{-4}$ mg/Wh for Pb, and 0.7 ± 0.2 mg/Wh for Li, all at 100% SOC. All of these elements were near detection limits at low SOC except for Cd. Only Pb was consistently present for LFP tests at between $(30 \pm 8) \times 10^{-5}$ mg/Wh

and $(9 \pm 3) \times 10^{-4}$ mg/Wh, with no dependence on SOC. Hg and Cd were detected sporadically. EFs for major compositions and elements are summarized in Tables S2 and S5, respectively.

3.2.3. EFs for Anions and Acidic Gases

EFs for acidic gases HF, HCl, HNO₃, and H₂SO₄ and their corresponding particulate anions are shown in Figure 8. These acidic gases are toxic and corrosive and can cause adverse effects on human health and material integrity if not neutralized after emission.

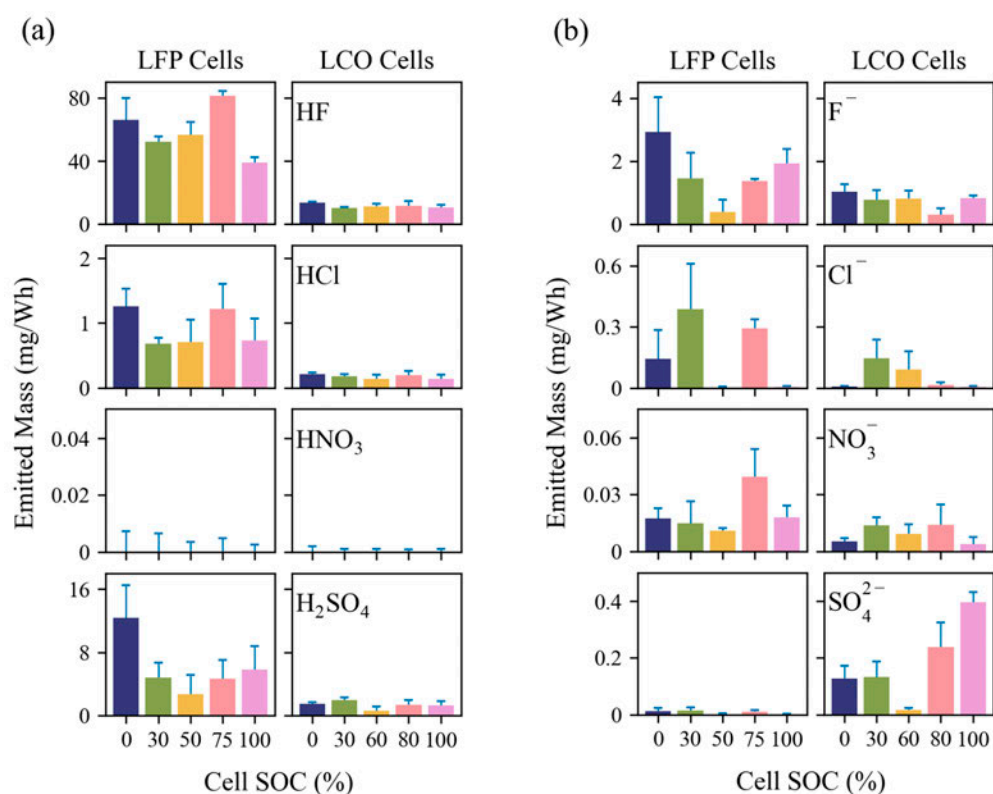


Figure 8. Emission factors for (a) selected acidic gases and (b) corresponding particulate anions. The error bars represent the larger of the propagated analytical uncertainty or the standard error within each SOC and are symmetric.

Gaseous EFs were higher than that of particulate emissions (by cell type) for all compounds shown except for NO₃⁻, where HNO₃ emission was not detected. Gaseous HF and particulate F⁻ had the highest EFs for all LIB types and SOC. Gaseous HF EFs ranged from 39 to 81 mg/Wh for LFP tests and 10 to 14 mg/Wh for LCO tests. Neither cell type showed a relationship between SOC and HF emissions. Particulate F⁻ EFs ranged from 0.4 to 3 mg/Wh for LFP tests and 0.3 to 1 mg/Wh for LCO tests. Particulate F⁻ had a distinct minimum at 50% SOC for LFP tests, corresponding to the minimum PM_{2.5} EF (Figure 7a). Gaseous H₂SO₄ was the second most emitted acidic compound, with much higher gaseous emissions than particulate. Gaseous EFs were 3–12 mg/Wh for LFP tests and 0.6–2 mg/Wh for LCO tests. Unlike gaseous H₂SO₄ and all other acidic emissions shown, particulate SO₄²⁻ emissions were much higher for LCO tests than LFP tests, where only trace concentrations were detected. LCO tests had EFs of 0.1–0.4 mg/Wh for most SOC, but only trace amounts for 60% SOC. Both gaseous and particulate emissions were low for 50/60% SOC, but this was more pronounced for LFP tests. Particle SO₄²⁻ emissions for LCO tests increased with SOC with the exception of 60% SOC. Gaseous HCl was consistently detected, averaging 0.9 mg/Wh for LFP tests and 0.2 mg/Wh for LCO tests. Particulate Cl⁻ was present sporadically for both cell types with EFs of up to 0.4 mg/Wh.

Only particulate NO_3^- was detected at between 0.005 and 0.04 mg/Wh, with higher emissions from LFP tests.

Table 2 compares the gaseous EFs of HF and HCl with the values reported in the literature. LFP HF emissions in this study (39–81 mg/Wh) compared well with the literature values which range at 12–350 mg/Wh. Emissions from LCO tests (10–14 mg/Wh) were lower than the 30–40 mg/Wh found in the literature, but this may be due to the limited number of studies [25,40]. Other cell types were reported to release a comparable 1–200 mg/Wh of HF. HCl emissions were reported by Diaz et al. [40] at 8–125 mg/Wh for LFP and LCO cells, significantly higher than those measured in this study (0.1–1.3 mg/Wh). No published EF values for HNO_3 or H_2SO_4 were found. Acidic gas and particulate ion EFs for all cell types and SOC levels are listed in Tables S3 and S4, respectively.

Table 2. Comparison of gaseous emissions of HF and HCl to the literature sources, grouped by cell chemistry. EFs for HNO_3 and H_2SO_4 were not found in the literature.

Source	Cell Type	HF (mg/Wh)	HCl (mg/Wh)
This study	LFP	39–81	0.7–1.3
[40]	LFP	350	125
[25]	LFP	12–24	
[41]	LFP	40–125	
This study	LCO	10–14	0.14–0.21
[40]	LCO	30	8
[25]	LCO	30–40	
[42]	Unknown	20–200	
[43]	LMO	40–70	
[44]	NMC/LFP	23–36	
[45]	NMC	1–10	
[46]	NMC	4.2	

3.2.4. Relationship between Emission Factors and Combustion Temperatures

Figure 9 shows the relationship between the EF and the maximum measured combustion temperature for LFP tests. Cell SOC is also labeled by symbol coloring to show groupings. With the exception of two outlier tests (Tests C1 and C8 in Figure 9a,b), EFs for $\text{PM}_{2.5}$ and OC were found to decrease with increasing combustion temperature. This is expected as higher temperatures indicate more complete combustion that will release more gaseous CO_2 instead of carbonaceous particulate. In contrast, PO_4^{3-} increased with combustion temperatures, indicating more efficient conversion of P-containing electrolyte and cathode materials to particle emissions. Low EC emissions were found at lower combustion temperatures as expected; however, EC was not necessarily higher at high temperatures. Figure 9c shows that the 50% SOC tests had temperatures $>600^\circ\text{C}$ but the EC emissions were comparable to those of low temperature tests. Particulate F^- and gaseous HF (Figure 9e,h) had the opposite trend, with F^- decreasing (for SOC levels $>30\%$) and HF increasing with increasing temperature. This is possibly due to more HF remaining in the gas phase at higher temperatures rather than condensing to form particles. Li^+ emissions were hardly detected below 600°C and increased rapidly thereafter, indicating a potential exponential relationship. More tests with temperatures reaching $>700^\circ\text{C}$ are needed to explore this. Figure 9g shows that the part of $\text{PM}_{2.5}$ mass with unidentified composition increased with temperature, a correlation that could help determine what additional components (e.g., unmeasured phosphorous compounds) [26] need to be analyzed for.

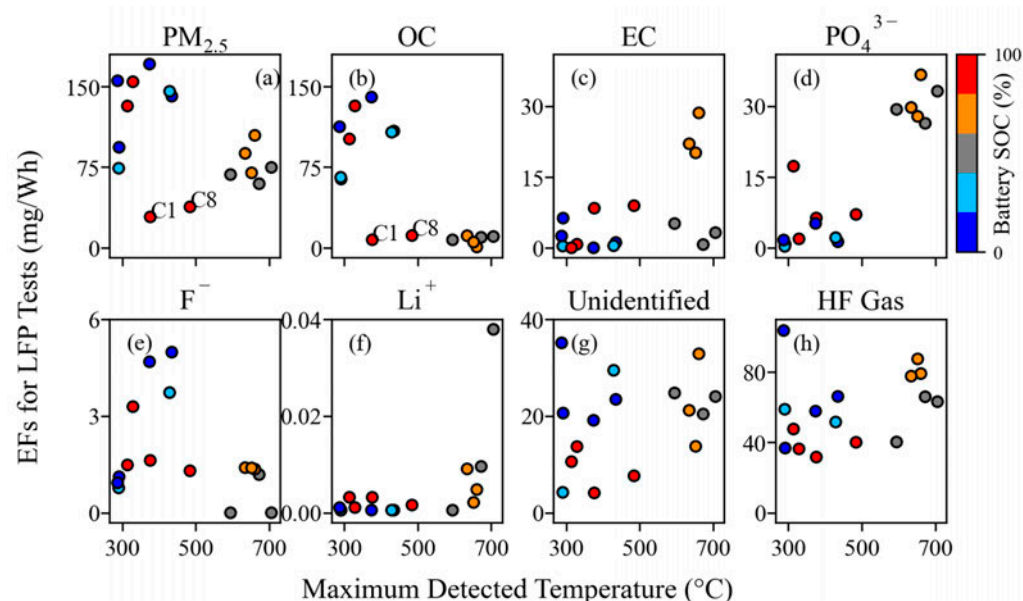


Figure 9. Relationship between EFs and maximum detected combustion temperature for LFP tests. LCO tests showed little correlation, possibly due to poor temperature measurement, and are not shown.

3.3. Cell Mass Losses

Each cell was weighed before and after combustion to determine the cell mass that was lost. LFP cells lost 2.2–2.4 g/Wh during combustion for all except 50% SOC, which lost noticeably less at 1.8–2 g/Wh. LCO cells lost roughly half the mass per Wh of cell capacity at 0.95–1.2 g/Wh. As shown in Table S6, this equates to a loss of ~20% of the cell mass for both cell types. However, only approximately 0.6–1.2% of the total cell mass, or 3–6% of the lost cell mass, was emitted as $PM_{2.5}$ that reached the filter assemblies. The proportion of lost cell mass that was emitted as $PM_{2.5}$ was highest for low SOC LFP tests, where 5–6% was converted to $PM_{2.5}$. All other SOC groups averaged 3–4%. By subtraction, this means that 94–97% of the lost mass, or roughly 16–22% of total cell mass, was emitted as either settleable PM or as gas emissions. As shown in Table S6, cell mass losses from previous studies range from 15% to 60%, with settleable particle mass accounting for 5–50% of the original cell mass, depending on the cell chemistry, casing, and SOC [5,6,10,47]. Even though $PM_{2.5}$ only accounted for a small fraction of the total PM mass emitted from LIB fires, it accounts for most of the particle numbers that can be inhaled by humans and presents great health risks.

4. Discussion and Conclusions

This study conducted a detailed characterization of the size distribution of particles with diameter less than 10 μm emitted from LIB fires and determined EFs for size-segregated particle number, particle mass, and $PM_{2.5}$ chemical constituents. The analysis covered two cell types at a range of SOC. While emissions may vary significantly depending on the specific cell types or LIB pack sizes, the cells tested represent two common cell chemistries and form-factors, in terms of both cell construction and energy capacity, that are used widely in modern LIB applications. As such, the findings offer valuable insights into the characteristics of emission from LIB combustion. The key conclusions are as follows:

- (1) LIB fires emit high concentrations of fine and ultrafine particles. The particle number distributions showed a dominant mode centered around 100 nm, with additional modes centered around 20 nm and 300 nm. The particle mass distributions showed that the venting-only LFP tests in this study with 0% and 30% SOC had one mode centered around 460 nm, while all other tests had bimodal distributions, with one fine particle mode centered around 400 nm and a coarse particle mode centered

around 8 μm . $\text{PM}_{0.1}$ and $\text{PM}_{0.1-1}$ together accounted for >99.9% of PM_{10} numbers, while $\text{PM}_{0.1-1}$ dominated the PM_{10} mass at 77–89%. Super-micron particles have non-negligible mass contributions, with $\text{PM}_{1-2.5}$ and $\text{PM}_{2.5-10}$ accounting for up to 4% and 18% of PM_{10} mass, respectively. $\text{PM}_{1-2.5}$ was emitted during cell venting, after TR and when flaming did not occur, while $\text{PM}_{2.5-10}$ was emitted just before and during TR onset for all tests.

- (2) Venting-only combustion (LFP tests at 0% SOC, 30% SOC, and some at 100% SOC) showed smooth evolution of PSDs, while the onset of TR caused a discontinuity in particle size distribution by increasing both mode diameter and concentration and releasing coarse mode particles (1–10 μm). Particles from LCO tests in this study (GNMD 90–130 nm; GMMD 387–673 nm) were generally larger than those from LFP tests (GNMD 74–114 nm; GMMD 415–568 nm). The GMMD increased with SOC.
- (3) LFP tests had higher EFs for particle number and mass than LCO tests. The EFs for ultrafine particles ($\text{PM}_{0.1}$) ranged from 1.5×10^{13} to 3.6×10^{13} particles/Wh for LFP and 3.2×10^{12} to 9.6×10^{12} particles/Wh for LCO tests. LFP tests had $\text{PM}_{2.5}$ EFs between 67 and 140 mg/Wh with more variability at different SOC, whereas LCO tests had lower and more consistent $\text{PM}_{2.5}$ mass EFs between 35 and 45 mg/Wh. Even though $\text{PM}_{2.5}$ only accounted for 0.25–1.5% of the total cell mass and a small fraction of total PM mass emitted from LIB fires, it accounts for the most particle numbers that can be inhaled by humans and presents great health risks.
- (4) LIB fires emit acidic gases, such as HF, HCl, and H_2SO_4 . Gaseous HF ranged from 39 to 81 mg/Wh for LFP tests and 10 to 14 mg/Wh for LCO tests. These toxic and corrosive gases may represent great hazards to people and properties.
- (5) Emissions are highly dependent on cell type, SOC, and combustion temperatures. The emitted $\text{PM}_{2.5}$ mass can depend on cell SOC by a factor of two, while the emitted OC, EC, and PO_4^{3-} can differ by a factor of 10 or more. PO_4^{3-} increased with combustion temperature, particularly when it reached >500 $^\circ\text{C}$. Toxic metal emissions increased with SOC, but only for LCO tests, with no trend for LFP tests. Acidic gas emissions depended primarily on cell type, indicating that cell design is crucial to lowering emissions of HF and other corrosive compounds. The emission dependence on LIB cell properties should be considered when evaluating the overall hazard that each LIB pack presents. For example, while LFP cells are known to be more thermally stable, results from this study show that they may release more HF and can generate higher particulate concentrations than LCO cells.

The emissions documented here have implications for health and safety as well as environmental contamination. Considering that modern LIB packs can contain hundreds to millions of Wh of energy capacity, the effects of significant emissions of volatile organics, ultrafine and fine particles, hazardous metals, and acidic compounds must be taken into consideration when responding to and cleaning up after LIB fires.

LIB fires are becoming more common as LIB use expands. The combustion process can occur quickly and without warning, limiting the effectiveness of fire suppression. Therefore, a detailed understanding of the resulting emissions is imperative to allow for effective fire response and cleanup. This paper has detailed the expected fine particulate emissions in a way that allows extrapolation to any size LIB pack if the burned LIB energy capacity is known. This resource will allow first responders, LIB manufacturers, and responsible authorities to plan for and respond to LIB fires.

Supplementary Materials: The following supporting information can be downloaded at: <https://www.mdpi.com/article/10.3390/batteries10100366/s1>, Table S1. EFs for four size fractions in particles/Wh and mg/Wh for particle number and particle mass, respectively; Table S2. EFs in mg/Wh for $\text{PM}_{2.5}$ chemical constituents; Table S3. EFs in mg/Wh for acidic gases for all cell types and SOC; Table S4. EFs in mg/Wh for particulate anions and cations for all cell types and SOC; Table S5. EFs in mg/Wh for elements for all cell types and SOC; Table S6. A comparison of cell mass loss rates to the literature values [5,6,10,47].

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Abbreviations

Δt	sampling duration
C	concentration
Cd	cadmium
Cl^-	chloride
Co	cobalt
D_p	particle diameter
E	battery cell energy capacity
EC	elemental carbon
EEPS	engine exhaust particle sizer
EF	emission factor
ELPI	electrical low-pressure impactor
F^-	fluoride
GMMD	geometric mass mean diameter: mean diameter of a particle mass distribution in logarithmic scale
GNMD	geometric number mean diameter: mean diameter of a particle number distribution in logarithmic scale
GNSD	geometric number standard deviation: standard deviation of a particle number distribution in logarithmic scale
HCl	hydrochloric acid
Hg	mercury
HF	hydrofluoric acid
HNO_3	nitric acid
H_2SO_4	sulfuric acid
IC	ion chromatography
ICP-MS	inductively coupled plasma mass spectrometry
KOH	potassium hydroxide
LCO	lithium cobalt oxide
Li	lithium
Li^+	lithium ion
LIB	lithium-ion battery
LFP	lithium iron phosphate
NMC	nickel manganese cobalt
NO_3^-	nitrate
OC	organic carbon
P	phosphorus
Pb	lead
PM	particulate matter
PM_x	particles with aerodynamic diameters $\leq x \mu\text{m}$

PO ₄ ^{3−}	phosphate
PSD	particle size distribution
Q	flow rate
SMPS	scanning mobility particle sizer
SO ₄ ^{2−}	sulfate
SOC	state of charge
TR	thermal runaway
Wh	watt hours
XRF	X-ray fluorescence

References

1. Bugryniec, P.J.; Resendiz, E.G.; Nwophoke, S.M.; Khanna, S.; James, C.; Brown, S.F. Review of gas emissions from lithium-ion battery thermal runaway failure—Considering toxic and flammable compounds. *J. Energy Storage* **2024**, *87*, 111288. [\[CrossRef\]](#)
2. Li, W.; Xue, Y.; Feng, X.; Rao, S.; Zhang, T.; Gao, Z.; Guo, Y.; Zhou, H.; Zhao, H.; Song, Z.; et al. Characteristics of particle emissions from lithium-ion batteries during thermal runaway: A review. *J. Energy Storage* **2024**, *78*, 109980. [\[CrossRef\]](#)
3. Xu, Y.; Wang, Y.; Chen, X.; Pang, K.; Deng, B.; Han, Z.; Shao, J.; Qian, K.; Chen, D. Thermal runaway and soot production of lithium-ion batteries: Implications for safety and environmental concerns. *Appl. Therm. Eng.* **2024**, *248*, 123193. [\[CrossRef\]](#)
4. Xu, Y.; Wang, Y.; Chen, D. Soot formation and its hazards in battery thermal runaway. *J. Aerosol Sci.* **2024**, *181*, 106420. [\[CrossRef\]](#)
5. Zhang, Y.; Wang, H.; Li, W.; Li, C.; Ouyang, M. Size distribution and elemental composition of vent particles from abused prismatic Ni-rich automotive lithium-ion batteries. *J. Energy Storage* **2019**, *26*, 100991. [\[CrossRef\]](#)
6. Zhang, Y.; Wang, H.; Li, W.; Li, C. Quantitative identification of emissions from abused prismatic Ni-rich lithium-ion batteries. *eTransportation* **2019**, *2*, 100031. [\[CrossRef\]](#)
7. Wang, H.; Zhang, Y.; Li, W.; Li, C.; Ouyang, M. Particles released by abused prismatic Ni-rich automotive lithium-ion batteries. *WSEAS Trans. Syst. Control* **2020**, *15*, 30–38. [\[CrossRef\]](#)
8. Wang, H.; Wang, Q.; Jin, C.; Xu, C.; Zhao, Y.; Li, Y.; Zhong, C.; Feng, X. Detailed characterization of particle emissions due to thermal failure of batteries with different cathodes. *J. Hazard. Mater.* **2023**, *458*, 131646. [\[CrossRef\]](#)
9. Wang, G.; Kong, D.; Ping, P.; Wen, J.; He, X.; Zhao, H.; He, X.; Peng, R.; Zhang, Y.; Dai, X. Revealing particle venting of lithium-ion batteries during thermal runaway: A multi-scale model toward multiphase process. *eTransportation* **2023**, *16*, 100237. [\[CrossRef\]](#)
10. Essl, C.; Golubkov, A.W.; Gasser, E.; Nachtebel, M.; Zankel, A.; Ewert, E.; Fuchs, A. Comprehensive Hazard Analysis of Failing Automotive Lithium-Ion Batteries in Overtemperature Experiments. *Batteries* **2020**, *6*, 30. [\[CrossRef\]](#)
11. Yang, Y.; Fang, D.; Maleki, A.; Kohzadi, S.; Liu, Y.; Chen, Y.; Liu, R.; Gao, G.; Zhi, J. Characterization of Thermal-Runaway Particles from Lithium Nickel Manganese Cobalt Oxide Batteries and Their Biotoxicity Analysis. *ACS Appl. Energy Mater.* **2021**, *4*, 10713–10720. [\[CrossRef\]](#)
12. Held, M.; Tuchschild, M.; Zennegg, M.; Figi, R.; Schreiner, C.; Mellert, L.D.; Welte, U.; Kompatscher, M.; Hermann, M.; Nachev, L. Thermal runaway and fire of electric vehicle lithium-ion battery and contamination of infrastructure facility. *Renew. Sustain. Energy Rev.* **2022**, *165*, 112474. [\[CrossRef\]](#)
13. Bordes, A.; Papin, A.; Marlair, G.; Claude, T.; El-Masri, A.; Durussel, T.; Bertrand, J.-P.; Truchot, B.; Lecocq, A. Assessment of Run-Off Waters Resulting from Lithium-Ion Battery Fire-Fighting Operations. *Batteries* **2024**, *10*, 118. [\[CrossRef\]](#)
14. ICRP. Human Respiratory Tract Model for Radiological Protection. ICRP Publication 66. Ann. ICRP 24 (1-3). 1994. Available online: <https://www.icrp.org/publication.asp?id=icrp%20publication%2066> (accessed on 2 July 2024).
15. Thangavel, P.; Park, D.; Lee, Y.C. Recent Insights into Particulate Matter (PM(2.5))-Mediated Toxicity in Humans: An Overview. *Int. J. Env. Res. Public Health* **2022**, *19*, 7511. [\[CrossRef\]](#) [\[PubMed\]](#)
16. Premnath, V.; Wang, Y.; Wright, N.; Khalek, I.; Uribe, S. Detailed characterization of particle emissions from battery fires. *Aerosol Sci. Technol.* **2022**, *56*, 337–354. [\[CrossRef\]](#)
17. Wang, X.L.; Grose, M.A.; Avenido, A.; Stolzenburg, M.R.; Caldow, R.; Osmondson, B.L.; Chow, J.C.; Watson, J.G. Improvement of Engine Exhaust Particle Sizer (EEPS) size distribution measurement—I. Algorithm and applications to compact-shape particles. *J. Aerosol Sci.* **2016**, *92*, 95–108. [\[CrossRef\]](#)
18. Wang, X.L.; Grose, M.A.; Caldow, R.; Osmondson, B.L.; Swanson, J.J.; Chow, J.C.; Watson, J.G.; Kittelson, D.B.; Li, Y.; Xue, J.; et al. Improvement of Engine Exhaust Particle Sizer (EEPS) size distribution measurement—II. Engine exhaust particles. *J. Aerosol Sci.* **2016**, *92*, 83–94. [\[CrossRef\]](#)
19. Hinds, W.C. *Aerosol Technology, Properties, Behavior, and Measurement of Airborne Particles*, 2nd ed.; Wiley: Los Angeles, CA, USA, 1999.
20. Goupil, V.; Gaya, C.; Boisard, A.; Robert, E. Effect of the heating rate on the degassing and combustion of cylindrical Li-Ion cells. *Fire Saf. J.* **2022**, *133*, 103648. [\[CrossRef\]](#)
21. Wang, X.L.; Chancellor, G.; Evenstad, J.; Farnsworth, J.E.; Hase, A.; Olson, G.M.; Sreenath, A.; Agarwal, J.K. A Novel Optical Instrument for Estimating Size Segregated Aerosol Mass Concentration in Real Time. *Aerosol Sci. Technol.* **2009**, *43*, 939–950. [\[CrossRef\]](#)

22. Wang, X.L.; Zhou, H.; Arnott, W.P.; Meyer, M.E.; Taylor, S.; Firouzkouhi, H.; Moosmüller, H.; Chow, J.C.; Watson, J.G. Evaluation of gas and particle sensors for detecting spacecraft-relevant fire emissions. *Fire Saf. J.* **2020**, *113*, 102977. [\[CrossRef\]](#)
23. Lee, J.; He, S.; Song, G.; Hogan, C.J. Size distribution monitoring for chemical mechanical polishing slurries: An intercomparison of electron microscopy, dynamic light scattering, and differential mobility analysis. *Powder Technol.* **2022**, *396*, 395–405. [\[CrossRef\]](#)
24. Wang, X.L.; Firouzkouhi, H.; Chow, J.C.; Watson, J.G.; Carter, W.; De Vos, A.S. Characterization of gas and particle emissions from open burning of household solid waste from South Africa. *Atmos. Chem. Phys.* **2023**, *23*, 8921–8937. [\[CrossRef\]](#)
25. Larsson, F.; Andersson, P.; Blomqvist, P.; Mellander, B.-E. Toxic fluoride gas emissions from lithium-ion battery fires. *Sci. Rep.* **2017**, *7*, 10018. [\[CrossRef\]](#) [\[PubMed\]](#)
26. Claassen, M.; Bingham, B.; Chow, J.C.; Watson, J.G.; Wang, Y.; Wang, X. Characterization of Lithium-Ion Battery Fire Emissions—Part 1: Chemical Composition of Fine Particles (PM_{2.5}). *Batteries* **2024**, *10*, 301. [\[CrossRef\]](#)
27. Lithiumwerks. APR18650M1B Nanophosphate® Technology Data Sheet. Austin, Texas. 2023. Available online: <https://lithiumwerks.com/products/lithium-ion-18650-cells/> (accessed on 2 July 2024).
28. AA Portable Power Corp. Model 544792 Polymer Lithium-Ion Battery Data Sheet. Available online: <https://www.batteryspace.com/prod-specs/3175.pdf> (accessed on 2 July 2024).
29. Tian, J.; Chow, J.C.; Cao, J.; Han, Y.; Ni, H.; Chen, L.-W.A.; Wang, X.; Huang, R.; Moosmu, H.; Watson, J.G. A biomass combustion chamber: Design, evaluation, and a case study of wheat straw combustion emission tests. *Aerosol Air Qual. Res.* **2015**, *15*, 2104–2114. [\[CrossRef\]](#)
30. U.S. EPA. Method 1—Sample and Velocity Traverses for Stationary Sources. Code of Federal Regulations, Title 40, Part 60, Appendix A-1. Available online: <https://www.ecfr.gov/current/title-40/chapter-I/subchapter-C/part-60/appendix-Appendix%20A-1-%20to-%20Part%2060> (accessed on 6 August 2024).
31. Järvinen, A.; Aitomaa, M.; Rostedt, A.; Keskinen, J.; Yli-Ojanperä, J. Calibration of the new electrical low pressure impactor (ELPI+). *J. Aerosol Sci.* **2014**, *69*, 150–159. [\[CrossRef\]](#)
32. Saari, S.; Arffman, A.; Harra, J.; Rönkkö, T.; Keskinen, J. Performance evaluation of the HR-ELPI+ inversion. *Aerosol Sci. Technol.* **2018**, *52*, 1037–1047. [\[CrossRef\]](#)
33. Marjamäki, M.; Keskinen, J.; Chen, D.-R.; Pui, D.Y.H. Performance Evaluation of the Electrical Low-Pressure Impactor (ELPI). *J. Aerosol Sci.* **2000**, *31*, 249–261. [\[CrossRef\]](#)
34. Keskinen, J.; Pietarinen, K.; Lehtimäki, M. Electrical low pressure impactor. *J. Aerosol Sci.* **1992**, *23*, 353–360. [\[CrossRef\]](#)
35. Maricq, M.M.; Podsiadlik, D.H.; Chase, R.E. Size Distributions of Motor Vehicle Exhaust PM: A Comparison Between ELPI and SMPS Measurements. *Aerosol Sci. Technol.* **2000**, *33*, 239–260. [\[CrossRef\]](#)
36. Wang, X.; Firouzkouhi, H.; Chow, J.C.; Watson, J.G.; Ho, S.S.H.; Carter, W.; De Vos, A.S. Chemically speciated air pollutant emissions from open burning of household solid waste from South Africa. *Atmos. Chem. Phys.* **2023**, *23*, 15375–15393. [\[CrossRef\]](#)
37. Eldering, A.; Solomon, P.A.; Salmon, L.G.; Fall, T.; Cass, G.R. Hydrochloric acid: A regional perspective on concentrations and formation in the atmosphere of Southern California. *Atmos. Environ. Part A Gen. Top.* **1991**, *25*, 2091–2102. [\[CrossRef\]](#)
38. Sturges, W.T.; Harrison, R.M. The use of nylon filters to collect HCl: Efficiencies, interferences and ambient concentrations. *Atmos. Environ. (1967)* **1989**, *23*, 1987–1996. [\[CrossRef\]](#)
39. U.S. EPA. *Initial List of Hazardous Air Pollutants with Modifications*; Air Toxics Assessment Group, U.S. Environmental Protection Agency: Research Triangle Park, NC, USA, 2020. Available online: <https://www.epa.gov/haps/initial-list-hazardous-air-pollutants-modifications> (accessed on 6 August 2024).
40. Diaz, F.; Wang, Y.; Weyhe, R.; Friedrich, B. Gas generation measurement and evaluation during mechanical processing and thermal treatment of spent Li-ion batteries. *Waste Manag.* **2019**, *84*, 102–111. [\[CrossRef\]](#)
41. Andersson, P.; Blomqvist, P.; Lorén, A.; Larsson, F. *Investigation of Fire Emissions from Li-Ion Batteries*; SP Technical Research Institute of Sweden: Borås, Sweden, 2013.
42. Larsson, F. *Lithium-Ion Battery Safety-Assessment by Abuse Testing, Fluoride Gas Emissions and Fire Propagation*; Chalmers Tekniska Högskola: Göteborg, Sweden, 2017.
43. Ribière, P.; Grugeon, S.; Morcrette, M.; Boyanov, S.; Laruelle, S.; Marlair, G. Investigation on the fire-induced hazards of Li-ion battery cells by fire calorimetry. *Energy Environ. Sci.* **2012**, *5*, 5271–5280. [\[CrossRef\]](#)
44. Sturk, D.; Rosell, L.; Blomqvist, P.; Ahlberg Tidblad, A. Analysis of li-ion battery gases vented in an inert atmosphere thermal test chamber. *Batteries* **2019**, *5*, 61. [\[CrossRef\]](#)
45. Bordes, A.; Marlair, G.; Zantman, A.; Herreyre, S.; Papin, A.; Desprez, P.; Lecocq, A. New insight on the risk profile pertaining to lithium-ion batteries under thermal runaway as affected by system modularity and subsequent oxidation regime. *J. Energy Storage* **2022**, *52*, 104790. [\[CrossRef\]](#)
46. Willstrand, O.; Pushp, M.; Andersson, P.; Brandell, D. Impact of different Li-ion cell test conditions on thermal runaway characteristics and gas release measurements. *J. Energy Storage* **2023**, *68*, 107785. [\[CrossRef\]](#)
47. Koch, S.; Fill, A.; Birke, K.P. Comprehensive gas analysis on large scale automotive lithium-ion cells in thermal runaway. *J. Power Sources* **2018**, *398*, 106–112. [\[CrossRef\]](#)

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Characterization of Particulate Emissions From Thermal Runaway of Lithium-Ion Cells

Over the past decade, there has been a significant acceleration in the adoption of lithium-ion (Li-ion) batteries for various applications, ranging from portable electronics to automotive, defense, and aerospace applications. Lithium-ion batteries are the most used energy storage technologies due to their high energy densities and capacities. However, this battery technology is a potential safety hazard under off-nominal conditions, which may result in thermal runaway events. Such events can release toxic gaseous and particulate emissions, posing a severe risk to human health and the environment. Particulate emissions from the failure of two different cell chemistries—lithium iron phosphate (LFP) and nickel manganese cobalt oxide (NMC)—were studied. Experiments were conducted at multiple states of charge (SOC), and three repeats were performed at each SOC for each cell chemistry to examine the repeatability/variability of these events. Particulate emissions were characterized in terms of particulate matter mass (PM_{2.5}), black carbon, and particle number (PN)/size. Failure of a single cell led to a significant release of particulate emissions, with peak emission levels being higher at the higher SOC. A high level of variability was observed for a specific SOC for LFP cells, while NMCs exhibited relatively less variability. In general, much higher particulate emissions were observed for NMCs compared to LFPs at each SOC. For NMCs at 100% SOC, peak PN levels were $\sim 2.5 \times 10^{+09}$ particles/cc (part/cc), and black carbon levels were ~ 60 mg/m³. For LFPs at 100% SOC, peak PN levels were $\sim 9.0 \times 10^{+08}$ part/cc, and black carbon levels were 2.5 mg/m³.

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Keywords: lithium-ion batteries, thermal runaway, particulate emissions, energy storage

1 Introduction

Lithium-ion (Li-ion) batteries have significantly impacted our modern world, serving as a vital source of power for a variety of portable electronic devices and electric vehicles, as well as contributing to renewable energy storage systems [1]. Nevertheless, the widespread use of Li-ion batteries has raised concerns about their safety, specifically the risk of thermal runaway events [2]. Thermal runaway can occur due to various off-nominal conditions, including electrical, mechanical, and thermal abuse [3]. Under these conditions, due to the increased internal temperature of Li-ion cells, a series of exothermic chemical reactions are triggered, leading to catastrophic failures, fires, and even explosions of the flammable gases released during this event [4].

Thermal runaway presents a critical safety concern due to the potential venting of gases and the ejection of particulate matter. Characterizing these safety risks is important for various stakeholders, including emergency responders, health researchers, government regulators, and other stakeholders responsible for developing standards and building solutions that advance human safety and sustainability. Research has shown that various gaseous species such as carbon dioxide, carbon monoxide, hydrogen, alkanes, alkenes, acid gases (such as hydrofluoric acid [HF]), and volatile organic compounds (VOCs) are produced during thermal runaway events [5–9]. These gaseous emissions introduce immediate risks, notably the potential for flammability and fire hazards. Accumulation of combustible gases in confined spaces can lead to explosive conditions, increasing the likelihood of fires or explosions. Additionally, VOC emissions can contribute to air pollution, which can have long-term environmental consequences. Some gases such as HF, formaldehyde, nitric oxide, and toluene [9] are highly toxic to human health, evident by their very low immediately dangerous to life and health levels.

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While extensive research has been conducted in analyzing gases released during thermal runaway [10], investigation of particulate emissions during thermal runaway events is relatively less explored. Particulate matter emitted during thermal runaway can include materials like metals, metal oxides, carbonaceous soot, and organic chemicals. These emissions can transport toxic and hazardous materials, posing additional health and environmental risks [11]. Depending on their composition, morphology, concentration, and size distribution, these particles can have adverse health effects upon inhalation, contaminate soil and water sources, and impact ecosystems [12]. Moreover, these particles can include high-speed, high-temperature particles that can potentially ignite the flammable gases that are also released, further exacerbating the thermal runaway process [13]. Such particles could also influence and lead to other failure phenomena, such as high-voltage arc discharges, which could have serious consequences. Therefore, understanding and characterizing this ejecta are crucial for both comprehending the dynamics of thermal runaway and mitigating associated health, environmental, and fire risks. Multiple studies have observed the inherent variability of thermal runaway events [14,15]. It is to be noted that the variability in thermal runaway events can directly impact particulate emissions and release profiles during such occurrences.

A few recent studies have focused on characterizing particulate emissions during thermal runaway. The focus of all these studies has been on larger particulates ($>2.5\text{ }\mu\text{m}$ in size). Zhang et al. [16] conducted an experiment to analyze gaseous and solid particle emissions during thermal runaway of a commercial 50 Ah, 3.65 V nickel manganese cobalt (NMC) cell. Gas chromatography and mass spectrometry (GC-MS) were used to determine the composition of the gases, while particle size analysis, ion chromatography, and inductively coupled plasma mass spectrometry were used for particle emission analysis. They detected 30 elements, most with a size less than 0.5 mm in diameter and a median size of approximately 397 μm . Chen et al. [17] utilized Fourier transform infrared, X-ray diffraction analysis, and GC-MS techniques to characterize the composition of ejected powder during the thermal runaway of an 18650 Li-ion cell. Their findings revealed that the ejected powder mainly consisted of carbon, metals, metal oxides, and organic chemicals. Additionally, the study reported particle size distributions ranging from 8.49 to 300.00 μm in ambient air and 5.94 to 210.04 μm in an Argon atmosphere. Barone et al. [18] utilized scanning electron microscopy (SEM) and energy-dispersive X-ray spectroscopy to investigate the morphology and elemental composition of particulates emitted during thermal runaway incidents in lithium iron phosphate (LFP), NMC, and lithium cobalt oxide batteries. Yang et al. [19] utilized SEM, X-ray fluorescence, and laser particle size analysis to examine the morphology, determine chemical composition, and analyze the particle size distribution of particles emitted during thermal runaway events in NMC cathodes. Furthermore, they investigated the biotoxicity of these particles and compared it to the pristine cathode material. Their study revealed an immediate inhibition of bacterial respiratory activities and observed cell membrane damage following a 5-h exposure to emitted particles. Wang et al. [13] conducted a study investigating particle emissions during the thermal runaway of Li-ion cells with various NMC cathodes. This study examined elemental composition, particle size distribution, morphology, and crystal structure. Their study concluded that most of the particles emitted were less than 0.85 mm and contained elements with potential health and environmental risks. In general, very few studies reported in the literature investigated emissions of smaller particles from battery fires. These particles tend to remain airborne for much longer durations, thereby posing a risk to human health and the environment. Premnath et al. [20] examined fine (less than 2.5 μm) particulate emissions during thermal runaway events in LFP and NMC modules. Their study encompassed measurements of particulate matter mass (PM_{2.5}), real-time particle number and size, as well as real-time black carbon. Notably, their findings revealed emission rates that were 5–6 orders of magnitude higher than those observed in

the exhaust of a contemporary heavy-duty diesel engine. This study also demonstrated that a significant release of particles falls well within the respirable size range.

This research program was formulated to address the critical knowledge gap that exists in the literature in understanding fine particulate emissions from battery fires. This program involved examining these emissions from cell-level failure events at different states of charge (SOC) varying from 100% to 10%. Experiments were conducted in triplicates to understand the variability/repeatability of these emissions. Thermal abuse was applied to the test articles in a controlled manner to initiate thermal runaway. Particulate emissions were characterized in detail using multiple state-of-the-art measurement techniques to gain information on black carbon, particle number, and particulate matter emissions. Cells from two different cathode chemistries were studied for pure scientific data gathering and not for comparison between the two chemistries. The results obtained are specific to the cell design, size, and chemistry studied.

2 Experimental Methods

2.1 Test Articles and Test Matrix. This paper focuses on two cell types composed of LFP and NMC cathode chemistries. The NMCs were pouch type, and the LFPs were prismatic metal can type. Both cells considered in this study were fresh cells. The NMC cells were harvested from a module used in an automotive electric vehicle, while the LFP cells were extracted from a battery module used for applications such as energy storage in recreational vehicles, vans, boats, and golf carts. Additional details about the two cell types are listed in Table 1.

The test matrix is shown in Table 2. Three repeats were conducted for each cell type at each SOC. External heating was used as the abuse mechanism, where the test article was heated at a rate of 10 °C/min until the maximum rated temperature of the heater pads was reached (220 °C). Once the peak temperature was achieved, the temperature was held at 220 °C until thermal runaway occurred. The objectives of the test matrix were to examine the repeatability/variability of particulate emissions during thermal runaway and to understand the impact of cell SOC on these emissions. No suppression systems were engaged at any point during these tests, and the thermal runaway event was allowed to unfold naturally.

The LFP cell was instrumented with five thermocouples and four heater pads at various locations, as shown in Fig. 1. One additional

Table 1 Specification of the test articles

Cell type	Specification
LFP cells	Nominal capacity: 50 Ah Nominal voltage: 3.2 V Weight: 1129 g
NMC cell	Nominal capacity: 71 Ah Nominal voltage: 3.7 V Weight: 985 g

Table 2 Test matrix

Cell chemistry	Repeats	State of charge (%)
LFP	1 pilot test	100
LFP	3	100
LFP	3	50
LFP	3	30
LFP	3	10
NMC	1 pilot test	100
NMC	3	100
NMC	3	50
NMC	3	30
NMC	3	10

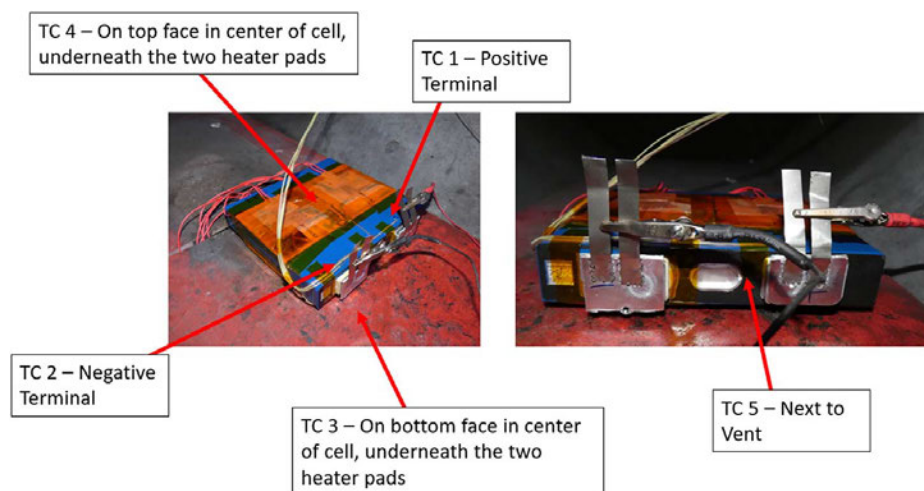


Fig. 1 LFP cell instrumentation and setup

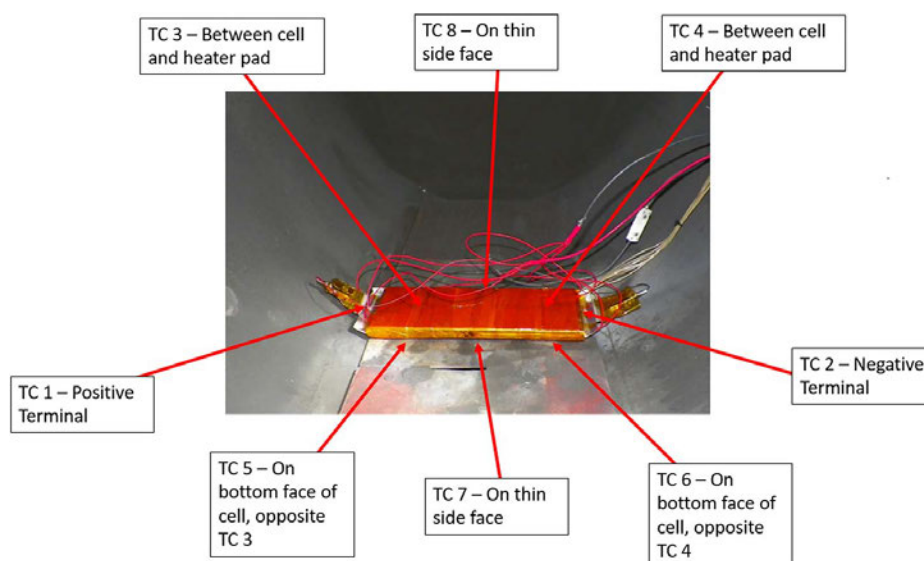


Fig. 2 NMC cell instrumentation and setup

Table 3 Cell conditioning protocol

Cell type	Charging protocol
LFP	Charge at 25 A until 3.6 V, then charge at 3.6 V until the current is less than 1 A Rest for 1 h Discharge at 25 A until 2.5 V Rest for 1 h Charge at 25 A until the desired SOC level based off discharge capacity calculated previously
NMC	Charge at 21.32 A until 4.2 V and then charge at 4.2 V until the current is less than 0.71 A Rest for 1 h Discharge at 35.5 A until 2.7 V Rest for 1 h Charge at 21.32 A until desired SOC level based off discharge capacity calculated previously

thermocouple was used to measure ambient temperature. The cell voltage was also monitored. Details for the NMC cell are shown in Fig. 2. The cell voltage was also monitored. Prior to each test, cells were charged to the test condition using the protocol shown in Table 3.

2.2 Test Chamber, Experimental Setup, and Emissions Analyzers. Testing was conducted in one of the test chambers at SwRI's Energy Storage Technology facility. The test chamber is equipped with a pollution abatement system that enables sampling of particulate and gaseous emissions from Li-ion battery fires. The

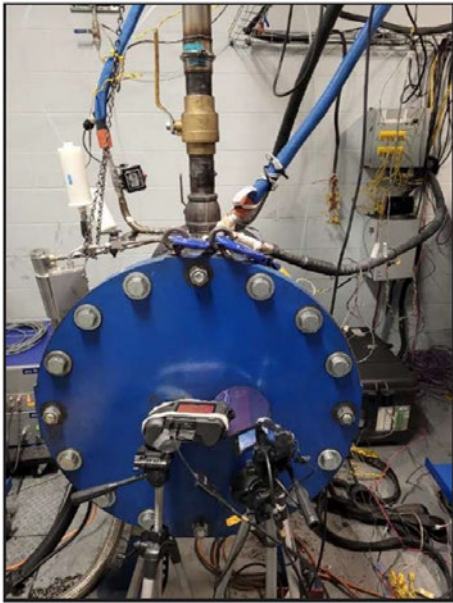


Fig. 3 Test chamber setup

test chamber and the exhaust piping from which particulate measurements were conducted are shown in Fig. 3. The chamber was sealed as much as possible to ensure that any effluents from the thermal runaway process passed through the exhaust piping at the top. Sample probes were installed in the pipe, and stainless-steel transfer lines were used to transport samples to the particulate analyzers. Good aerosol sampling practices were adopted to ensure minimal losses of particulates across the entire sample train from the probe to the instrument. Engineers and technicians controlled and observed the test outside the test cell to ensure personnel safety.

Figure 4 provides a schematic of the test setup. The particle instruments extracted samples from the pipe with a constant air/gas flowrate of ~ 34.8 lpm. Battery emissions were drawn into the exhaust pipe diluted by air from the ambient environment in the test cell. It is to be noted that sufficient oxygen was always available to sustain combustion.

Detailed characterization of particulate emissions was conducted using the methods described below. It is to be noted that the aerosol

sampling techniques used during this program were primarily to measure particles sub- $2.5\ \mu\text{m}$ in size—referred to as fine particulates. Battery fires result in the release of larger particulates as well. However, measurement of these emissions was beyond the scope of this program. Characterization of larger particulates requires careful design of isokinetic sample probes and low-loss sampling systems.

2.2.1 Combustion DMS500 Fast Particle Analyzer. The DMS500 is a real-time nanoparticle size spectrometer that simultaneously measures particle size distribution and number concentration [21]. The instrument can detect and measure particles from 5 nm to 1000 nm in size. The instrument operates on an electrical mobility (size to charge ratio) based classification scheme wherein it charges sample particles using a unipolar corona charger and, subsequently, classifies these particles using twenty-two electrometers based on their electrical mobility (a function of particle charge to size ratio). Signals from these electrometers are processed to generate spectral data on a real-time basis (1 Hz). The instrument also has a built-in dilution system that can be engaged to ensure that the electrometer channels do not get overwhelmed or saturated. For most of the campaign, a dilution setting of $\sim 1500:1$ was used.

2.2.2 AVL Micro-Soot Sensor. An AVL micro-soot sensor (MSS), which operates based on a photo-acoustic measurement scheme, was used to measure real-time black carbon concentration [22]. Sample particles are exposed to modulated laser light, which results in an increase in temperature of strongly absorbing black carbon particles. This, in turn, results in the heating of gas surrounding these particles, leading to the generation of sound waves that are detected by a sensitive microphone. This signal is proportional to the concentration of black carbon mass in the measurement chamber. The upper and lower limits of the MSS's detection capability are $50\ \text{mg}/\text{m}^3$ and $5\ \mu\text{g}/\text{m}^3$, respectively. The MSS includes an internal dilution system operated at a constant dilution ratio of 3:1.

2.2.3 Gravimetric Particulate Matter Mass Filter Measurement. A Sierra BG-3 partial flow sampling system was used to collect samples on TX40 media 47-mm diameter filters [23]. These filters were pre- and postweighed after sample collection to determine PM emission rate. The BG-3 includes a radial inflow porous stainless-steel dilution tunnel that helps dilute the sample with clean, particulate-free air with minimal losses. The ability to control sample dilution ensures that the filter loading

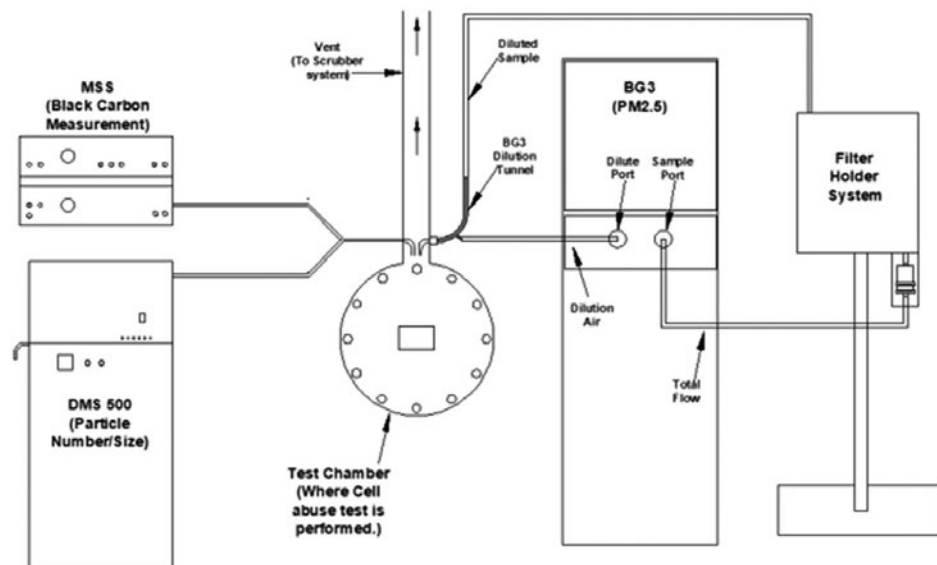


Fig. 4 Schematic of test chamber setup with particulate analyzers

Table 4 Summary of observations for the cell-level abuse test

Battery chemistry	SOC (%)	Round	Peak temperature (°C)	Fire?	Venting location	Pretest weight (g)	Posttest weight (g)
LFP	100	1	373	No	Vent between terminals	1131	920
LFP	100	2	293	No	Vent between terminals	1130	999
LFP	100	3	426	No	Vent between terminals	1127	919
LFP	50	1	271	No	Vent between terminals	1125	944
LFP	50	2	660	Yes	negative terminal	1126	941
LFP	50	3	303	No	back side, negative terminal side ^a	1130	946
LFP	30	1	244	No	back side, negative terminal side ^a	1130	957
LFP	30	2	242	No	Vent between terminals	1135	973
LFP	30	3	227	No	Vent between terminals	1133	997
LFP	10	1	230	No	Vent between terminals	1130	1014
LFP	10	2	231	No	Vent between terminals	1126	994
LFP	10	3	225	No	Vent between terminals	1127	1017
NMC	100	1	1300	Yes	Tabs	982	474
NMC	100	2	1300	Yes	Tabs	983	523
NMC	100	3	1300	Yes	Tabs	981	547
NMC	50	1	697	No	Tabs	980	783
NMC	50	2	754	No	Tabs	983	778
NMC	50	3	768	No	Tabs	981	788
NMC	30	1	576	No	Tabs	984	804
NMC	30	2	570	No	Tabs	982	805
NMC	30	3	561	No	Tabs	983	805
NMC	10	1	342	No	Tabs	982	833
NMC	10	2	384	No	Tabs	982	829
NMC	10	3	307	No	Tabs	981	839

^aInitial venting accompanied by sparks.

remains within acceptable levels, which would otherwise lead to the filter overloading and tearing (thereby voiding the measurement).

3 Results and Discussion

Results that include physical observations, black carbon emissions, particle number emissions and size signatures, and PM_{2.5} emissions will be discussed for all cell-level tests conducted as a part of this project. Table 4 summarizes observations made relative to the test article, and Table 5 summarizes particulate emissions data for all tests.

3.1 Physical Observations. All tests were executed using a similar process. Figure 5 serves as an example and provides information on how a specific test was executed from start to finish. Turning on the power to the heater was referred to as the start of the test. It took ~20 min from the start of the test until a peak temperature of 220 °C was reached (also the upper threshold temperature of the heater pads used). Instruments for detecting particulates were placed on stand-by mode once this temperature was reached. This mode prepared the instruments for sampling and led to ambient air being drawn over the test article, thereby resulting in a minor decrease in temperature, as highlighted in the figure. The instruments were placed on “sample mode” once thermal runaway occurred, confirmed visually by venting and by a rapid increase in test article temperatures. The power to the heater pads was turned off during the onset of thermal runaway. Particulate measurements were collected either until the instruments were saturated or until concentrations returned to ambient levels. Following which, a blower system was engaged to evacuate the test chamber and exhaust emissions outside the building after flowing through a filtration/scrubbing system. Engagement of the scrubbing system resulted in a large draw of ambient laboratory air over the test article, thereby cooling it further. The test article was monitored in the test chamber to cool down to temperatures below 50 °C for at least an hour prior to conducting postmortem analysis.

Tables 4 and 5 summarize the results of all the tests conducted in this program. For the LFP chemistry, peak temperatures were highest for 100% SOC, and then steadily decreased as SOC

decreased. At 100% SOC, the average peak temperature was 364 °C, and it decreased to an average of 229 °C at 10% SOC. During one of the 50% SOC tests for the LFP chemistry, a fire was observed during thermal runaway, causing a spike in temperature resulting in a peak temperature of ~660 °C (round 2, 50% SOC, LFP test). For most of the LFP testing, venting occurred through the designated vent in the cell. However, for a few tests with 50% and 30% SOC, venting also occurred through the negative terminal, or the back side of the cell, opposite the terminals. Although peak temperatures observed for the NMC cells were much higher, fire was only observed during the 100% SOC tests. At 100% SOC, the thermocouples were destroyed in the fire for the NMC cells, therefore showing temperatures peaking out at the thermocouple limit. As the SOC decreased to 50%, 30%, and 10%, the average peak temperatures decreased to 740 °C, 569 °C, and 344 °C, respectively. For all NMC tests, venting occurred through the seals near the tabs on the cells.

3.2 Black Carbon Emissions. Thermal runaway of both cell chemistries resulted in black carbon emissions. There could be multiple sources for black carbon generation from a Li-ion cell. Graphite, which is a stable form of carbon, is commonly used as the anode active material in Li-ion cells. Carbon additives may also be used in electrodes to enhance electrical conductivity. For events accompanied by combustion and fire, these measurements may also include the emission from combustion. Detailed material characterization of the test articles was beyond the scope of this article and will be included as a part of a future publication. Figure 6 provides real-time black carbon emissions profiles for LFP and NMC tests at various SOC. Data from one of the repeats at each SOC have been presented in this figure to serve as an example of the release profile after the onset of thermal runaway. Typically, NMC tests resulted in higher black carbon levels than LFP tests. For most tests, peak emission levels were observed close to the onset of thermal runaway. Subsequently, additional excursions in black carbon levels were observed depending on how the event unfolded. For instance, the 100% NMC test yielded a peak close to 40 mg/m³ early on, and a secondary release after about 75 s with a peak of ~4 mg/m³. In the case of the 30% LFP test, black carbon activity was observed for ~300 s from the onset of thermal runaway. Environmental

Table 5 Summary of particulate emissions

Battery chemistry	SOC (%)	Round	PM2.5 emissions (g/h)	Black carbon emissions (mg/h)	Particle number emissions (part/h)
LFP	100	Pilot	1.75	0.47	$1.43 \times 10^{+14}$
LFP	100	1	1.78	0.18	$1.63 \times 10^{+14}$
LFP	100	2	1.24	0.02	$1.80 \times 10^{+13}$
LFP	100	3	2.58	0.52	$2.22 \times 10^{+14}$
LFP	50	1	0.28	0.00	$9.71 \times 10^{+12}$
LFP	50	2	1.38	0.17	$9.41 \times 10^{+13}$
LFP	50	3	0.52	0.04	$4.96 \times 10^{+13}$
LFP	30	1	0.94	0.18	$8.96 \times 10^{+13}$
LFP	30	2	0.03	0.00	$5.74 \times 10^{+11}$
LFP	30	3	0.06	0.00	$1.07 \times 10^{+10}$
LFP	10	1	0.05	0.00	$2.03 \times 10^{+10}$
LFP	10	2	0.18	0.00	$4.52 \times 10^{+12}$
LFP	10	3	0.01	0.00	$4.83 \times 10^{+11}$
NMC	100	Pilot	6.73	4.70	$2.89 \times 10^{+14}$
NMC	100	1	3.23	4.15	$1.15 \times 10^{+14}$
NMC	100	2	3.91	4.36	$1.21 \times 10^{+14}$
NMC	100	3	6.49	7.61	$1.25 \times 10^{+14}$
NMC	50	1	2.84	1.35	$1.93 \times 10^{+14}$
NMC	50	2	3.70	1.20	$1.61 \times 10^{+14}$
NMC	50	3	4.05	4.08	$1.73 \times 10^{+14}$
NMC	30	1	3.66	0.28	$5.32 \times 10^{+13}$
NMC	30	2	1.12	0.03	$9.11 \times 10^{+12}$
NMC	30	3	1.26	2.44	$9.31 \times 10^{+13}$
NMC	10	1	0.13	0.00	$4.52 \times 10^{+12}$
NMC	10	2	0.01	0.00	$0.00 \times 10^{+00a}$
NMC	10	3	0.01	0.00	$0.00 \times 10^{+00a}$

^aMeasured PN concentrations were similar to ambient background levels.

Protection Agency (EPA) guidelines suggest that PM concentrations at or below $12 \mu\text{g}/\text{m}^3$ to be safe over a 24-hr exposure window. Levels above $500 \mu\text{g}/\text{m}^3$ or $0.5 \text{ mg}/\text{m}^3$ are considered to be hazardous to human health. Black carbon is a key component of PM, and peaks of the order of $60 \text{ mg}/\text{m}^3$ were observed during some tests.

Figure 7 summarizes observations made for both cell chemistries in terms of average black carbon emissions and peak black carbon emissions observed for all tests conducted. Average emissions have been reported for the first 200 seconds following the onset of a thermal runaway, as the significant release of emissions occurred during this window. LFP tests conducted at a specific SOC exhibited considerable variability, making it difficult to extract a black carbon release trend as a function of SOC. It is to be noted that the variability was intrinsic to the test article, and the abuse test protocol applied for each test was the same. In the case of NMC tests, nominally, higher SOC resulted in higher peak emissions and higher average black carbon emissions. 10% SOC tests did not yield any black carbon emissions for both cell chemistries as the cells did not experience violent venting or thermal runaway. Some of the 30% SOC tests yielded black carbon emissions for both chemistries. In fact, one of the repeats at 30% SOC for the

NMC chemistry resulted in a peak close to $7 \text{ mg}/\text{m}^3$, which is a significant amount of black carbon emissions.

3.3 Particulate Number Emissions. Particulate number emissions were measured using the DMS500 Fast Particle Sizer. This analytical technique detects all particulate phase emissions ranging from 5 nm to 1000 nm, and simultaneously provides number concentration and size distribution. Figure 8 provides real-time emissions for LFP and NMC tests conducted at different SOC. High levels of particulate number emissions were observed for both chemistries. Peak levels observed with NMC tests were higher than those of LFP tests conducted at the corresponding SOC. Ambient PN levels in a laboratory setting or inside a house in the US are usually between 2000 and 4000 particles/cc (part/cc). Peak levels observed during the NMC tests were of the order of $2 \times 10^{+9}$ part/cc, which is 6 orders of magnitude higher than ambient levels. LFP tests yielded peak concentrations close to $8 \times 10^{+8}$ part/cc, which is also close to 6 orders of magnitude higher than ambient levels. Typically, the initial release during the onset of thermal runaway resulted in peak PN release levels. During some tests, high PN levels were observed even after the

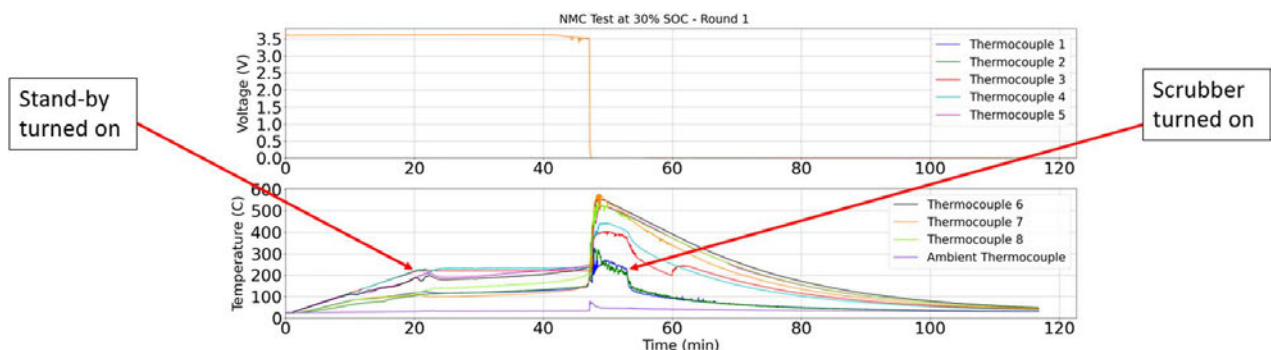


Fig. 5 Example plot outlining test sequence

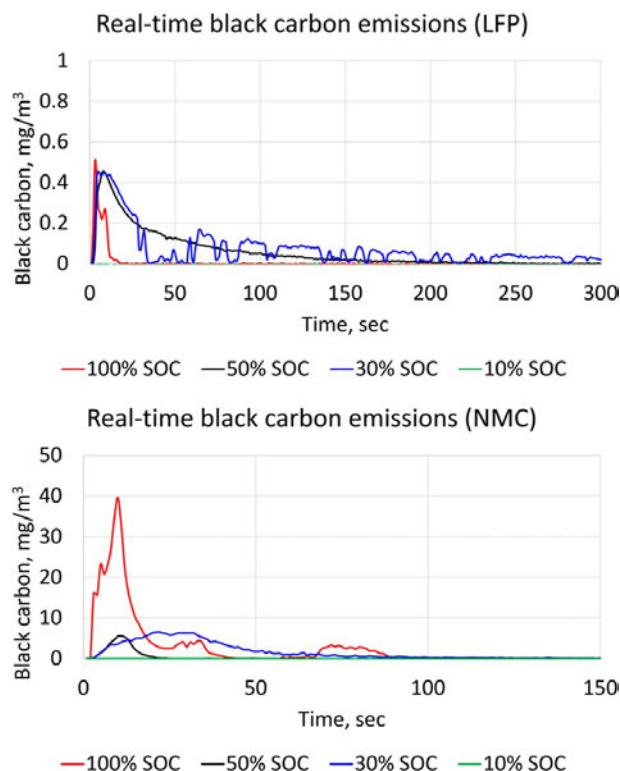


Fig. 6 Real-time profile of black carbon emissions

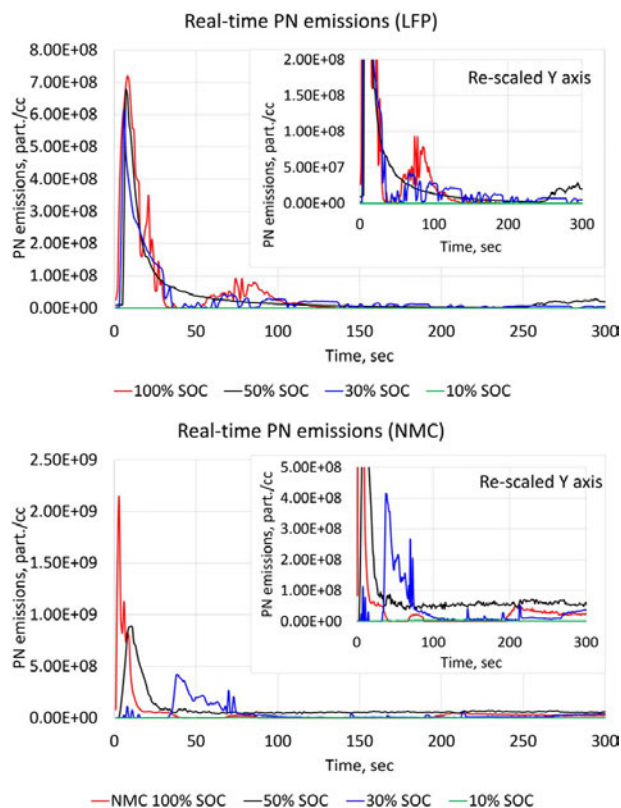


Fig. 8 Real-time profile of particle number emissions

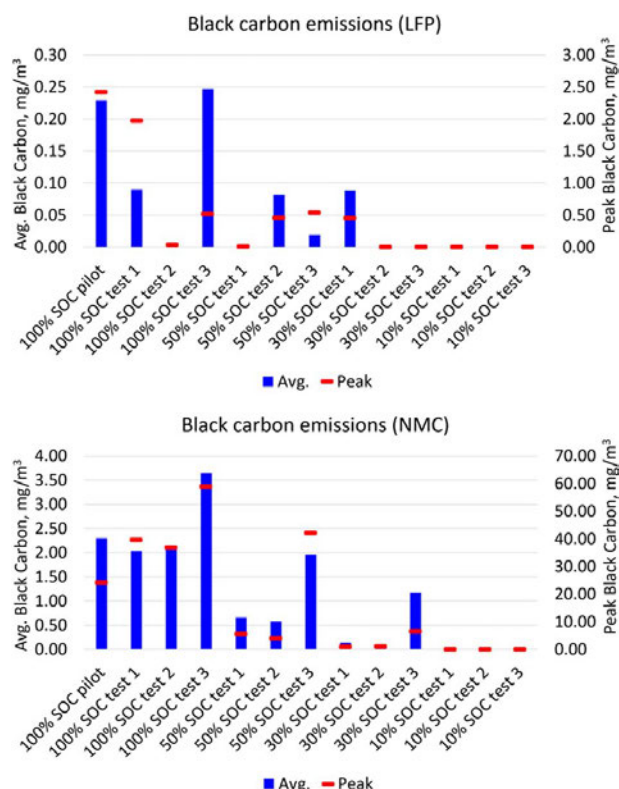


Fig. 7 Average and peak black carbon emissions for all tests

initial release event. It is to be noted that these high PN emissions were from the thermal runaway of a single cell. In real-world applications involving modules and packs composed of multiple cells, these emissions could scale dramatically if propagation of thermal runaway and failure occurs.

Figure 9 summarizes information for all tests conducted for each cell chemistry at various SOC levels. Similar to black carbon analysis, the average concentration observed over the first 200 s from the onset of thermal runaway has been presented. As observed with black carbon emissions, LFP tests exhibited variability in PN emissions as well as specific SOC levels. Interestingly, peak PN levels observed during some of the 100%, 50%, and 30% SOC tests were in a similar range of $\sim 8 \times 10^8$ part/cc. No significant PN emissions were observed during the 10% SOC tests. NMC tests resulted in higher PN emissions than LFP tests. In general, peak PN levels decreased with the decrease in SOC, although there were a couple of outliers. While no significant black carbon emissions were observed at this SOC. Further, average PN emissions observed during the first 200 s following the onset of thermal runaway were higher for the 50% SOC tests compared to the 100% SOC tests. This was largely due to the release profiles, as shown in Fig. 9. The 100% NMC tests resulted in a very high PN level during the initial release immediately after the onset of thermal runaway, after which PN returned to lower levels. In the case of 50% NMC tests, PN levels remained high even after the initial peak event, as shown in Fig. 8.

3.4 Particulate Matter Mass Emissions. Thermal runaway results in the release of particulate matter composed of various species, including carbonaceous soot, metals, carbonates, and various hydrocarbons, among others. The experimental technique used to characterize PM_{2.5} emissions from cell abuse involved sampling all ejected constituents on borosilicate glass fiber filters. These filters have very high collection efficiencies and serve as a sensitive technique to better understand thermal runaway emissions. Unlike the other two measurements described in the previous sections, this technique does not generate real-time information. Instead, this batch sampling method was used to determine PM_{2.5} emissions throughout the entire event. For this program, sample collection on the filter media was performed using a

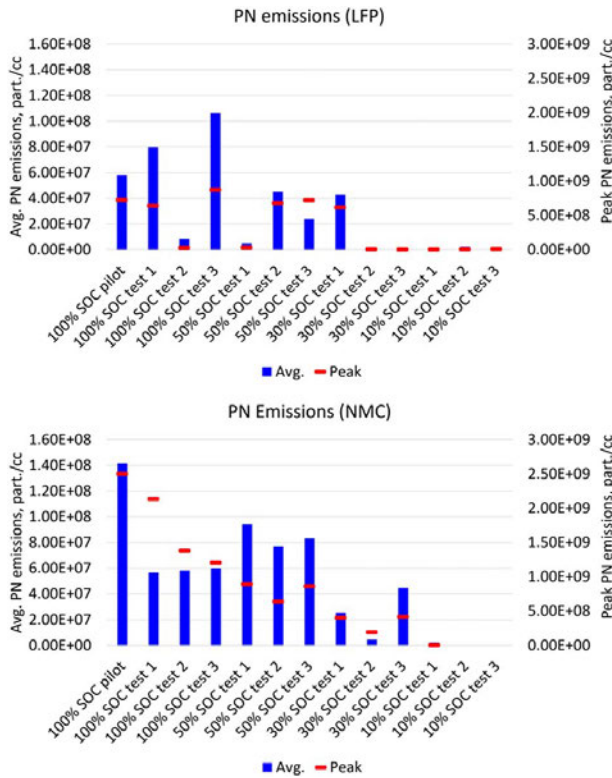


Fig. 9 Average and peak particle number emissions for all tests

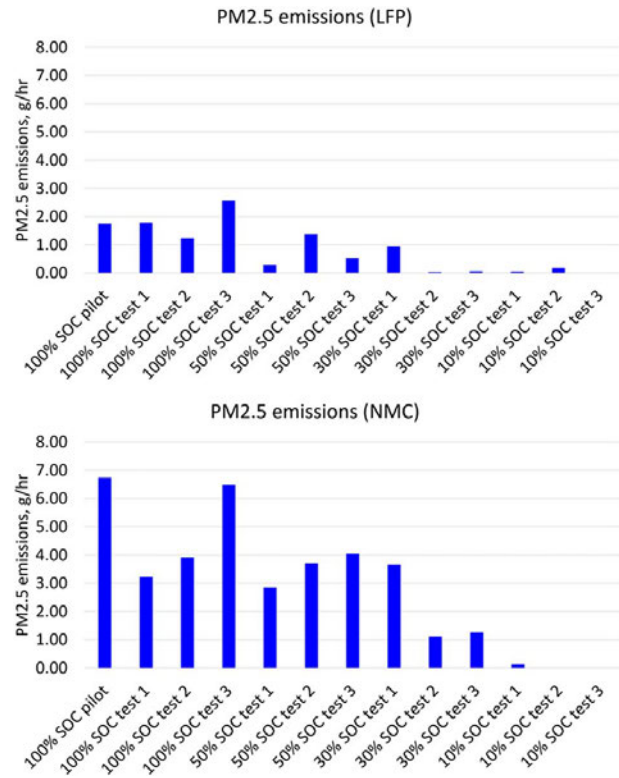


Fig. 10 PM2.5 emissions for all tests

Sierra BG-3 system. Sampling was started during the onset of thermal runaway and stopped either when the pressure drop across the filter reached 34 kPa (the upper threshold beyond which the sample filter will fail) or when PN concentration reported by the DMS500 reached ambient levels.

Figure 10 summarizes the PM2.5 emission rate for all tests conducted. PM2.5 emissions observed during NMC cell failures were higher than those observed with LFP tests. As observed with black carbon and PN emissions, LFP tests conducted at the same SOC exhibited variability in PM2.5 emissions. For both cell chemistries, 100% SOC resulted in higher PM2.5 emissions on an average basis, followed by 50%, 30%, and 10% SOC. This trend suggests that higher SOC results in a more dramatic failure event with more ejecta release, which has been observed by other researchers. Additional research needs to be conducted to determine the composition and morphology of PM2.5 collected on the filters, which is beyond the scope of this program.

3.5 Size Signatures of Particulate Emissions. Size distributions of thermal runaway emissions were characterized using the DMS500 Fast Particle Analyzer. Figure 11 provides size signatures for both cell chemistries at various SOC tested. Figure 12 summarizes the geometric mean number diameters of the size signatures for all tests conducted. One significant conclusion that can be drawn is the fact that the particulates emitted from cell failure were well within the human respirable size range. Further, high levels of emissions were also observed in the ultrafine particulate size range. These are defined as sub-100-nm particles and are known to be more toxic to human health as they tend to penetrate deeper into the human respiratory system and eventually diffuse into the bloodstream and reach various organs. In addition to the physical characteristics of these particulates, the chemical composition plays a critical role in terms of their impact on human health. For instance, cobalt is known to be poisonous, and its ingestion could lead to cardiomyopathy and a host of other health conditions that include nerve, vision, hearing, and thyroid problems [24]. Likewise, inhalation of nickel could lead to respiratory and neurological

illnesses, in addition to nickel being a known carcinogen [25]. Figure 12 suggests that higher test-to-test variability in GNMD was observed during LFP tests at a given SOC compared to NMC tests. Also, NMC tests resulted in particle size signatures

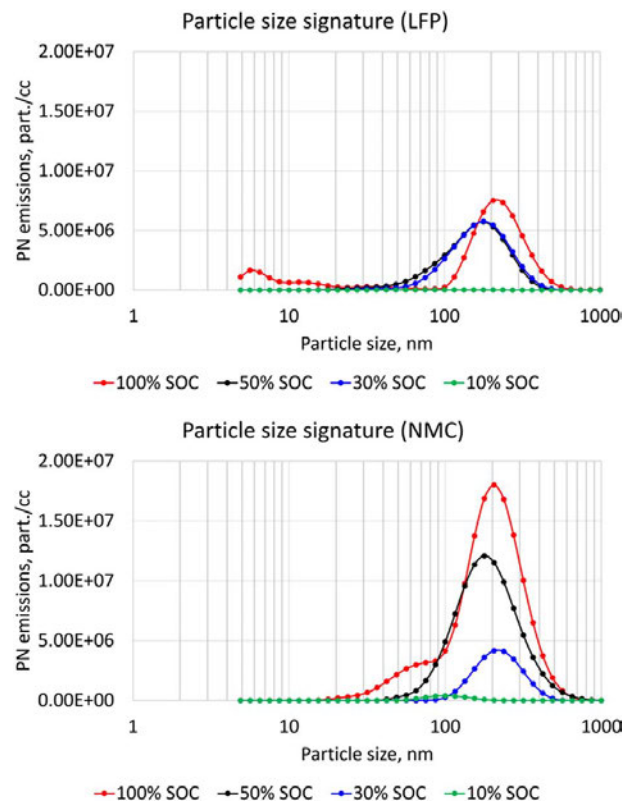


Fig. 11 Size signature of particulate emissions for all tests

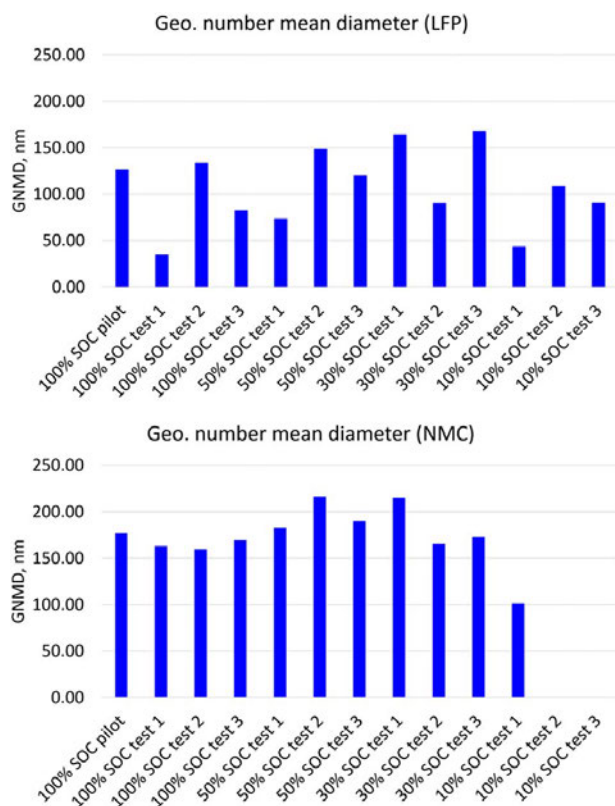


Fig. 12 Geometric number mean diameter analysis of size signatures for all tests

with larger size modes. Interestingly, for both cell chemistries, a decrease in SOC did not dramatically change the size signature, but instead, resulted in changes in peak levels.

4 Conclusions

The focus of this program was to conduct a detailed investigation of particulate emissions from Li-ion cell failure. Two commonly used Li-ion cell chemistries, LFP and NMC, were included as a part of this program. The LFP cells were prismatic metal can cells, while the NMCs were prismatic pouch cells. Cells were induced into thermal runaway using heating as the abuse method for all tests conducted. The test matrix was designed to gain knowledge of the impact of cell chemistry and SOC on thermal runaway particulate emissions and to understand the repeatability/variability associated with cell failure events. For each cell type, three repeats of overhear failure tests were conducted at 100%, 50%, 30%, and 10% SOC. All tests were conducted in a blast chamber equipped with an exhaust pipe from which particulate emissions were sampled from inception to completion of the thermal runaway process. Real-time black carbon emissions, PN, and size distribution were measured, along with PM_{2.5}, using state-of-the-art instrumentation. Results suggest the following:

- Thermal runaway of the cells tested as part of this program resulted in a significant release of particulate emissions. Peak PN concentrations of the order of $1 \times 10^{+09}$ were 6 orders of magnitude higher than typical ambient PN concentrations. Likewise, peak black carbon levels ($\sim 40 \text{ mg/m}^3$ observed during some tests) were also much higher than safe human exposure limits.
- Cell form factor, design, chemistry, composition of electrodes and electrolytes, and capacity could influence emission levels. At 100% SOC, NMC tests yielded average black carbon levels of $\sim 2.2 \text{ mg/m}^3$, PN levels of $\sim 6 \times 10^{+07}$ part/cc, and PM_{2.5}

emissions of $\sim 5.1 \text{ g/h}$. For tests conducted with LFP cells at 100% SOC, average black carbon levels ranging from 0.1 to 0.25 mg/m^3 , PN levels of $\sim 8 \times 10^{+07}$ part/cc, and PM_{2.5} emissions of $\sim 1.6 \text{ g/h}$ were observed.

- Results from three identical cell abuse tests at each SOC suggest that there can be significant variability in emissions. This observation is valid for both cell chemistries, with LFP cells exhibiting more variability than NMC cells.
- Among NMC tests, higher SOC tests resulted in higher peak PN concentrations. These concentrations were of the order of $1 \times 10^{+09}$ to $2 \times 10^{+09}$ part/cc. PM_{2.5} emissions also showed an increase with an increase in SOC. In the case of LFP tests, peak PN concentrations were reasonably similar for 100% and 50% tests (of the order of $7 \times 10^{+08}$ part/cc). However, average PN concentrations and PM_{2.5} decreased with a decrease in SOC.
- Size signatures of the thermal runaway process indicated emissions well within the respirable size range. Further, a large fraction of these particulates was in the ultrafine size range ($<100 \text{ nm}$), which is known to have a more adverse impact on human health.

Knowledge of particulate emissions from battery fires is essential to assess the impact such an event would have on personnel in the vicinity, first responders (and their protective gear), operators, and the environment. Such information is also critical to better understand failure modes such as high-voltage arcing and thermal runaway propagation and to develop mitigation strategies. This program serves to address a critical knowledge gap in understanding some aspects of particulate emissions from cell abuse at various states of charge for cells of two different cathode chemistries and capacities. Future research should examine other cell chemistries, formats, abuse mechanisms, and module and pack-level test articles. Further, additional details about particulate emissions, such as composition and morphology, should be examined.

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Funding Data

- UL Research Institutes.

Conflict of Interest

There are no conflicts of interest.

Data Availability Statement

The datasets generated and supporting the findings of this article are obtainable from the corresponding author upon reasonable request.

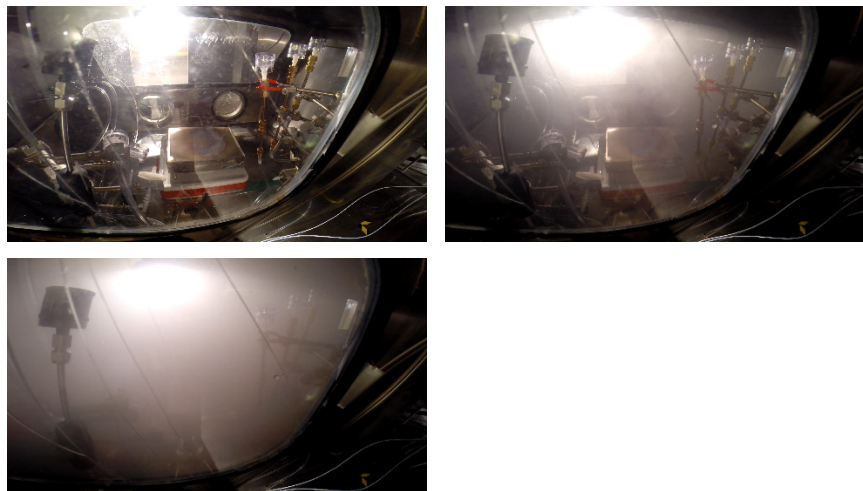
References

- [1] Jeevarajan, J. A., Joshi, T., Parhizi, M., Rauhala, T., and Juarez-Robles, D., 2022, "Battery Hazards for Large Energy Storage Systems," *ACS Energy Lett.*, **7**(8), pp. 2725–2733.
- [2] Velumani, D., and Bansal, A., 2022, "Thermal Behavior of Lithium and Sodium-Ion Batteries: A Review on Heat Generation, Battery Degradation, Thermal Runaway Perspective and Future Directions," *Energy Fuels*, **36**(23), pp. 14000–14029.

- [3] Essl, C., Golubkov, A. W., and Fuchs, A., 2020, "Comparing Different Thermal Runaway Triggers for Two Automotive Lithium-Ion Battery Cell Types," *J. Electrochem. Soc.*, **167**(13), p. 130542.
- [4] Wang, Q., Mao, B., Stolarov, S. I., and Sun, J., 2019, "A Review of Lithium Ion Battery Failure Mechanisms and Fire Prevention Strategies," *Prog. Energy Combust. Sci.*, **73**(Special issue Fire Safety), pp. 95–131.
- [5] Golubkov, A. W., Fuchs, D., Wagner, J., Wiltse, H., Stangl, C., Fauler, G., Voitic, G., Thaler, A., and Hacker, V., 2014, "Thermal-Runaway Experiments on Consumer Li-Ion Batteries With Metal-Oxide and Olivin-Type Cathodes," *RSC Adv.*, **4**(7), pp. 3633–3642.
- [6] Larsson, F., Andersson, P., Blomqvist, P., Lorén, A., and Mellander, B. E., 2014, "Characteristics of Lithium-Ion Batteries During Fire Tests," *J. Power Sources*, **271**, pp. 414–420.
- [7] Larsson, F., and Mellander, B. E., 2014, "Abuse by External Heating, Overcharge and Short Circuiting of Commercial Lithium-Ion Battery Cells," *J. Electrochem. Soc.*, **161**(10), pp. A1611–A1617.
- [8] Larsson, F., Andersson, P., and Mellander, B. E., 2016, "Lithium-Ion Battery Aspects on Fires in Electrified Vehicles on the Basis of Experimental Abuse Tests," *Batteries*, **2**(2), p. 9.
- [9] Nedjalkov, A., Meyer, J., Köhring, M., Doering, A., Angelmahr, M., Dahle, S., Sander, A., Fischer, A., and Schade, W., 2016, "Toxic Gas Emissions From Damaged Lithium Ion Batteries – Analysis and Safety Enhancement Solution," *Batteries*, **2**(1), p. 5.
- [10] Qiu, M., Liu, J., Cong, B., and Cui, Y., 2023, "Research Progress in Thermal Runaway Vent Gas Characteristics of Li-Ion Battery," *Batteries*, **9**(8), p. 411.
- [11] Sun, J., Li, J., Zhou, T., Yang, K., Wei, S., Tang, N., Dang, N., Li, H., Qiu, X., and Chen, L., 2016, "Toxicity, a Serious Concern of Thermal Runaway From Commercial Li-Ion Battery," *Nano Energy*, **27**, pp. 313–319.
- [12] Quant, M., Willstrand, O., Mallin, T., and Hynynen, J., 2023, "Ecotoxicity Evaluation of Fire-Extinguishing Water From Large-Scale Battery and Battery Electric Vehicle Fire Tests," *Environ. Sci. Technol.*, **57**(12), pp. 4821–4830.
- [13] Wang, H., Wang, Q., Jin, C., Xu, C., Zhao, Y., Li, Y., Zhong, C., and Feng, X., 2023, "Detailed Characterization of Particle Emissions Due to Thermal Failure of Batteries With Different Cathodes," *J. Hazard. Mater.*, **458**, p. 131646.
- [14] Zhang, L., Yang, S., Liu, L., and Zhao, P., 2022, "Cell-to-Cell Variability in Li-Ion Battery Thermal Runaway: Experimental Testing, Statistical Analysis, and Kinetic Modeling," *J. Energy Storage*, **56**, p. 106024.
- [15] Walker, W. Q., Darst, J. J., Finegan, D. P., Bayles, G. A., Johnson, K. L., Darcy, E. C., and Rickman, S. L., 2019, "Decoupling of Heat Generated From Ejected and Non-Ejected Contents of 18650-Format Lithium-Ion Cells Using Statistical Methods," *J. Power Sources*, **415**, pp. 207–218.
- [16] Zhang, Y., Wang, H., Li, W., and Li, C., 2019, "Quantitative Identification of Emissions From Abused Prismatic Ni-Rich Lithium-Ion Batteries," *ETransportation*, **2**, p. 100031.
- [17] Chen, S., Wang, Z., and Yan, W., 2020, "Identification and Characteristic Analysis of Powder Ejected From a Lithium Ion Battery During Thermal Runaway at Elevated Temperatures," *J. Hazard. Mater.*, **400**, p. 123169.
- [18] Barone, T. L., Dubaniewicz, T. H., Friend, S. A., Zlochower, I. A., Bugarski, A. D., and Rayyan, N. S., 2021, "Lithium-Ion Battery Explosion Aerosols: Morphology and Elemental Composition," *Aerosol Sci. Technol.*, **55**(10), pp. 1183–1201.
- [19] Yang, Y., Fang, D., Maleki, A., Kohzadi, S., Liu, Y., Chen, Y., Liu, R., Gao, G., and Zhi, J., 2021, "Characterization of Thermal-Runaway Particles From Lithium Nickel Manganese Cobalt Oxide Batteries and Their Biototoxicity Analysis," *ACS Appl. Energy Mater.*, **4**(10), pp. 10713–10720.
- [20] Premnath, V., Wang, Y., Wright, N., Khalek, I., and Uribe, S., 2022, "Detailed Characterization of Particle Emissions From Battery Fires," *Aerosol Sci. Technol.*, **56**(4), pp. 337–354.
- [21] Reavell, K., Hands, T., and Collings, N., 2002, "A Fast Response Particulate Spectrometer for Combustion Aerosols," *SAE Trans.*, pp. 1338–1344.
- [22] Schindler, W., Haisch, C., Beck, H. A., Niessner, R., Jacob, E., and Rothe, D., 2004, "A Photoacoustic Sensor System for Time Resolved Quantification of Diesel Soot Emissions," *SAE Trans.*, pp. 483–490.
- [23] Fanick, E. R., and Premnath, V., 2020, "Comparison of Partial and Total Dilution Systems for the Measurement of Polycyclic Aromatic Hydrocarbons and Hydrocarbon Speciation in Diesel Exhaust," SAE Technical Paper 2020-01-2190, p. 8.
- [24] Wahlqvist, F., Bryngelsson, I. L., Westberg, H., Vihlborg, P., and Andersson, L., 2020, "Dermal and Inhalable Cobalt Exposure – Uptake of Cobalt for Workers at Swedish Hard Metal Plants," *PLoS One*, **15**(8), p. e0237100.
- [25] Genchi, G., Carocci, A., Lauria, G., Sinicropi, M. S., and Catalano, A., 2020, "Nickel: Human Health and Environmental Toxicology," *Int. J. Environ. Res. Public Health*, **17**(3), p. 679.

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Vented Gases and Aerosol of Automotive Li-ion LFP and NMC Batteries in Humidified Nitrogen under Thermal Load



Titel	Undersökning av ventilerade gaser och aerosoler från litiumjonbatterier av typen LFP och NMC i kvävgasatmosfär
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Cover photo: Gas evolution after thermal runaway and rupture of a Li-ion pouch cell battery

David Sturk, Autoliv Development AB, Sweden

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Sammanfattning

Detta är en studie av ventilerade gaser från litium-jonbattericeller vid termisk rusning i inert atmosfär. Batterier av påscelltyp för bilindustri med litium-järn-fosfat (LFP) och litium-nickel-mangandioxid-kobolt (NMC) katodkemi exponerades för värme i fuktat (motsvarande typisk utomhusmiljö) kvävegasflöde tills dess att cellen sprack p.g.a. invärtes gstryck. Med gaskromatografi-masspektrometri och Fouriertransform infraröd spektroskopi gjordes en skanning av kända och okända substanser. Metalliska grundelement i den partikulära aerosolen analyserades med induktivt kopplad plasma masspektrometri.

Bland de detekterade ämnena finns flera toxiska ämnen, till vår kännedom ej tidigare rapporterade vid ventilerings efter termisk rusning. Många av ämnena är irriterande för huden och andningsvägarna. Huruvida de ventilerade gaserna når en exponeringsnivå som kan vara farlig är situationsbunden och slutsatser kring detta kan inte dras i denna rapport. Därför är det viktigt med fullskaliga försök för att utreda exponeringsnivåer för insatspersonal såväl för passagerare i händelse av termisk rusning till följd av exempelvis en krock.

Tre av de detekterade ämnena, etyl etylfosfonofluoridat, etyl metylfosfonofluoridat, och metyl metylfosfonofluoridat har strukturella likheter med kemiska stridsmedel i gruppen organiska fosforföreningar. Uppskattningar av dessa ämnens kvantiteter och eventuella toxicitet vid ett verklighetstroget scenario av en trafikskadehändelse kräver en ny studie med ett större provsystem och mängd provmaterial. Därutöver cellstudier för att studera deras eventuella inhiberingsförmåga av enzymet acetylkolinesteras.

Denna rapport innehåller en förteckning över identifierade kemiska ämnen vid ventilerings tillsammans med en toxikologisk verkansbedömning i de fall sådana data finns tillgängliga. Den experimentella uppställningen är beskriven för att på ett säkert sätt kunna utföra dessa studier.

Nyckelord: E-fordon, Litium-jonbatteri, ventilation, gaser, termisk rusning, säkerhet, irriterande ämnen, toxikologisk riskbedömning, LFP, NMC, aerosol

Summary

Chemical compounds were screened in a study on gases ventilated from Li-ion battery cells during thermal runaway in an inert environment. Automotive Li-ion pouch cells of lithium iron phosphate (LFP) and lithium nickel manganese cobalt oxide (NMC) cathode chemistry were exposed to heat-up in humidified nitrogen atmosphere until rupture caused by internally build up pressure. By means of analytic instruments such as Gas Chromatography-Mass Spectrometry and Fourier Transform Infrared Spectroscopy a screening of known and unknown constituent gaseous species was performed. Elemental metal contents in the aerosol phase were analyzed with Inductively Coupled Plasma Mass Spectrometry.

Among the detected compounds there are several, to our knowledge, of potential toxic concern previously not reported to ventilate during thermal runaway of Li-ion batteries. Many of the compounds are irritating to the skin and the respiratory system. Whether the exposure of vented gases reach a dangerous dose in the case of real world accident involving Li-ion batteries cannot be concluded from this study. To assess first responders' occupational exposure risk, as well as victim exposure in e.g. a car accident, a full scale experiment is needed including propagation of thermal runaway in a stack of battery cells.

Three of the detected compounds; ethyl ethylphosphonofluoridate, ethyl methylphosphonofluoridate, and methyl methylphosphonofluoridate are very similar to some organophosphorous chemical warfare agents. However, in order to perform a qualitative estimation of their quantities and possible toxicities in relation to a realistic scenario of a traffic accident, additional studies need to be performed. These include studies on a larger test system and quantity of tested material as well as *in vitro* studies to evaluate if they are potential enzyme inhibitors of acetylcholine esterase. This report includes description of the full experimental setup to safely perform such studies. A list of identified compounds together with toxicological assessment is given.

Keywords: e-vehicle, Li-ion battery, ventilation, gases, thermal runaway, safety, irritating compounds, toxicological assessment, LFP, NMC, aerosol

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1 Introduction

In the event of an E-vehicle crash, quick and competent rescue of the vehicle occupants requires new knowledge and up-dated training to minimize the time to final treatment for those injured. This was the conclusion drawn by the Swedish Civil Contingencies Agency (MSB) in 2011 after analyzing the results of a survey that they had sent out to Swedish first responders. This led to the initiation of a Triple Helix research project, E-Vehicle Safe Rescue (2012-2014), where abusive testing on e-vehicle batteries (cells/modules/packs) were conducted by Swedish partners from the industry, research institutes and academics and the results were compiled by MSB into a free web-based educational material for first responders [Sturk *et al.* 2013]. Amongst mechanical, electrical and thermal abuse tests, bonfire tests on lithium iron phosphate (LFP) and nickel manganese cobalt (NMC) Li-ion battery cells were conducted, analyzed and published [Sturk *et al.* 2015].

In January 2015, MSB identified that a remaining task to be investigated was the composition of gases that can be vented without fire from a critically damaged automotive battery such as the LFP and NMC Li-ion batteries. Those two types of Li-ion batteries were chosen as they are the two most common ones found in commercialized automotive HV batteries. The present study addresses this concern and during autumn of 2015 the Swedish Defense Research Agency (FOI) performed a series of experiments based on battery material provided from E-Vehicle Safe Rescue and Autoliv Development AB.

The results of this investigation will provide MSB with further material to be added to the existing education for first responder rescue routines at the event of a traffic incident involving E-Vehicles.

1.1 Commission

According to quote “FOI-2015-1534 / MSB-2015-09-07”, the team of researchers at FOI were to investigate the ventilated gases from Li-ion pouch cells of the mentioned chemistries by means of Gas Chromatography-Mass Spectrometry (GC-MS) and Fourier Transform Infrared Spectroscopy (FTIR) in order to screen for known and unknown gaseous species. Examples of known gases or vapors that can be generated during critical thermal failure of Li-ion cells are carbon dioxide, carbon monoxide (CO), hydrogen fluoride (HF), fluorocarbons, hydrogen, and vaporized or aerosolized electrolyte alkyl carbonates. [Huang *et al.* 2008, Kawamura *et al.* 2006, Roth, E. P. 2008, Wang *et al.* 2006, Wilken *et al.* 2012, Wilken *et al.* 2013, Yang *et al.* 2006a, Yang *et al.* 2006b, Hammami *et al.* 2003].

The produced scientific report should contain also assessment of which of the identified gases that may result in an intoxication of individuals in the event of a sufficient exposure. The results fulfill requirements set out by MSB.

2 Experimental settings

2.1 Facilities

A 65 m³ room with stainless steel walls and with air supply from the outside after temperature adjustment and filtered through a particle filter ensured researchers safety. Exhaust air from the steel room passed through aerosol filters and carbon filters before it was released to the surroundings. Risk assessment prior the experiments demonstrated that the route for exposure was via inhalation and via the skin (particularly HF). To reduce the exposure of toxic compounds a protective mask, protective suit, gloves and boots were worn during critical parts the experiments.

In this room, a glove box was used as test chamber for the thermal challenge of the lithium ion batteries. Inside the glove box a small stainless steel box (referred to as “battery box”) was used as a compartment for the lithium ion pouch cell, see figure 1. This glove box of 0.38 m³ served as a conditioning volume for ventilated gases/aerosol. An exit tube was attached between the glove box and the fan system in order to ventilate the gases and particles after degassing and collection of gases and aerosols.



Figure 1. Battery box with two connections, one for nitrogen gas inlet and one for temperature probe, dimension 204 mm x 264 mm x 64 mm.

Thermal load was applied by an electric resistive heater (simple stove top) until the pouch cell erupted and released hot gases and aerosol. To prevent the emitted gas to take fire experiments were performed in nitrogen atmosphere. It was assumed that when CO₂ decreased to a certain level so had O₂ and the threshold of 15% O₂ to prevent open flame could with excess be fulfilled [Fredholm and Saarkoppel 2012]. Thus O₂ levels were followed indirectly by studying CO₂ level evolution on GASMET after the glove box was sealed and evacuated by inserting N₂.

Dry nitrogen was moistened with water (MilliQ) to 63% RH (20 g H₂O /h at 30 l/min N₂ which corresponds to 11 g water/m³ at 20°C) to simulate typical outdoor climate. Nitrogen humidification was done with a Controlled Evaporator Mixer, type W-300 and a mini CORI-FLOW digital Mass Flow meter (Bronkhorst High-Tech B.V., NL), figure 2. The nitrogen flow to the Mixer was set with a mass flow controller (0-100 L/min). The humidified nitrogen flow was connected to the battery box. Temperatures inside the battery box and the glove box were measured with thermocouples connected to two handheld TESTO 435-4 instruments (Testo inc. Germany) connected to laptops for data logging. To monitor the battery box inside the glove box during the experiments, one digital video action camera (GoPro 3) was used, with Wi-Fi the recording pictures could be seen on the screen of a connected cellular phone. See the glove box with sampling equipment in figures 3 and 4.

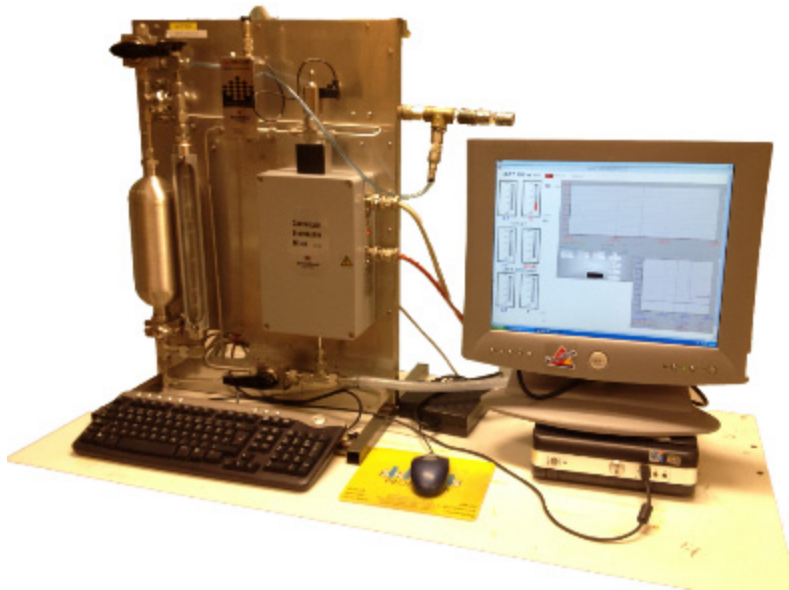


Figure 2. A mass flow controller and a humidifier were used to supply the battery box with humidified nitrogen.



Figure 3. The glove box of 0.38 m³ used to condition vented gases and aerosols.



Figure 4. Inside the glove box with all sampling equipment setup and the battery box on the heater ready to receive thermal load until ventilation.

2.2 Sampling methods and analysis protocol

2.2.1 Filter and adsorbents

Air sampling with Tenax tube (PerkinElmer, USA), Carbosieve tube (PerkinElmer, USA) and 25 mm GF filter (PALL, USA), connected to a sampling train, was done with PocketPump's (SKC inc., USA) at a flowrate of 100 mL/min. The GF filter was used to protect the gas sampling tubes of Carbosieve and Tenax. The sampled air passed the Tenax tube first, removing semi volatile compounds, volatile compounds passed the Tenax tube and were collected on the Carbosieve tube. Particulate aerosols were collected on 37 mm PVC filter (PALL, USA) and a set of 47 mm GF filter and an Anasorb 747 adsorbent tube. The sampling rate for the PVC filter was 5 L/min (Leland Legacy pump, SKC inc, USA) and 2 L/min for the 47 mm GF filter with Anasorb 747 tube (AirCheck2000, SKC inc.).

2.2.2 GC-MS

The adsorbent tubes (Tenax and Carbosieve) were analysed by thermal desorption gas chromatography (TD-GCMS). The instrumentation used were TD: TurboMatrix 350 ATD, Perkin Elmer; GC: Agilent 7890A Gas Chromatograph; MS: Agilent 5975C MSD. Adsorbent tubes were desorbed at 250°C during 3 min using a split ratio of 1:10 (inlet split 200 mL/min, tube desorb 20 mL/min). The cold trap used to refocus the analytes was set to 2°C for Tenax adsorbent tubes using a Thermal Desorber Trap, Packed with Tenax (M041-3535, Perkin Elmer, CT, USA). The analytes from the Carbosieve adsorbent tubes were refocused onto an Air Monitoring Trap (M041-3628, Perkin Elmer, CT, USA) kept at a temperature of -30°C. After the initial desorption stage the trap was rapidly heated to 250°C to introduce the analytes to the GC after a second split of 1:10. The temperature of the valve and transfer line were 180°C and 200°C respectively. Chromatographic separation was performed using DB-5 MS analytical column (Agilent J&W scientific, USA) with a length of 30 m x 0.25 mm and a film thickness of 0.25 µm. The GC profile was as follows: initial temperature was kept at 40°C for 1 min, then increased to 100°C at a rate of 5°C/min and then ramped at 15°C/min until 300°C. The mass spectrometer was operated in full scan mode (mass 29-400) to allow identification. Samples were quantified against calibrations generated using toluene in methanol spiked onto an adsorption tube. The raw data files were evaluated using the software AMDIS.

2.3 Metal analysis

The aerosol sample collected on PVC-filter were dissolved using a microwave oven and HNO₃ / HF in a Teflon vessels. The metal analyses (elements) were performed with ICP-SFMS according to EN 17294 to 1.2 (modified) and EPA Method 200.8 (modified) by ALS Scandinavia, Sweden.

2.3.1 FT-IR

During the experiments gas measurements were done with two FTIR (Fourier transform infrared spectroscopy) instruments, a GASMET model DX-4030 instrument (Gasmeter Technologies OY, Finland) and a Bruker Alpha instrument with a 5 cm gas cell (Bruker Corporation, Germany). Of particular interest are volatile gases that do not adsorb onto the Carbosieve tubes; HF and CO. HF has a rotational-vibrational structure in the region 3600 to 4200 cm⁻¹. The P-branch is interfered by overlapping water signal and only the R-branch (> 4000 cm⁻¹) can be used. Due to the longer path length GASMET analyzer was found best for low ppm levels. HF reference spectra and a method based on the only the R-branch was used

to study HF. CO has absorptions at 2114 cm^{-1} and 2180 cm^{-1} (NIST) and is predesigned gas in the library of GASMET.

Bruker Alpha was used together with a 5 cm optical path length gas cuvette. 16 scans with double-sided forward-backward collection in repeated mode measurements. Background was taken as the heat was started. The spectral window was extended to 5000 cm^{-1} to accommodate possible HF absorptions.

2.4 Test procedure

The empty battery box was conditioned at over $250\text{ }^{\circ}\text{C}$ to degas possible interfering substances. Background measurements were done with an empty battery box in a similar fashion as sharp measurements. Totally four experiments were done with two types of Li-ion batteries, two with single LFP pouch cells and two with single NMC pouch cells; referred to as LFP1 (3.2 V), LFP2 (3.26 V), NMC1 (3.84 V), and NMC2 (3.84 V).

The battery was placed inside the battery box. Amounts of degassed substance was determined from battery weights before and after the experiment. To prevent the plastic casing of the battery pouch cell to glue onto the surface of the battery box during the thermal release an alumina foil was placed between the battery and the box. A perforated metal U-profile clamped between the battery and the ceiling of the battery box prevented the battery to swell in the middle and ensured optimal thermal connection.

The battery box was closed and placed on the electric heater, and nitrogen supply and the temperature probe was connected. Air samplers were placed inside the glove box, tubings to the FTIR instruments were connected and a temperature probe was connected to the glove box. The door to the glove box was closed and the humidified nitrogen flushed the glove box until the carbon dioxide concentration inside the glove box was reduced to a low value ($< 1/3$ of initial concentration).

The battery box was heated until the release of aerosol to the glove box. At that sign the power to the heater was shut down. Sampling started immediately and proceeded 10 min. On the first run (LFP1), it was discovered that gas concentrations inside the glove box were too high for GASMET as its spectra were saturated. The sampled gas to the instrument was then diluted 100 times with a Dekati double diluter (Dekati Ltd, Finland).

Following the 10 min, the humidification of nitrogen ended and the nitrogen flow increased to 50 L/min to evacuate the glove box to minimize the risk of open flame as the door is opened and volatile electrolyte gases is exposed to oxygen in surrounding air.

After about another 10 min the glove box was visually clear and filter and adsorbents were collected and sealed.

Between the experiments the battery box and glove box was cleaned with moistened paper towels.

3 Results

3.1 Thermal runaway

The pouch cells ruptured on the long sides as opposed to the weakest point which was anticipated to be around the tabs, figure 5. For the gas evolution this was not of any concern.

The temperature evolution inside the battery box is shown in figure 6. Heating of the battery box also created a small temperature rise inside the glove box, figure 7.

The humidity inside the battery box and inside the glove box were not recorded by a humidity sensor but can be estimated from the absolute humidity of $11 \text{ g H}_2\text{O/m}^3$ and the recorded temperatures in the battery box. At the maximum temperatures recorded during thermal runaway this corresponds to $< 1\%$ RH in all four experiments. In the glove box on the other hand humidities were between 7.9% RH at 46°C and 11% RH at 39°C .



Figure 5. Ruptured pouch cell of LFP type. NMC type is similar.

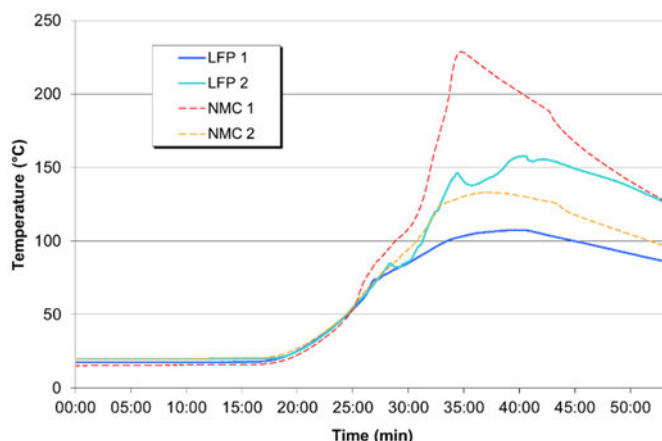


Figure 6. Temperatures inside the battery box. We emphasize that temperature evolution inside the battery box may well depend on how the pouch opened relative the position of temperature measurement head and when the heater is powered off.

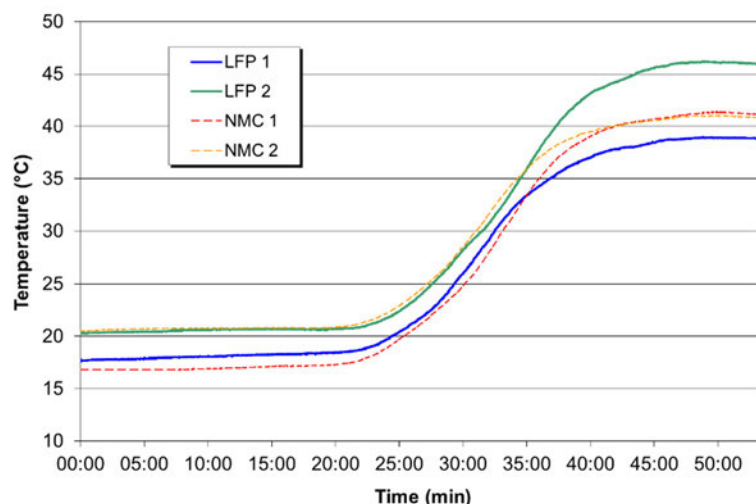


Figure 7. Temperatures inside the glove box.

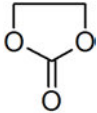
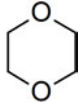
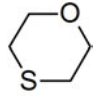
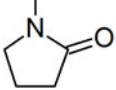
3.2 Chemical analysis

Substances identified from adsorbent tubes by means of ATD-GCMS are summarized in table 1 and 2 for LFP and NMC batteries respectively. Compounds were identified using the software AMDIS (Automated Mass spectral Deconvolution and Identification System). The selection criteria for a positive identification was a match factor above 90.

The most abundant compounds emitted from the batteries were carbonates. Diethyl carbonate (DEC) and ethylmethyl carbonate (EMC) found in the LFP battery and dimethyl carbonate (DMC) and EMC in the NMC battery. In addition, both battery types contained large amounts of 1,3-dioxolan-2-one.

Similar compounds are detected for the different battery types. However, in the NMC batteries phosphonofluoridates were detected which were not found in the LFP batteries. The volatile compounds previously reported with FT-IR such as phosphorylfluoride, pentafluorophosphate and hydrogenfluoride were not detected in our experiment. Notably is that a sweet smell occurred after ventilation, even though high safety precautions were taken in the setup to avoid exposure.

Table 1. Compounds emitted from LFP battery.

Chemical compound	Structure	CAS#	Concentration (µg/m ³)
1,3-dioxolan-2-one		96-49-1	300
1,4-dioxane		123-91-1	0.2
1,4-oxathiane		15980-15-1	10
1-methylpyrrolidin-2-one		872-50-4	5

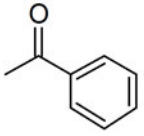
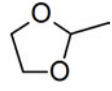
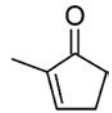
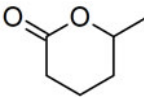
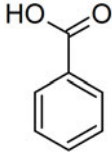
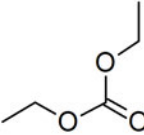
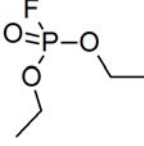
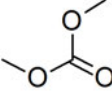
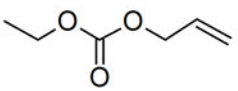
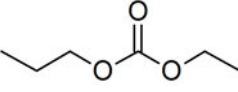
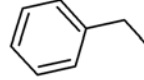
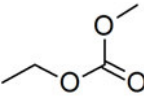
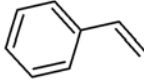
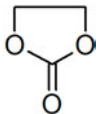
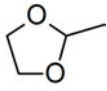
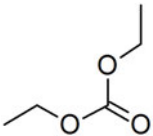
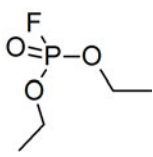
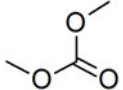
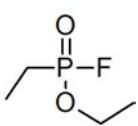
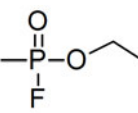
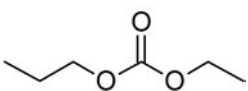
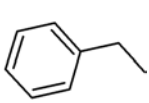
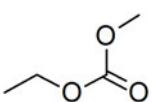
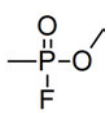
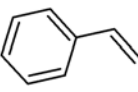
Chemical compound	Structure	CAS#	Concentration ($\mu\text{g}/\text{m}^3$)
1-phenylethanone		98-86-2	1
2-methyl-1,3-dioxolane		497-26-7	2
2-methylcyclopent-2-en-1-one		1120-73-6	0.2
6-methyloxan-2-one		823-22-3	0.1
Benzoic acid		65-85-0	0.1
Diethyl carbonate (DEC)		105-58-8	1000
Diethyl phosphorofluoridate		358-74-7	0.5
Dimethyl carbonate (DMC)		616-38-6	1
Ethyl prop-2-enyl carbonate		1469-70-1	0.1
Ethyl propyl carbonate		35363-40-7	0.1
Ethylbenzene		100-41-4	2
Ethylmethyl carbonate (EMC)		623-53-0	1000
Styrene		100-42-5	5

Table 2. Compounds emitted from NMC battery.

Chemical compound	Structure	CAS#	Concentration ($\mu\text{g}/\text{m}^3$)
1,3-dioxolan-2-one		96-49-1	100
1,4-oxathiane		15980-15-1	2
2-methyl-1,3-dioxolane		497-26-7	10
Diethyl carbonate (DEC)		105-58-8	30
Diethyl phosphorofluoridate		358-74-7	10
Dimethyl carbonate (DMC)		616-38-6	300
Ethyl ethylphosphonofluoridate		650-20-4	0.2
Ethyl methylphosphonofluoridate		673-97-2	0.5
Ethyl propyl carbonate		35363-40-7	0.01
Ethylbenzene		100-41-4	0.1
Ethylmethyl carbonate (EMC)		623-53-0	1000
Methyl methylphosphonofluoridate		353-88-8	0.02
Styrene		100-42-5	5

3.3 Metal analysis and Total Suspended Particles (TSP)

In order to investigate the presence of metal compounds emitted from the batteries, ICP-MS analysis was performed on two filters (one replicate) for each trial. Although filters clogged after a few minutes of sampling because of the dense aerosol, concentrations could still be retrieved because pumps recorded sampled volume. Of special interest were lithium and cathode elements such as manganese, cobalt and nickel. The variation between sample and replicate is large and for many elements one of the samples were below the detection limit of the method. As the primary interest is to indicate any presence of metal element, the highest levels for each battery type is reported in table 4. Also blank levels (empty glove box) show that the experiment setup release metals, note that blank levels are subtracted in table 4. Many of the selected elements are present in the aerosol. To our knowledge this is the first report on metal contents of vented aerosols from Li-ion batteries.

LFP

Interestingly lithium was detected in the LFP batteries but not in the NMC batteries. Trial LFP2 had the highest lithium levels of 46 $\mu\text{g}/\text{m}^3$. Nickel, cobalt, and manganese were found in both battery types at lower levels (see also toxicological assessment below). Note that LFP batteries were prior to NMC to avoid contamination of the setup.

NMC

High concentrations of TSP are found in all trials and NMC types seems to release more particles than LFP. Averages of the two sample heads in each trial are reported in table 3. The between trial variation is large, from 1.4 g/m^3 for LFP1 to 7 g/m^3 for NMC1. The amount released range from 30 to 49 g and follow TSP values with $\text{NMC} > \text{LFP}$.

The intact cells are of similar weights within the same type of cell. NMC batteries are heavier than LFP (386 vs. 236 g).

Table 3. Elements detected by ICP-MS, background from glove box subtracted. TSP values are averages of the two sample heads in each trial. Released amounts are calculated from weights before and after thermal runaway.

Element (down)	Trial (right)	LFP1	LFP2	NMC1	NMC2
TSP*	g/m^3	1.4	5.3	7.0	5.3
Released amount	g	30	43	49	46
Co	$\mu\text{g}/\text{m}^3$	0.036	0.11	0.39	0.016
Li	$\mu\text{g}/\text{m}^3$	2.2	46	n.d.	n.d.
Mn	$\mu\text{g}/\text{m}^3$	1.2	1.5	2.8	1.1
Ni	$\mu\text{g}/\text{m}^3$	0.24	1.2	1.1	1.8
Pb	$\mu\text{g}/\text{m}^3$	1.5	0.21	0.09	0.31
Sn	$\mu\text{g}/\text{m}^3$	6.8	1.8	0.37	2.0

*TSP, Total Suspended Particles
n.d. not detected

3.4 FT-IR

Spectra contain in general many absorptions in the fingerprint region, figure 8, and in the high-wavenumber region, figure 9. Many of these are absorptions likely from electrolyte vapors, see ref [Wilken *et al.* 2012]. Of special interest are gases that are not readily collected onto the gas adsorbents. Previously known compounds are POF_3 and PF_5 [Yang *et al.* 2006b]

which are intermediates in HF evolution. However, none of these can readily be assigned from IR spectra. Particularly a peak at 989 cm^{-1} has previously been assigned to the asymmetric stretch of P-F of POF_3 [Yang *et.al.* 2006b] but since spectra lack P=O stretch at 1415 cm^{-1} as well as the symmetric P-F stretch mode 873 cm^{-1} we do not infer any POF_3 from IR spectra. Similar observations are made for PF_5 . Absorptions in the IR spectra in the region $940\text{--}1040\text{ cm}^{-1}$ are thus likely to originate from other fluorophosphorous compounds. It is known that both POF_3 and PF_5 have their major evolution between 170°C and 200°C [Wilken *et al.* 2012] which is over the measured temperature in the battery box for three of four trials. Whether this is the cause of their absence is beyond the scope of this study.

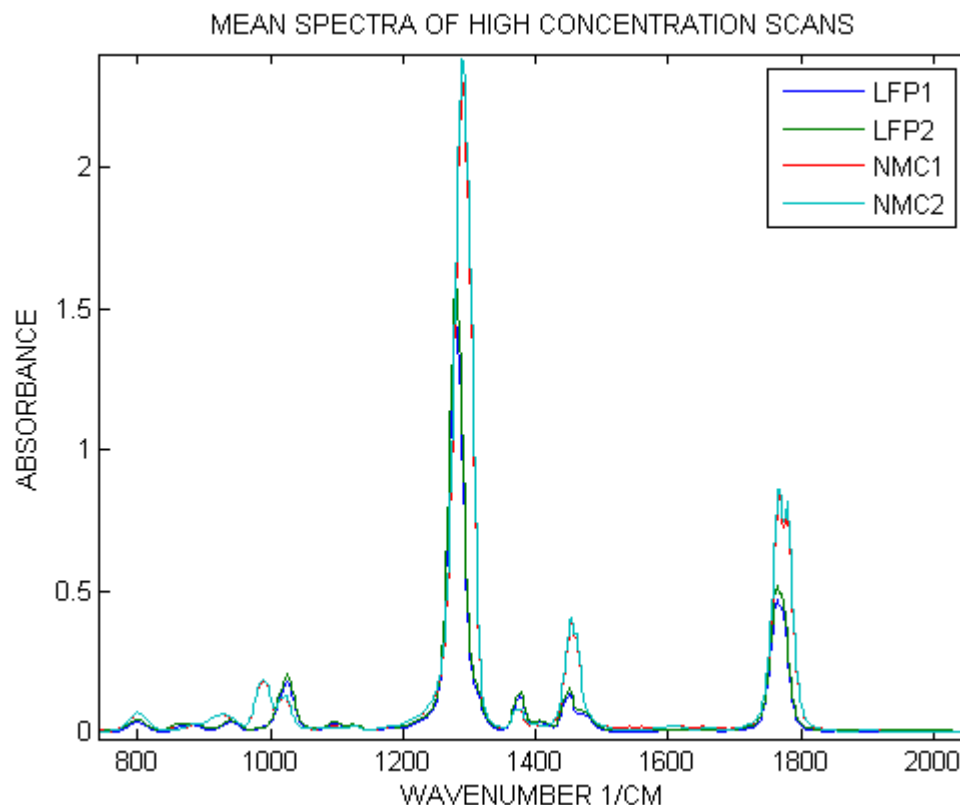


Figure 8. FT-IR spectra of the fingerprint region.

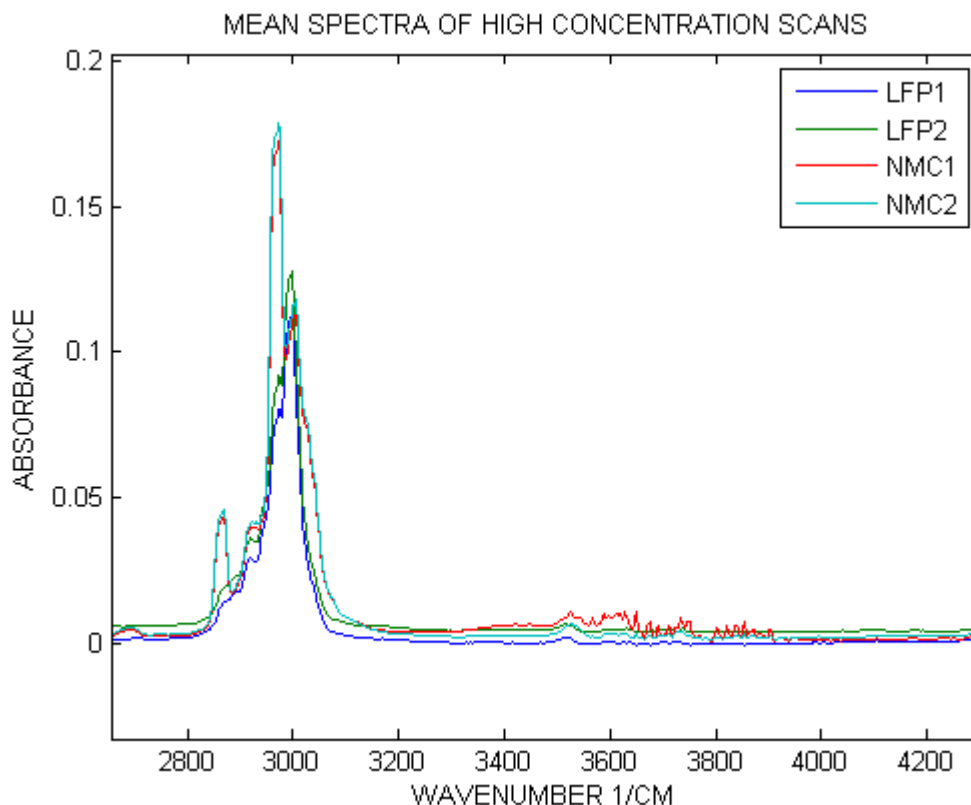


Figure 9. FT-IR spectra of the high-wavenumber region.

Only for trial LFP1, positive HF signal could be inferred from GASMET analyzer. This is because this trial was run without sample dilution and hence chances are greatest to observe HF. Qualitative agreement is found, however, there is an interfering signal that makes quantitative assignment ambiguous. With the current method levels are indicated at ~50 ppm at maximum for LFP1. Other trials (LFP2, NMC1, and NMC2) the sample gas were diluted with air 1:100 and no HF could be detected. Dräger tubes were co-deployed (without dilution) and repeated use show that HF levels varied rather quickly over the battery ventilation, with highest levels measured in the early stage of ventilation. Possibly because of the time-point of sampling, the highest levels of HF was found for NMC cells and numbered >200 ppm. CO is analyzed at 2114 and 2180 cm^{-1} [<http://webbook.nist.gov/>]. There is no evidence found of CO in FT-IR spectra. Due to budget/time limitations no effort has been done for a full spectral band assignment, but table 4 of found absorptions is included for future reference.

Table 4. Infra-red absorption bands of vented gases for both LFP and NMC battery types. Strength is subjective.

Wavenumber cm ⁻¹	Type (B=both)	Strength
799	B	w
928	NMC	vw
942	LFP	vw
989	NMC	m
1024-1028	B	m
1097	B	vw
1126	B	vw
1280	LFP	vs
1289	NMC	vs
1371	B	w
1379	B	w
1444	LFP	w
1452	LFP	w
1454	NMC	m
1463	NMC	m
1764	LFP	s
1769	NMC	s
1779	NMC	s
1773	LFP	s
2109	NMC	vw
2211	NMC	vw
2342	B	m
2362	B	m
2866	NMC	w
2968	NMC	m
2972	LFP	w
2996	LFP	m
3005	NMC	w
3515	LFP	m
3523	NMC	m
3600-4000	B	m
4451	LFP	w
4453	NMC	w

vw – very weak

w – weak

m – medium

s – strong

vs – very strong

3.5 Toxicological assessment

The overall goal is to evaluate health risks during rescue in case of thermal runaway of automotive Li-ion batteries. The present experiments have detected a complex mixture of organic chemicals and metals emitted from two battery chemistries of automotive Li-ion pouches. The following summary gives a first indication of potential hazards related to single exposures via air to these chemicals but cannot fully take into account the possible consequences of mixed exposures. Most gaseous compounds are detected in low concentrations during heating of one battery pouch. However, the values for total suspended particles are well above limit level value for total dust (5 mg/m^3). To evaluate the relation between concentrations in air and risk for toxic effects during rescue, a full scale experiment is necessary. For several of the detected substances there were no or very little toxicological data available in the databases used in this overview.

In brief, several of the chemicals are mildly to severely irritating to the eyes, respiratory tract and to the skin. For instance, both ethylbenzene and styrene are harmful to the airways. Hydrogen fluoride is corrosive and may cause serious systemic intoxication after inhalation or skin exposure. Also lithium induces corrosive effects to both the skin and respiratory system.

Lithium levels are above the ceiling limit value in this experimental setup for one trial (LFP2). Other metal elements analyzed are far below the level limit value. It should however be noted that the current metal analysis does not include measurements of their physical and chemical properties which could highly affect the potential for toxic effects. Manganese, for instance, is considered more neurotoxic when in an ionic state as compared to large particles.

Several chemicals can be taken up by the skin, including 1,4-dioxane, ethylbenzene, styrene, 1-methylpyrrolidin-2-one and cobalt. Of these, two of the chemicals, 1,4-dioxane and cobalt, are classified by the Swedish Work Environment Authority as carcinogens.

Three of the chemicals are phosphonofluoridates or phosphorofluoridates. Their structural similarity to some organophosphorus compounds (e.g. nerve agent sarin) indicate that they potentially could be highly toxic. The first step to evaluate if these chemicals may cause toxicity would be to analyze their potential to inhibit the enzyme acetylcholinesterase. If these chemicals can inhibit the enzyme to a relevant degree this should be followed by an *in vitro* screening of medical countermeasures.

Table 5. Toxicological assessment

Chemical compound	CAS no. ¹	TOXNET ² Skin eyes, respiratory tract	Hygienic limit values ³	Labelling of chemicals according to ECHA ⁴ : Hazard codes for chem-phys & human health
1,3-dioxolan-2-one	96-49-1	Irritating to eyes, skin and airways.	---	NA
1,4-dioxane	123-91-1	Very irritating to eyes, skin and airways. Possibly carcinogenic. LC ₅₀ mice inhalation. 37 g/m ³ /2 h LC ₅₀ rats inhalation. 46 g/m ³ /2 h	Limits (1996) LLV 35 mg/m ³ TV 90 mg/m ³ (H) The substance can easily be absorbed through the skin. (C)The substance is carcinogenic	Highly flammable liquid and vapour (H225), Causes serious eye irritation (H319), May cause respiratory irritation (H335), Suspected of causing cancer (H351), May form explosive peroxides (EUH019), Repeated exposure may cause skin dryness or cracking (EUH066)
1,4-oxathiane	15980-15-1	NA	---	# Flammable liquid and vapour (H226), Causes skin irritation (H315), Causes serious eye irritation (H319) May cause respiratory irritation (H335)
1-methylpyrrolidin-2-one	872-50-4	Irritating to eyes and skin. Headaches and chronic eye irritation have been reported at levels of 2.9 g/m ³ .	Limits (2015) LLV 40 mg/m ³ STV 80 mg/m ³ (H) The substance can easily be absorbed through the skin. (R) The substance may impair fertility.	Causes skin irritation (H315), Causes serious eye irritation (H319), May cause respiratory irritation (H335), May damage fertility or the unborn child (H360D)
1-phenylethanone	98-86-2	Irritating to eyes and skin	---	Harmful if swallowed (H302), cause serious eye irritation (H319)
2-methyl-1,3-dioxolane	497-26-7	Induce muscle weakness LC ₅₀ mice inhalation.:59 g/m ³ LC ₅₀ rats inhalation.:80 g/m ³ (ChemIDplus)	---	# Highly flammable liquid and vapour (H225), Causes serious eye irritation (H319).

Chemical compound	CAS no. ¹	TOXNET ² Skin eyes, respiratory tract	Hygienic limit values ³	Labelling of chemicals according to ECHA ⁴ : Hazard codes for chem-phys & human health
2-methylcyclopent-2-en-1-one	1120-73-6	NA	---	# Flammable liquid and vapour (H226)
6-methyloxan-2-one	823-22-3	NA	---	# Causes serious eye irritation (H319)
Benzoic acid	65-85-0	Vapours may induce irritation to eyes, skin and airways. (Mild irritant)	---	Causes skin irritation (H315), Causes serious eye damage (H318), Causes damage to the lungs through prolonged or repeated exposure (H372)
Cobalt (Co)	7440-48-4	Cough, shortness of breath, wheezing, asthma. May induce itching at skin exposure.	Group limit values (2011) LLV Inhalable dust 0.02 mg/m ³ (C) The substance is carcinogenic (H) The substance can easily be absorbed through the skin. (S) The substance is sensitizing.	May cause an allergic skin reaction (H317). May cause allergy or asthma symptoms or breathing difficulties if inhaled (H334)
Diethyl phosphofluoridate	358-74-7	An organophosphorus chemical. Old studies indicate that the chemical is toxic after exposure via skin or inhalation (ChemIDplus).	---	NA
Diethylcarbonate (DEC)	105-58-8	Very irritating to eyes, skin and airways.	---	# Flammable liquid and vapour (H226)
Dimethyl carbonate (DMC)	616-38-6	Irritating to eyes, skin and airways. LD ₅₀ rats inhalation. ≥14 mg/m ³ /4h	---	Highly flammable liquid and vapour (H225)
Ethyl phosphonofluoridate	650-20-4	NA, but is an organophosphorus chemical	---	NA

Chemical compound	CAS no. ¹	TOXNET ² Skin eyes, respiratory tract	Hygienic limit values ³	Labelling of chemicals according to ECHA*: Hazard codes for chem-phys & human health
Ethyl methylphosphonofluoridate	673-97-2	NA, but is an organophosphorus chemical	---	NA
Ethyl prop-2-enyl carbonate	1469-70-1	NA	---	Flammable liquid and vapor (H226), Causes skin irritation (H315), Causes serious eye irritation (H319)
Ethyl propyl carbonate	35363-40-7	NA	---	NA
Ethylbenzene	100-41-4	Irritating to eyes, skin and mucous.	Limits (2015) LLV 220 mg/m ³ STV 884 mg/m ³ (H) The substance can easily be absorbed through the skin.	Highly flammable liquid and vapor (H225), May be fatal if swallowed and enters airways (H304), Harmful if inhaled (H332), May cause damage to organs (hearing) (H373)
Ethylmethyl carbonate (EMC)	623-53-0	NA	---	# Flammable liquid and vapor (H226), Causes skin irritation (H315), Causes serious eye irritation (H319), May cause respiratory irritation (H335)
Hydrogen fluoride (HF)	7664-39-3	Highly toxic and highly corrosive. Dermal exposure leads to severe burns. HF is adsorbed through skin. Inhalation can lead to pulmonary edema and pneumonia. Skin or inhalation may provide high risk of hypocalcemia.	Limits (2015) NGV 1.5 mg/m ³ STV 1.7 mg/m ³ For exposure to a mixture of fluorides and hydrogen fluorides, the level limit value for fluorides shall be applied.	Fatal if swallowed (H300), Fatal in contact with skin (H310), Causes severe skin burns and eye damage (H314), Fatal if inhaled (H330)

Chemical compound	CAS no. ¹	TOXNET ² Skin eyes, respiratory tract	Hygienic limit values ³	Labelling of chemicals according to ECHA ⁴ : Hazard codes for chem-phys & human health
Lead (Pb)	7439-92-1	Both skin and respiratory exposure can lead to systemic effects including effects on the nervous system (CDC). Suspected carcinogen.	Group limit value (2011) LLV Inhalable dust 0.1 mg/m ³ Respirable dust 0.05 mg/m ³ (B) Exposure approaching existing professional hygienic limit values and simultaneous exposure to noise levels approaching the action value of 80 dB can cause damage to hearing. (M) Medical supervision may be required for handling of this substance. (R) The substance may impair fertility.	NA
Lithium (Li)	7439-93-2	Corrosive with risk for serious effects on eyes, skin and mucus. Risk for oedema and corrosive damage on the respiratory tract.	Group limit value (2011) STV Inhalable dust 0,02 mg/m ³	In contact with water releases flammable gases which may ignite spontaneously (H260), Reacts violently with water (EUH014), Causes severe skin burns and eye damage (H314)
Manganese (Mn)	7439-96-5	Can be neurotoxic chemical	Group limit values (2000) NGV Total dust 0.2 mg/m ³ Respirable dust 0.1 mg/m ³ ---	NA
Methyl methylphosphonofluoridate	353-88-8	NA, but is an organophosphorus chemical		NA

Chemical compound	CAS no. ¹	TOXNET ² Skin eyes, respiratory tract	Hygienic limit values ³	Labelling of chemicals according to ECHA*: Hazard codes for chem-phys & human health
Nickel (Ni)	7440-02-0	Nickel may sensitization of the skin.	Group limit value (1978) LLV Total dust 0.5 mg/m ³ (S) The substance is sensitizing.	Refers to dust of nickel (<1mm in diameter) May cause an allergic skin reaction (H317), Suspected of causing cancer (H351), Causes damage to organs through prolonged or repeated exposure (H372)
Styrene	100-42-5	Irritating to eyes, skin and mucous. Health effects include changes in color vision, fatigue, reduced alertness, concentration and balance.	Limits (2011) LLV 43 mg/m ³ STV 86 mg/m ³ (H) The substance can easily be absorbed through the skin. (B)Exposure approaching existing professional hygienic limit values and simultaneous exposure to noise levels approaching the action value of 80 dB can cause damage to hearing. (M) Medical supervision may be required for handling of this substance.	Flammable liquid and vapor (H226), Causes skin irritation (H315), Causes serious eye irritation (H319), Harmful if inhaled (H332), Suspected of damaging fertility or the unborn child (H361d), Causes damage to organs (hearing) (H372)
Tin (Sn)	7440-31-5	Irritating to eyes, skin and respiratory system.	Group limit value (2011) LLV Inhalable dust 2 mg/m ³	NA

Footnotes and affiliations, see next page.

¹CAS no.: Chemical number according to Chemical Abstract Service

²TOXNET: Toxicology data network at NIH (<http://toxnet.nlm.nih.gov/>). The Hazardous Substances Data Bank was used as the database if not otherwise stated. The information presented here focuses on toxicity after exposure via the skin, respiratory tract and eyes.

³Hygienic limit values. Information from (www.av.se; AFS 2015:7 in Swedish).

⁴Labelling according to CLP. Information from ECHA (<http://echa.europa.eu/>). Labelling of chemical, physical and human health hazards are presented giving an indication of possible effects.

[#]The chemical does not have a harmonized classification and labelling. Labelling is presented in the table when most companies have registered the same phrases.

LD₅₀: Lethal dose where 50% of the individuals die.

LC₅₀: Lethal concentration where 50% of the individuals die.

LLV: Level limit value (Nivågränsvärde in Swedish).

STV: Short-term value (Korttidsvärde in Swedish). These values are used for toxic chemicals that do not have Ceiling limit values.

NA: Not available

4 Discussion and conclusion

Among the detected compounds there are several, to our knowledge, of toxic concern previously not reported to ventilate during thermal runaway of Li-ion batteries. Many of the compounds are irritating to the skin and the respiratory system. Whether the exposure of vented gases reach a dangerous dose in the case of real world accident involving Li-ion batteries cannot be concluded from this study. To assess first responders' occupational exposure risk, as well as victim exposure in e.g. a car accident, a full scale experiment is needed including propagation of thermal runaway in a stack of battery cells.

Furthermore, three of the detected compounds; ethyl ethylphosphonofluoridate, ethyl methylphosphonofluoridate, and methyl methylphosphonofluoridate are structurally very similar to the chemical warfare agent sarin but as previously stressed, their real toxicity in a realistic e-vehicle traffic accident scenario cannot be estimated based on the present study. Studies should be conducted to determine if these affect the neurotransmitter regulator enzyme acetylcholinesterase and until then extra attention should be paid.

Also the reverse is true, previously reported PF_5 , POF_3 , $\text{POF}(\text{OH})_2$, $\text{POF}_2(\text{OCH}_2\text{CH}_2)_n\text{F}$, and POF_2OMe [Wilken *et al.* 2012, Wilken *et al.* 2013, Grutzke *et al.* 2015] as well as 2-fluoroethanol and bis-(2-fluoroethyl)-ether [Hammami *et al.* 2003] are not observed here. This study is a pilot study of an entire pouch cell in humidified nitrogen atmosphere whereas previous studies focus on isolated systems of electrolyte and/or electrode materials. This may explain the difference in observed compounds, and the reaction pathways outlined [Wilken *et al.* 2013] may not necessarily be very different from this study.

Some unforeseen problems in the project were encountered. Breakthrough of carbonates from Tenax to Carbosieve adsorbents made the analysis of smaller gaseous compounds such as HF, PF_5 , and POF_3 cumbersome. Luckily, FT-IR is a technique suited to study these compounds. Time evolution of IR spectra has not been studied here but it is something that would gain further insight in the evolution pathway chemistry.

We experienced this project to be too small for a satisfactory and full chemical analysis of ventilated Li-ion batteries. This included as full spectral assignment of vibrational modes in FT-IR spectra, time-evolution analysis of the same, adsorbent sampling separating on volatility to avoid contaminating Carbosieve adsorbent, and analysis of TSP fraction with regards to less volatile organic and in-organic contents.

We did learn some good lessons from the experimental work. Because of the complex environment we recommend specific sensor for certain compounds, e.g. HF and CO. Online FT-IR measurements (GASMET) can be used to indirect (by CO_2 declination) monitor oxygen levels to avoid open flame of the vented gases. Last, FT-IR spectral changes was the first indicator of chemical emitted from battery ventilation, before any greater amount of visual smoke.

5 Literature cited

Fredholm Lotta, Saarkoppel Fredrik. In Swedish. Sänkt syrehalt förebygger bränder. BrandSäkert, 2012, Vol.7 pp. 28-30, www.brandsakert.se

Grutzke M., Kruger S., Kraft V., Vortmann B., Rothermel S., Winter M., Nowak S., Investigation of the Storage Behavior of Shredded Lithium-Ion Batteries from Electric Vehicles for Recycling Purposes. ChemSusChem, 2015. 8(20): p. 3433-8.

Hammami Amer, Raymond Nathalie and Armand Michel. Runaway risk of forming toxic compounds. Nature, 2003, Vol. 424, pp. 635-636.

Huang Qian, Yan Manming and Jiang Zhiyu. Thermal study of organic electrolytes with fully charged cathodic materials of lithium-ion batteries. J. Solid State Electrochem., 2008, Vol. 12, pp. 671-678. DOI 10.1007/s10008-007-0400-7.

Kawamura Tetsuya, Okada Shigeto and Yamaki Jun-ichi. Decomposition reaction of LiPF₆-based electrolytes for lithium ion cells. J. of Power Sources, 2006, Vol. 156, pp. 547-554.

Roth E. P. Abuse Response of 18650 Li-Ion Cells with Different Cathodes Using EC:EMC/LiPF₆ and Screening of Vented Gases and Aerosols from Li-Ion LFP and NMC Batteries under Nitrogen atmosphere EC:PC:DMC/LiPF₆ Electrolytes., ECS Transactions, 2008, Vol. 11, pp. 19-41.

Sturk David, Hoffman Lars. In Swedish. E-fordons Potentiella Riskfaktorer vid en Trafikskadehändelse. 2013, Rapport från FFI projektet Räddningskedjan. Available at <http://e-fordon.blogspot.se/>.

Sturk David, Hoffmann Lars and Tidblad Annika Ahlberg. Fire tests on E-Vehicle Battery Cells and Packs. Traffic Injury Prevention. Feb 25, 2015.

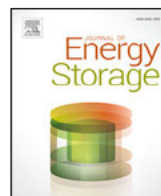
Wang Qingsong, Sun Jinhua, Yao Xiaolin, Chunhua Chen., Micro calorimeter study on the thermal stability of lithium-ion battery electrolytes. J. of Loss Prevention in the Process Industries, 2006, Vol. 19, pp. 561-569. doi: 10.1016/j.jlp.2006.02.002.

Wilken Susanne, Treskow Marcel, Scheers Johan, Johansson Patrik and Jacobsson Per, Initial stages of thermal decomposition of LiPF₆-based lithium ion battery electrolytes by detailed Raman and NMR spectroscopy. RSC Advances, 2013, Vol. 3, pp. 16359-16364.

Wilken Susanne, Johansson Patrik and Jacobsson Per. Infrared spectroscopy of instantaneous decomposition products of LiPF₆-based lithium battery electrolytes., Solid State Ionics, 2012, Vol. 225, pp. 608-610.

Yang Hui *et al.*, A Review of Li-Ion Cell Chemistries and their Potential use in Hybrid Electric Vehicles., J. Ind. Eng. Chem., 2006a, Vol. 12, pp. 12-38.

Yang Hui, Zhuang Guorong V and Ross Jr. Philip N. Thermal stability of LiPF₆ salt and Li-ion battery electrolytes containing LPF₆., J. of Power Sources, 2006b, Vol. 161, pp. 573-579. doi: 10.1016/j.jpowsour.2006.03.058.



Review article

Review of gas emissions from lithium-ion battery thermal runaway failure — Considering toxic and flammable compounds

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ABSTRACT

Lithium-ion batteries (LIBs) present fire, explosion and toxicity hazards through the release of flammable and noxious gases during rare thermal runaway (TR) events. This off-gas is the subject of active research within academia, however, there has been no comprehensive review on the topic. Hence, this work analyses the available literature data to determine how battery parameters affect the variation in off-gas volume and composition, to determine the flammability and toxicity hazards of different battery chemistries. It is found on average that: (1) NMC LIBs generate larger specific off-gas volumes than other chemistries; (2) prismatic cells tend to generate larger specific off-gas volumes than other cell forms; (3) generally a higher SOC leads to greater specific gas volume generation; (4) LFP batteries show greater toxicity than NMC; (5) LFP is more toxic at lower SOC, while NMC is more toxic at higher SOC (relative to themselves); and (6) LFP off-gas has a greater flammability hazard. Further, recommendations are presented so that significant improvements in research can be made to advance the understanding of LIB off-gas further. Finally, this work is a critical resource to the battery community to aid the risk assessment of LIB TR fire, explosion and toxicity hazards.

1. Introduction

As with traditional transportation fuels (i.e. hydrocarbons such as petrol and diesel), electrochemical energy sources like Li-ion batteries (LIBs) have inherent safety risks related to fire and toxicity [1–5]. The maturity of the internal combustion engine (ICE) means that the hazards of vehicle fires under different scenarios are well understood. However, with the accelerating uptake of LIB-powered electric vehicles (in the road, rail and marine sectors) there is a concern that there will be a growing knowledge gap between understanding the hazards of traditional and alternative fuel vehicles [6]. There have been several reviews of the growing literature on LIB safety, focusing on the mechanisms of cell thermal runaway (TR) and fire phenomenon [6,7]. However there are no reviews that specifically and comprehensively analyse LIB off-gas which is known to be flammable (containing hydrogen and hydrocarbons) leading to explosion and loss of life [8,9]. Further, for those involved in or near a TR event, toxic gas emissions (e.g. CO and HF) present an additional threat. However, the composition of the off-gas is complex and needs extensive consideration.

The risk of fire, explosion or vapour cloud ignition extends to stationary energy storage, EVs and marine applications, where incidents have occurred in reality [e.g. 9–11], showing that this is a real and present hazard. Adequate risk assessments are required to manage and mitigate this fire/explosion hazard and to aid emergency responders in understanding hazards they may walk into [12,13]. However, this can only be done with accurate and extensive knowledge of the off-gas volume and components (i.e. H₂, hydrocarbons, flammable solvent vapours, CO₂ dilution effect) to determine the explosion potential of the off-gas in a given scenario. This is also true for the toxicity of the off-gas, where determining the extent to which individuals are at risk of breaching *Immediate Danger to Life and Health* or exposure limits is essential. For example, in scenarios where individuals are exposed to off-gas in confined spaces, such as from the failure of personal transport stored in the home [14,15] or EV failure in a car garage [4,16]. As will be shown below, there is no extensive analysis of LIB off-gas considering the influence of battery variables, which is necessary for a thorough understanding of its hazards.

There has been some work to understand the overall off-gas behaviour. Baird et al. [17] compiled the gas emissions of ten papers

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Nomenclature

<i>DEC</i>	Diethyl carbonate
<i>DMC</i>	Dimethyl carbonate
<i>EC</i>	Ethylene carbonate
<i>EMC</i>	Ethyl methyl carbonate
<i>EV</i>	Electric vehicle (automotive)
<i>HRR</i>	Heat release rate
<i>ICE</i>	Internal combustion engine
<i>IQR</i>	Inter quartile range
<i>LCO</i>	Lithium cobalt oxide
<i>LFL</i>	Lower flammability limit
<i>LFP</i>	Lithium iron phosphate
<i>LIB</i>	Lithium ion (Li-ion) battery
<i>LMO</i>	Lithium manganese oxide
<i>LTO</i>	Lithium titanate
<i>NCA</i>	Lithium nickel cobalt aluminium oxide
<i>NMC</i>	Lithium nickel manganese cobalt oxide
<i>PC</i>	Propylene carbonate
<i>SEI</i>	Solid electrolyte interphase
<i>SOC</i>	State of charge
<i>THC</i>	Total hydrocarbons
<i>TR</i>	Thermal runaway

showing gas composition related to different cell chemistries and SOC, while Li et al. [18] compiled the gas emissions of 29 tests under an inert atmosphere. However, in both cases, no analysis is made relating chemistry, SOC, etc. to off-gas composition. Instead, flammability limit calculations and flame speed analysis were conducted. The most compressive review to date is by Rappsilber et al. [19] who analysed 76 papers focusing on peak heat rate, total heat, CO emission, total gas volume, and HF production. However, the overall volume and specific CO emissions trends are determined by grouping all the cell chemistry data for each SOC, while for HF only LFP cells are analysed.

This work aims to address the lack of a comprehensive review of LIB gas emissions during TR via collating and analysing data available in the literature. Within this aim the objectives are to understand how battery parameters affect the variation in off-gas volume and composition, and what battery can be considered least hazardous. Overall it provides a crucial resource that can be used in the risk assessment of LIB TR fire and explosion hazards.

The remainder of the introduction will outline the demand for LIBs, explain what a LIB is and describe the hazardous thermal runaway phenomenon. Section 2 will then review the literature on LIB off-gas composition and toxicity. In Section 3 the methods are presented for gathering and processing the literature data. Section 4 presents the analysis and discussion of the literature data. Section 5 provides concluding remarks.

1.1. Overview of lithium-ion batteries and thermal runaway

From powering personal electronics devices, LIBs are becoming the widely adopted energy source for automotive electric vehicles (EVs) and personal transport (bicycles and scooters). They are also increasingly used in ships and trains as auxiliary or primary power sources. Further, the aviation sector is also developing EVs based on LIB energy sources for electric vertical take-off and landing (eVTOL) as well as for light/short hall aircraft. Lastly, LIBs are also used in commercial battery energy storage (BESS) for grid support as well as domestic energy storage. With such growing use in terms of quantity and scale, there are increasing opportunities for LIB failure to cause greater harm. This is specifically true in certain situations where failure is more critical,

such as in the marine and aviation sector where there is limited scope to distance people and assets from the failure.

LIBs are widely used as they have a high energy density, long cycle life and are low cost. Cell performance can be altered by materials selection, with common cell chemistries consisting of lithium cobalt oxide (LCO), lithium iron phosphate (LFP), lithium manganese oxide (LMO), lithium nickel cobalt aluminium oxide (NCA), lithium nickel manganese oxide (NMC) and lithium titanate (LTO). Generally NMC and NCA are used as high-energy cells, while LFP cells have a lower specific energy capacity but a larger specific power capacity and longer life span [20]. Further, battery performance (energy and power capabilities) is scalable by the number and configuration (series and parallel connections) of cells. Cells can be manufactured in different forms, commonly cylindrical, pouch or prismatic with capacities ranging from 1 Ah to 300 Ah. These characteristics allow LIBs to be customised for the variety of different applications mentioned above. However, even with these advantages, LIBs suffer from the phenomenon of thermal runaway (TR), explained in the next section.

1.1.1. LIB thermal runaway

The process of TR, see Fig. 1, involves the exothermic chemical decomposition of the battery cell materials leading to vast heat generation and temperature rise. This is accompanied by the generation of gasses from the decomposition process that can be flammable and toxic, and can lead to smoke, hot sparks and jet flames ejected from the cell [6]. Many reactions take place during the decomposition of the cell; however, the main stages consist of solid electrolyte interphase (SEI) breakdown, anode-electrolyte reaction, cathode-electrolyte reactions and electrolyte decomposition, see Table 1 for further details. Further, in a module or pack the heat from one cell can cause a cascading failure or propagation throughout the pack, increasing the overall hazard from failure. For module or pack failure this includes the production of large amounts of flammable gas that can lead to explosions [8,9].

Typically this occurs when the off-gas is confined to an enclosed space, thus not naturally dissipating like in the open, and either (1) an ignition source occurs when the fuel-air ratio is in the explosion range [8]; or (2) if additional air is allowed to enter (by opening a door to address the failure [9]) bringing the fuel-air ratio from above the upper explosion limit (UEL) into the explosion range. However, at least one unconfined off-gas explosion has taken place, where a large cloud of smoke accumulating around an electric bus suddenly erupts in flames [10], indicating that this smoke is a vapour cloud igniting [21]. This behaviour is complex, so to understand this hazard more it is important to determine how much volume and what species of gases are generated for different cell chemistries, forms, SOC, scale (cell/module/pack) and abuse.

2. Gas generation and toxicity — literature review

This section summarises the findings of individual literature sources regarding volume of gas produced (Section 2.1), gas composition (Section 2.2), toxicity (Section 2.3), presence of electrolyte vapour (Section 2.4), other influential factors including the effect of abuse scenarios (Section 2.5) and results from module and pack tests (Section 2.6).

2.1. Gas volume produced

The volume of gas LIBs generate during TR can be influenced by several conditions, including capacity, SOC and chemistry. At 100% SOC LFP cells are shown to generate a lower volume of gas than other chemistries (LCO, NMC, LMO), 0.4 L/Ah to 1.4 L/Ah versus 1.28 L/Ah to 21 L/Ah respectively [24–32]. In contrast, large NMC prismatic cells (41 Ah) and LFP cells (5.5 Ah) have been shown to generate similar off-gas volumes, 1.64 L/Ah and 1.83 L/Ah respectively (in nitrogen) [33]. While, LTO cells have been shown to generate significantly more off-gas than LFP cells, but less than NMC [34]. Further, higher nickel-content

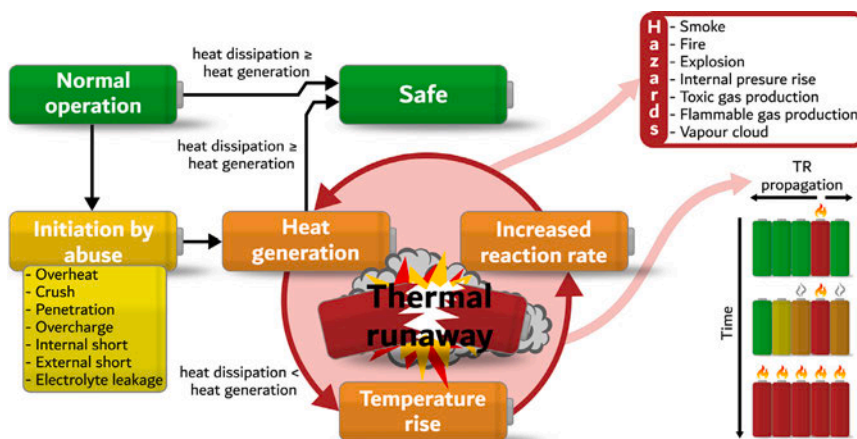


Fig. 1. Process of thermal runaway from initiation to propagation and resulting hazards [22].

Table 1

Summary of main reactions of LIB cell TR failure.

Source: Edited from [23].

Temperature (°C)	Reaction behaviour
>70	Li salt decomposition and reaction with solvent and Solid electrolyte interphase (SEI)
90–130	SEI breaks down leading to anode-electrolyte reaction. Low heat generation.
90–230	Li-electrolyte reaction occurs, leading to gas production e.g. C_2H_4 , C_2H_6 and C_3H_6 .
120–220	Electrolyte vaporises, leading to additional gas generation, cell pressurisation and initial venting. Separator melts at 130°C to 190°C.
160	Heat generation increases — from “self heating” to “thermal runaway”. Violent gas and particle release (second venting).
200–300	Electrolyte decomposition occurs. At TR there is a rapid temperature rise, the metal oxide cathode decomposes to produce oxygen. O_2 leads to the oxidation of the electrolyte $\rightarrow CO_2$ and H_2O

NMC cells produce more gas than counterparts with less nickel [31,32]. Overall, the absolute vent gas volume increases with capacity [31, 32,35]. Limited work has compared cell forms, but NMC pouch and prismatic cells have been shown to generate similar volumes of gas production [36].

In many studies gas volume production is shown to increase as SOC increases for various chemistries, including NMC [23,30,37–39], LFP [25,40], NCA [25] and LCO [26]. However, little to no correlation between gas volume and SOC is observed in some studies on LFP and NMC cells [25,41]. Further, the greatest increase in emissions is typically seen over the range of 50% to 100% SOC [7,30]. Also, the increase in gas production with SOC has shown to be greater for NCA than for LFP cells. While at higher SOC LFP is typically shown to produce less off-gas than other chemistries, at lower SOC volumes can be comparable between chemistries, but in some cases LFP can generate more [7,25]. Increased gas generation at higher SOC is attributed to higher electrode potentials and more reactive cell materials [7]. The occurrence of combustion may also have an effect. When the failure of NMC cells did not lead to combustion (at 0% SOC) more gas production was recorded [41].

The peak rate of gas production has been shown to be proportional to SOC [7,39], while in other work this is not true [42]. Peak gas rates of individual species (CO , CO_2 and HF) have been shown to increase with SOC [43].

2.2. Gas composition

The main components of the off-gas are CO_2 , CO , H_2 and hydrocarbons, while under certain conditions significant amounts of electrolytes

and water can be present. The list of hydrocarbons generated can be many, but those typically measured/reported are CH_4 , C_2H_6 , C_2H_4 , C_2H_2 , C_3H_6 and C_3H_8 . Further, there are possibly hundreds of components present in the off-gas, including non-hydrocarbons and toxins, that are infrequently reported [44–46]. The toxic components of main concern are CO , HF , HCl , HCN , NO_x , and SO_2 [7]. Table 2 summarises the hazards and exposure limits of the main off-gas components. Exposure limits are the concentrations of hazardous substances in air, averaged over 8 h or 15 min (long-term and short-term limits respectively), that are set to prevent harmful effects on people's health [47]. The reaction mechanisms towards species production are well discussed in the literature [see, for example, 24,25,32,37,48–51] and are not a topic for discussion here.

Comparing the off-gas composition from different sources highlights the variation in components and their volume fractions between and within chemistries. Typically, literature sources identify LFP as generating more CO_2 and H_2 than other chemistries that generate mostly CO and THC , in terms of volume fraction [24,32,34,56]. However, high levels of H_2 , CO_2 and lower levels of CO have been reported in NMC cells [33], while higher CO_2 volumes have been reported for NCA compared to LFP cells [25]. It has been reported that the percentage of CO and H_2 production is proportional and inversely proportional to energy density, respectively [32]. This inherently links to nickel content for the NMC cells; however, later work shows less correlation [31]. Further, comparing solid-state cells versus “typical” Li-ion cells shows a reduction in gas production (by 10%) and a lower proportion of THC and H_2 (by 30% each), while the proportions of CO_2 and CO are larger (by 20% for CO) for the solid-state cells [57].

Koch et al. [27] have done one of the most extensive testing regimes of 51 NMC cells ranging from 20 Ah to 80 Ah. Significant amounts of

Table 2
Common LIB off-gas components, hazards and exposure limits.

Substance	Hazard ^a	Exposure Limit, 8 h (mg/m ³) ^b	Exposure Limit, 15 min (mg/m ³) ^b
Carbon dioxide, CO ₂	Cause headaches, dizziness, confusion, loss of consciousness, and asphyxiation at high concentrations [52].	9150	27 400
Carbon monoxide, CO	Toxic if inhaled, may damage the unborn child, causes damage to organs through prolonged or repeated exposure and is an extremely flammable gas.	23	117
Hydrogen, H ₂	Extremely flammable.		See note ^d
Hydrocarbons	Flammable.		See note ^d
Hydrogen fluoride, HF	Fatal if swallowed, is fatal in contact with skin, is fatal if inhaled and causes severe skin burns and eye damage.	1.5	2.5
Hydrogen chloride, HCl	Severe skin burns and eye damage, is toxic if inhaled, may damage fertility or the unborn child, causes serious eye damage, may cause damage to organs through prolonged or repeated exposure, may be corrosive to metals, may cause respiratory irritation and contains gas under pressure and may explode if heated.	2	8
Hydrogen cyanide, HCN	Fatal if swallowed, is fatal in contact with skin, is fatal if inhaled, causes damage to organs through prolonged or repeated exposure, is very toxic to aquatic life (with long lasting effects) and is an extremely flammable liquid and vapour.	1	5
Nitrogen dioxide, NO ₂	Fatal if inhaled, causes severe skin burns and eye damage; and may cause or intensify fire (oxidiser).	0.96	1.91
Sulphur dioxide, SO ₂	Severe skin burns and eye damage and is toxic if inhaled.	1.3	2.7
Solvents	Highly flammable liquid and vapour [53]. Very irritating to eyes, skin and airways [44].	DEC 700, PC 8.5 ^c	DEC 1000, PC 8.5 ^c

^a Cited from the European Chemicals Agency [53], unless otherwise stated.

^b Cited from the HSE [47], unless otherwise stated.

^c Cited from the IFA [54] for the solvents DEC (Romania) and PC (Germany), other common electrolyte solvents (e.g. EC and DMC) are not listed.

^d Hydrocarbons, such as CH₄, are described as acting as simple asphyxiants without other significant physiologic effects when they are present in high concentrations, hence are not given limit values as the significant factor is the availability of oxygen [55]. This is assumed for H₂ as well.

CO₂ are recorded at low-vent gas volumes as combustion could still be complete even in the limited oxygen availability of the closed system. Koch et al. found that (for vent gas volumes greater than 50 l where combustion was limited) 7 components make up over 99% of the off-gas volume. These components (and their average volume proportions) are CO₂ (36.6%), CO (28.3%), H₂ (22.3%), C₂H₄ (5.6%), CH₄ (5.2%), C₂H₆ (1.0%) and C₃H₆ (0.5%). These averages have a large variation, with CO, CO₂ and H₂ being within the error of each other. Further, there is no correlation between composition and the vent gas volume.

SOC is shown to affect the off-gas composition ratio and the number of components produced. At 0% SOC the gas can be mainly CO₂ [17], or a few components (~5) including CO and solvents, while this can reach several tens of components at higher SOC [58]. Generally, it has been found that the percentage volume of CO₂ in the off-gas reduces while CO increases as SOC increases for various chemistries (LCO, NCA, LFP) [7,17,25,26,59]. At the same time, THC and H₂ have been found to remain steady across SOC for LCO cells [26]. For LFP and NCA cells H₂ is seen to increase in volume fraction [25], while H₂ and THC are found to increase in volume fraction for LCO and LFP cells [17], with SOC. However, other works have shown contradictory results, where CO₂ increases and THC reduces with SOC increase [37], or the fraction of THC reduces along with CO₂ with SOC increase [59]. Also, H₂ was only detected at 100% for LFP cells [42].

Peak concentrations of gases and production rates increase with SOC [39,60,61]. However, the reduction of CO₂ volume fraction can be attributed to the increase in absolute production of other gases while CO₂ concentration remains stable [60,62]. The reduction of hydrocarbons and increase in CO and CO₂ at higher SOC (in a nitrogen atmosphere) has been attributed to combustion events [37,39], however Willstrand et al. [7] shows contradictory behaviour. Nevertheless,

it has been noted that higher SOC enhances combustion, leading to a greater HRR and higher concentrations of CO₂ and CO [63,64]. More so, the relationship of combustion efficiency (turning CO to CO₂) is complex and dependent on temperature (generally proportional to SOC), abuse method, oxygen availability and gas generation rate [7].

Studying the flammability limit of cell off-gas, for NMC cells (18650, 3.5 Ah) the lower explosion limit (LEL) increases up to a maximum of 10.15% at a SOC of 50% and then reduces at higher SOC, while the upper explosion limit (UEL) continually increases [65]. This means a SOC of 50% has the lowest risk of combustion in an open space. However, the narrowest flammability range is at a SOC of 30%, hence this SOC is recommended for the storage and transport of LIBs. The trend in LEL is attributed to the change in multi-carbon chain gas components. When there are more gas components with longer carbon-chains the LEL reduces.

Electrolyte composition is shown to affect the off-gas from cell abuse. For different LFP 18650 cells, where EMC is always present and makes up approximately 50% of the electrolyte solvent, there is a weak trend between increased EMC proportion and reduced gas generation, while the percentage of CO₂ increases with increased EMC proportion [66]. Studies on electrolytes alone have shown that the concentration of lithium salt affects the ratio of off-gas species, while an increase in salt concentration leads to greater instability and gas production [67]. The individual solvents (e.g. EC, DEC, EMC and DMC) also vary in stability and gas volume production, with EC and DEC being the least thermally stable and reactivity increasing with the addition of LiPF₆ salts [68]. Further, the use of binary mixtures leads to electrolytes that have stability broadly related to the solvent-mole ratios. For EC, EMC and DMC solvents with 1.2M LiPF₆ gas emissions are dominated by CO₂ (70%), but for DEC (1.2M LiPF₆) CO₂ and C₂H₆

account for 30% each; however, tens of components are recorded in total in all cases. Similar behaviour has been shown by Fernandes et al. [69], but with thermal decomposition of EMC and particularly DMC occurring at lower temperatures. Further, there is uncertainty in the actual reaction mechanism for the thermal decomposition of battery solvents [70].

Additionally, the presence of the cathode and anode materials in a cell complicates the reaction behaviour. Intercalated lithium in the anode can react with the solvents to produce hydrocarbons, while oxygen released from the cathode decomposition can lead to decomposition [71,72]. The reaction pathways to gas generation are numerous and complex and the readers are referred to existing reviews on the mechanism of thermal runaway, see Refs. [71,72]. Additives have also been used to improve anode-electrolyte stability and impede fires [73]. Flame retardants can reduce fires leading to less CO₂ but more H₂ and CO. Further, in the near future it is expected that all-solid-state cells will be commercial, in which their solid electrolyte has increased stability and reduced heat of reaction, dramatically improving cell safety but with little know of the amount and type of gas produced [74]. Due to these numerous variables (i.e. solvents, solvent ratios, salt concentrations and additives) and that few works know or record the electrolyte composition in battery safety/off-gas tests, electrolyte composition is not considered in further discussions as a variable affecting off-gas characteristics.

2.3. Toxicity

Several toxic compounds commonly discussed include CO, HCl, HCN, NO, SO₂, HF, fluorinated carbonates, POF₃, COF₂, acrolein, and formaldehyde [29,38,50,60,61,75–77]. Non-quantified work has detected up to 35 toxic substances which vary with chemistry and SOC [78]. Understanding the presence of these compounds is important as inhalation of these gasses can result in dizziness, headache, loss of consciousness, coma or even death [6].

The risks of these emissions, however, depend on the scenario of the incident. In the outside environment, HF will likely rise and quickly dissipate. Whereas, in enclosed spaces HF will increase and be problematic if gasses are not evacuated. More full-scale tests are needed to fully establish this and assess the effects of fire suppression on toxicity.

For LMO cells the total amount of HCl generated remains stable with SOC but the amount of SO₂ increases with SOC, while NO shows no correlation [60]. The change in HCN concentration is also shown to be minimal over increased SOC [38]. For NMC prismatic cells (94 Ah), more SO₂ was produced at 50% SOC than at 100% SOC while the maximum temperature was 510°C compared to 620°C, respectively [77]. The rate of HF production increases with SOC [61,75,76], while for a stack of LFP pouch cells the peak HF production rate is seen at 50% SOC [79].

For LCO, NCA and LFP cells, it is shown that there is a downward trend of specific HF production with SOC increase [50,60,61,75]. In some cases for LFP cells, HF is only detected at low SOC (at or below 25% SOC) [7]. However, peak HF production for LFP cells has been seen at 50% SOC [79] or 100% SOC [80]. Also, CO and HF production has been shown to increase with SOC (for 22 Ah LFP cells) [42].

Comparing HF production between cell chemistries, LCO prismatic and LFP cylindrical cells are shown to produce similar specific amounts of HF, so do NCA and LFP pouch cells [76], as well as LFP and LMO/NMC [29]. However, Sturk et al. [75] show that LFP cells generate significantly more HF than their NMC counterparts. HF is found to be the largest contributor to toxicity for LFP cells, while for LCO and NMC it is Acrolein [50]. From this, LFP cells produce the largest theoretical contamination volume (300 m³) versus LCO and NMC cells (200 m³) (considering 18650 cells). Overall, there is no strong trend for the theoretical contamination value with SOC, because as HF decreases with SOC there is an increase in other toxins such as CF₄.

For other toxins, the number of combustion organic products increases with SOC, up to a SOC of 100%, and then there is a significant reduction at a SOC of 150% [81]. In total, over a hundred species were detected, but more products are seen in LCO, NMC and LMO cells than in LFP cells. For NMC and LCO cells a higher SOC showed more substances, while for LFP cells the greatest amount of substances was detected at 30% SOC [78]. Six very toxic substances identified as 2-propenal, methyl vinyl ketone, propanedinitrile, propanenitrile, 1,2-dimethyl-hydrazine and thiocyanic acid ethyl ester could be detected. Further, phosphonofluoridates have been detected in NMC but not LFP cells [44].

The cell form may influence HF production. LFP pouch cells are shown to produce an order of magnitude more HF than cylindrical cells [60,76]. However, NMC pouch and prismatic cells show similar volumes and composition of gas, including HF [36]. Further, abuse type can affect toxic gas generation. Thermal abuse produces a greater contamination volume than nail penetration, as penetration overall produces less gas [50].

While the dependence of SO₂ generation on SOC is not definitive, one theory attributes it to the fact that the formation of SO₂ requires significant heat [60]; hence, it only occurs at higher SOC where the TR temperature is higher. Conversely, less HF is theoretically formed at higher SOC as the higher TR temperatures promote bond breaking leading to the production of smaller molecules [50], or give rise to more complete burning [82] where the reaction between HF and hydrocarbons can consume the free HF [83], aided by the increase of hydrocarbons at higher SOC. The increase of CF₄ with SOC is not explained [50]. The fluorine within the cell materials can also lead to POF₃, from reaction with water, which may be more toxic than HF [6]. Even though HF has a high toxicity rating, the large amounts of CO make this the main toxicity concern [84].

2.4. Electrolyte vapour

During TR, the electrolyte can exist in the off-gas as a vapour and presents additional flammability, irritability and toxicity hazards. The combined off-gas/electrolyte mixture is sometimes termed the “vapour cloud” [21] and can exist due to a lack of heat initiating combustion or due to a lack of oxygen limiting oxidation such that the solvent remains as a fume [45].

The solvents DMC, EMC and DEC (common in electrolytes) are highly volatile with low boiling points of 90°C to 129°C and high relative evaporation rates (between 1 and 3) [85]. PC and EC can be considered less volatile with higher boiling points of 242°C to 249°C and lower relative evaporation rates (<0.005). The vapour densities of solvents used in Li-ion cells are heavier than air and so will accumulate at the ground when the liquid solvent evaporates. However, the relative vapour density (ρ_{rel}) for a chemical compared to air (where $\rho_{rel} = 1$, $\rho_{rel} < 1$ and $\rho_{rel} > 1$ would correspond to a vapour that is neutral, buoyant and dense respectively) depends on the molecular weight of the vapour, temperature of the vapour, temperature of the air and the saturated concentration of the vapour in the air [86]. Therefore, the behaviour of a TR event, i.e. the temperature of failure (influencing vapour temperature), the rate and magnitude of vapour discharge and the rate of vapour dispersion, will influence how the relative vapour density changes with time. Further, the presence of solvents still in the liquid state, as aerosols, can lead to a cloud initially behaving as dense due to the relatively heavy droplets even if the emitted vapour mixed with air would normally (initially) be buoyant or dense [86]. As such, initially buoyant clouds can turn dense and vice-versa, adding further complexity to understanding and predicting the hazard of vapour cloud explosions in relation to LIBs.

For the common electrolyte carbonates, the volume of solvent required to evaporate to reach hazardous levels with mild transient effects (within a 1-meter distance from a vehicle) is 2.24 mL, 1.40 mL, 0.17 mL, 0.13 mL for DMC, EC, PC and DEC respectively. Lithium salts

lead to HF production, which is toxic and corrosive: 20 mL of 1 M LiPF_6 electrolyte can release enough HF in a 62 m^2 room to cause serious permanent health effects. Further, Diaz et al. [50] have shown that a single 18650 cell can lead to a contaminated volume of 100 m^3 to 400 m^3 based on the off-gas considering emitted solvents, CO, HF and other toxic components.

Although the presence and hazards of the electrolyte vapour are known, it is not frequently measured in gas analysis studies. However, overcharging of LFP cells shows that the off-gas can have a composition that is 60% electrolyte (where no fire or burning is noted in this test) [48]. Other work has detected, but not quantified, the existence of the solvents [7,29,87]. For various cell chemistries, the most abundant compounds were stated to be carbonates, which for LFP cells were DEC and EMC, and for NMC cells where DMC and EMC [44].

2.5. Composition in relation to other factors

The composition of off-gas is shown to vary with different stages of TR. At initial venting, CO_2 dominates, followed by hydrocarbons and a minimal amount of H_2 [88,89]. In some cases, electrolyte solvents are recorded and in a significant quantity, while in others, H_2 is not present [23,37,48]. During TR there is a marginal increase in H_2 , while during the deflagration there are significant amounts of CO and H_2 [88]. Also, electrolyte solvents are not recorded after TR [23]. However, aged cells show more CO and hydrocarbons at venting compared to fresh cells [89].

Other phenomena exist, including the retentively cool off-gas released early on in TR (compared to later stages) sinking to the floor (due to the vapour densities of solvents) [41]. Comparing “standard” versus “violent” TR, where violent is defined as maximum cell temperature over 250°C and more than 0.5 l of off-gas, there is more CO_2 and H_2 in violent cases [90]. LFP cells show no visible outburst of gas at SOC lower than 50% [82]. Similarly, for large LFP cells no fire or TR is seen at 0% SOC, but at 50% and above sustained jet fires are present [42].

The volume of gas produced by different abuse methods has been compared. Abuse by puncture (from projectile) shows less gas production than other failure modes (wedge-shaped penetration, impact and thermal) [2]. Different thermal abuse methods are shown to lead to similar results, except for reactor heating (where decomposition of electrolyte solvent occurs on the hot reactor walls after venting) leading to more gas being produced [28]. However, other work shows that a greater heating rate increases gas generation volume [90]. It is suggested that more gas is produced in air (allowing for combustion) than in a nitrogen atmosphere because the pyrolysis of organics produces a solid char [50]. Overcharge is shown to generate more gas (by more than 50%) than over-temperature or nail penetration [36]. Aged cells, with less remaining capacity, show less gas generation as more electrolyte is consumed in cells with less remaining capacity [91]. On overcharge, more gas is generated as a cell is increasingly overcharged [92], with the amount of CO_2 increasing at a greater rate than other gases (CO, H_2 and THC) [66,92,93]. However, Willstrand et al. [7] did not show any evidence to suggest that the abuse method affects the total gas volume produced or composition; but the abuse type does affect gas production rate, mass loss, and maximum temperature of the cell as much as its state of charge.

In open space, less gas is produced at a lower rate with the exception of CO_2 which has a higher rate [39]. Suggesting that more complete combustion occurs in open space, which conversely implies cells in a pack may not undergo complete combustion. In a nitrogen atmosphere hydrocarbons form a greater percentage of the off-gas than when abused in air [37]. For short circuit tests, more gas is produced in a nitrogen atmosphere and mainly consists of H_2 , CO_2 , CO and CH_4 , while in air the off-gas is mostly CO_2 and CO [94]. However, repeats of the same test can result in significant differences in results [95].

2.6. Abuse at larger scales

Studying the TR of LIBs at different scales, i.e. from cell to module and pack level, is important to understand how the hazards scale with battery size. However, only a few academic studies focus on the pack (i.e. EV) level concentrating on fire and toxicity behaviour.

EV fire safety has focused on similar gases to research on a cell level, namely CO_2 , CO, THC, NO_x , HF, HCl, HCN and heavy metals. The overall gas composition is similar between ICE and EVs. Over 95% of the combustion gas is CO_2 , a few percent is CO and less than 0.5% is THC, HCl, HF, NO_x and SO_x individually [1,3,4]. The most significant difference in EV and ICE vehicle toxicity is the quantity of HF. Lecocoq et al. [3] and Truchot et al. [1] both show that EVs generate double the amount of HF than ICE vehicles (1.5 kg versus 0.7 kg), while Willstrand et al. [4] show that EVs generate an order of magnitude greater HF than ICE vehicles (0.7 kg versus 0.01 kg). Further, the peak HF generation rate (3 g/s to 4 g/s) is similar in both EV and ICE vehicles, attributed to the rupture of the air-conditioning refrigerant [1,3]. However, the HF generation rate is greater for EVs than ICE vehicles (1 g/s versus 0 g/s to 0.5 g/s) at times later on in the vehicles burning — presumably due to the battery burning. HCl percentage composition is similar between vehicle types. In addition to gas production, battery fires lead to heavy metal deposits [2] that results in more heavy metals being produced in greater quantities by EV fires [5]. Due to the low toxic thresholds of these toxic substances, it is important to consider them for toxic evaluation, even though the total amounts produced are low [1].

Further to this discussion, it should be stated that these EV studies consider abuse leading to TR leading to fire, and not any situation where there is TR without fire. However, it is noted that there have been instances in the public where EVs have gone into TR without fire until the ignition of the off-gas [96]. Hence, there is a need to understand why and how large-scale battery TR failure leads straight to fires in some instances and to gas emissions and explosion hazards in others.

Moving up the battery scale (from cell to stack to module) an increasing volume fraction of CO , CO_2 , and H_2O is observed [8]. Further, the release of HF increases as module size increases, presumably due to simultaneous cell burning [75]. The production of gas (specific to capacity) is greater for a group of 10 cells with a similar capacity to one large cell, the gas production is 25% more at 100% SOC [97]. However, at 75% and 50% SOC there is a negligible difference between the group of cells and a single large cell. An array of the same capacity of a cell shows a tendency for higher CO_2 and CO fraction, with less H_2 . Further, in general, the greater number of cells and cell capacity leads to greater EV fire risk [6]. However, it is shown heat release rate (HRR) does not scale linearly with capacity, because not all cells burn at once. Hence, one should take caution scaling other hazards relating to batteries linearly with capacity.

2.7. Summary

In general, cell capacity and energy density are influencing factors of TR behaviour, affecting emitted venting gas volume, self-heating onset temperature and cell mass loss [27]. On a cell level, gas production typically ranges from 1 l/Ah to 3 l/Ah given all chemistries and in absolute terms increases with cell capacity. Under direct comparisons, LFP cells produce less gas than other chemistries in most studies. However, separate studies show that LFP may produce gas (l/Ah) on a similar scale to high-energy cells. Further, many studies have shown that gas production increases with SOC, but this is not true in all studies nor for all chemistries.

The off-gas mainly consists of CO_2 , CO, H_2 and hydrocarbons. However, the composition from the same as well as different cell chemistries and SOC varies considerably. The availability of oxygen (i.e. air vs inert atmosphere, or open vs closed space) affects the composition. Electrolytes can make up most of the off-gas but they are rarely

recorded. Given an abuse scenario, the composition is also different depending on how violent and hot the TR event occurs. Limited data on the overcharge of NMC cells shows more CO_2 is generated at higher SOC, while for LFP electrolyte vapours are the main component. The composition of the off-gas changes throughout TR, early on at low temperatures the initial venting can be mainly CO_2 , solvents or CH_4 (dependent on chemistry). This implies the hazard assessment should consider the different stages in TR and their relevance to fire/explosion risk. Many toxic components have been identified but HF and CO are the most recorded. It is shown a single 18650 cell can lead to a contaminated volume of 400m^3 and toxicity remains an issue at all SOC.

No standard testing procedure is used within the academic literature, unlike, for example, the use of accelerated rate calorimetry to determine thermal hazards of cells. Many authors use custom equipment with different gas monitoring setups, open/closed systems and systems of different enclosure sizes. This increases the difficulty in comparing results from different research groups and should be a consideration of the community to resolve.

The above raises the following questions which we aim to answer in our analysis. How does the chemistry, form, capacity and SOC impact the:

- specific gas production?
- toxicity (specifically CO and HF production)?
- gas composition?

3. Methods

To allow for a definitive analysis of the off-gas behaviour from LIB TR, a comprehensive literature search of the topic needs to be ensured. To do so, the literature search was conducted with the keywords “lithium-ion” or “Li-ion” or “LIB” and “cell” or “battery” or “EV” and “abuse” or “thermal runaway” or “fire” and “off-gas” or “venting” or “gas analysis”. From this, 60 papers [1–5,7,8,23–43,46,48–50,56,58–61,75–77,80,82,88–95,97–104] were deemed as appropriate and used to collate data on scale of battery (from cell to EV), form of cell, chemistry of cell, elemental ratio of cell chemistry, abuse type, cell capacity, test capacity or energy rating, SOC, nominal voltage (of cell), gas testing equipment, rate of $\text{CO}_2/\text{CO}/\text{HF}$, duration of test, total volume or mass of off-gas and quantities of individual components (g, mmol or litres) or percentage ratios. Conversion between units (g, mmol or litres) was done assuming standard temperature and pressure. The compiled data is supplied in full in the Supplementary Data. The literature data is then used to understand the influence of cell chemistry, cell type, battery scale, SOC and atmosphere on off-gas characteristics. The method of gas analysis and abuse type are recorded but not analysed for effect on LIB off-gas, and neither is the equipment setup (particularly closed versus open), but comments are made in the discussions.

4. Results and discussion — analysis of Li-ion off-gas emissions from literature

4.1. Bibliography summary

Given the literature collated, it can be seen from Fig. 2 that over the past ten years there has generally been an increasing number of papers published each year investigating the LIB off-gas, which to date leads to a total of 60 papers used in this analysis. Further subdivision of the number of papers that study specific form, cell type and chemistry can be found in the supplementary material. Within these papers most (53) focus on cell level studies while only 12 investigate larger scales including only 2 studying EV packs, see Fig. S1(a). Note that the number of papers to study specific categories can sum to more than the total (60) papers analysed as some papers may study more than one category, i.e. multiple cell forms or chemistries.

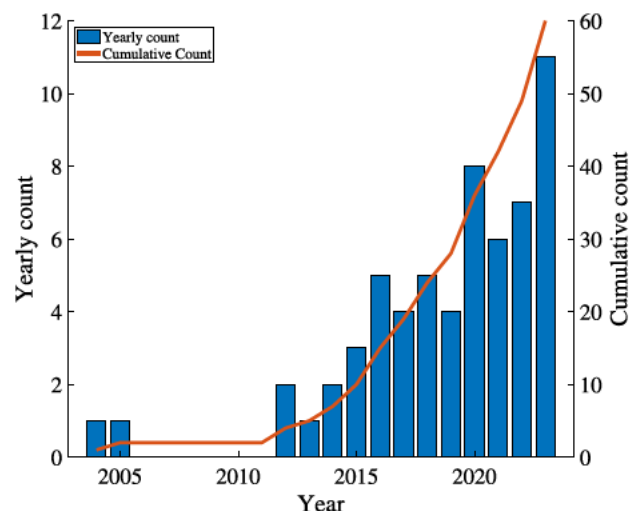


Fig. 2. Bibliography summary on a yearly basis, number of publications studying LIB TR off-gas.

At the cell level there are an approximately equal number of papers studying the off-gas of cylindrical, pouch and prismatic cells, see Fig. S1(b). Regarding chemistry, 31, 20 and 13 papers investigate NMC, LFP and LCO respectively, see Fig. S1(c). Further, NMC has mostly been studied in the past 4 years (2020–2023) with less studies on LFP in the last 3 years (2021–2023) than the 2 prior (2019–2020). Within the NMC chemistry a significant number of studies (over the entire date range) do not report the elemental ratios of nickel, manganese and cobalt, see Fig. S1(c). However, since 2020 there has been an increase in the number of studies reporting the elemental ratios, with the majority now being reported. In the last 2 years the most studied have been NMC811 and NMC622.

4.2. Total gas volume emitted

The first step in quantifying the hazard of LIB TR off-gas is determining the volume emitted. Fig. 3 presents the total amount of off-gas emitted at 100% SOC across a range of battery capacity scales, from cell to EV. This shows that as battery capacity increases so does the total amount of off-gas production. Further, it can be seen that below 0.02 kWh the results are mainly comprised of cylindrical cells with NCA, LFP and LCO chemistries. Above this, up to 1 kWh , the results are of prismatic and pouch cells, predominantly of NMC chemistry. For a more detailed view of these regions see Fig. S2 in the supplementary material. This highlights the lack of analysis of high capacity (10Ah to 100 Ah) LFP pouch/prismatic cells. Interestingly, the results below 0.02 kWh show more variation than above this value. This suggests that cylindrical cells may have a larger range of off-gas production than pouch or prismatic cells.

Further to the absolute volume of off-gas, Fig. 3(b) presents the specific volume of off-gas relative to the LIB capacity. This reveals that there is no correlation between capacity and the specific off-gas volume. To assess this further, and to determine the relative variability of each cell type, Fig. 4 presents a box plot of the specific off-gas generation for each cell type. For all box plots in this work the shaded box is the inter quartile range (IQR), the horizontal line within the shaded box is the median and the whiskers are the minimum and maximum values (excluding outliers). Individual data points are displayed as red scatter points while points more than 1.5IQR from the shaded box are deemed as outliers. From this it can be seen that more gas is typically produced by prismatic cells (where median values are 598 L/kWh - air, 573 L/kWh - inert) and pouch cells (502 L/kWh - air, 408 L/kWh - inert) than cylindrical cells (156 L/kWh - air, 157 L/kWh - inert). This

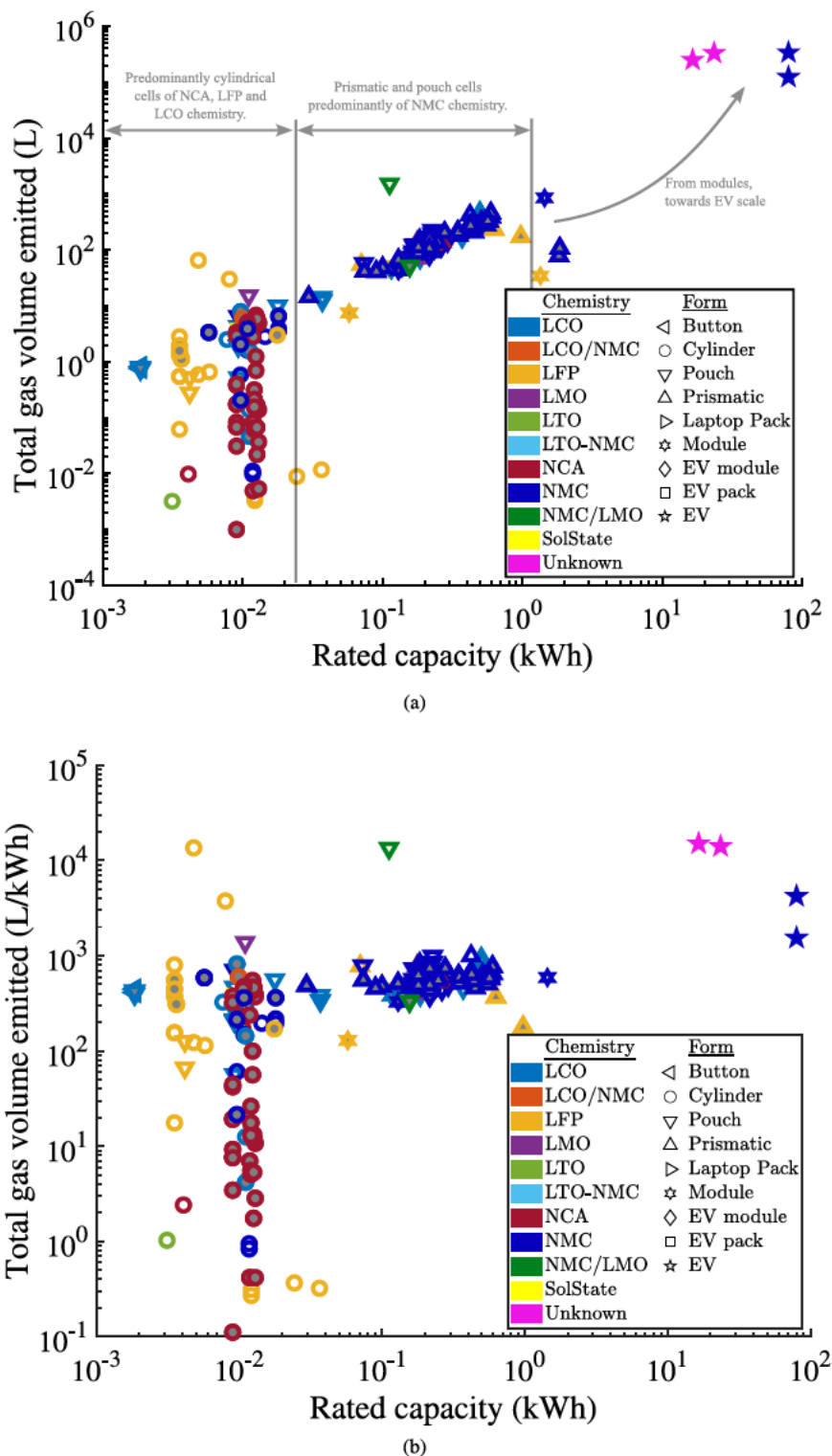


Fig. 3. Total amount of gas emitted from LIB failure given rated battery capacity for batteries at 100% SOC considering all chemistries and form factors (a) absolute values, (b) specific values. Unfilled points are from tests conducted in air, grey filled points are from tests conducted in an inert atmosphere, coloured filled points are of an unknown SOC but assume 100% SOC for this plot.

is similar to the behaviour shown by Rappsilber et al. [19]; however, here there are 93, 56 and 72 data points used within the cylindrical, pouch and prismatic cells types, respectively, compared to 12, 5 and 3 in Ref. [19]. Furthermore, here the data is split by atmosphere — air or inert. For cylindrical and prismatic cells the medians are similar for each atmosphere, while for pouch cells the inert atmosphere leads to less off-gas production. The variation in off-gas production is similar for

each cell type in air, while in an inert atmosphere pouch and prismatic cells have less variation than cylindrical.

Although Fig. 4 shows prismatic and pouch cells to produce more off-gas than cylindrical cells, Fig. 3 shows that for the collated data these cell types are predominantly NMC. For clarity, Table S1 shows the number of data points of each chemistry for each cell type in Fig. 4. From this it can be seen that the prismatic, pouch and cylindrical

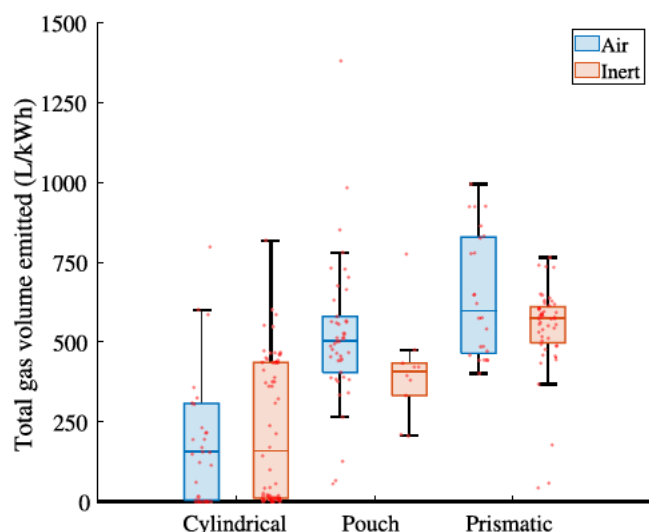


Fig. 4. Considering cell results only, the total amount of off-gas emitted from LIB failure at 100% SOC grouped by cell type. (The number of values in each category are Form [Air, Inert]: Cylindrical [31, 62], Pouch [46,10] and Prismatic [24, 48].).

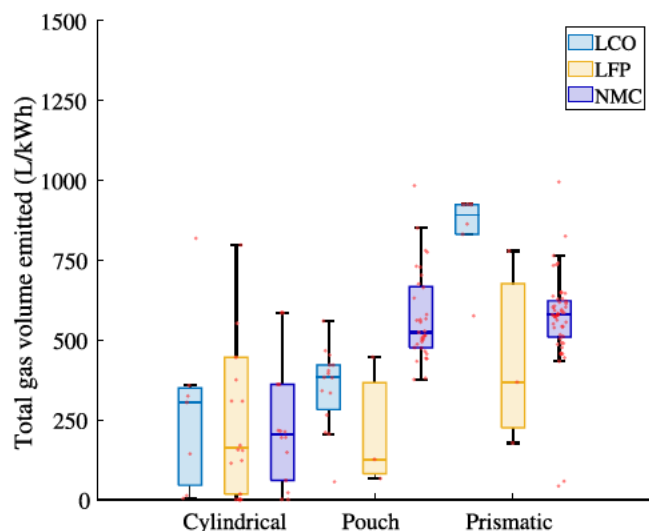


Fig. 5. Considering cell results for LCO, LFP and NMC chemistries, the total amount of off-gas emitted from LIB failure at 100% SOC grouped by cell type (both in air and an inert atmosphere). (The number of values in each category are Form [LCO, LFP, NMC]: Cylindrical [7, 18, 14], Pouch [15, 3, 33] and Prismatic [6, 3, 41].).

group data points are 68%, 59% and 15% NMC chemistry. As the literature has suggested (see Section 2.1), NMC produces more off-gas than other chemistries. Hence, it is necessary to analyse the off-gas volume regarding cell type and chemistry to determine if prismatic cells produce more off-gas due to their form or due to the data being mostly NMC. This is presented in Fig. 5 for LCO, LFP and NMC chemistries as the other chemistries lack data to compare across all cell types (see Table S1). Note that Fig. S3 shows outlines not present in Fig. 5 due to the scale used to improve clarity.

Fig. 5 shows that LCO prismatic and pouch cells have a greater generation of off-gas than cylindrical cells (where median values are 893L/kWh, 383L/kWh, 303L/kWh - prismatic, pouch, cylindrical respectively). This is similar for the NMC chemistry, however pouch and prismatic cells produce similar amounts of off-gas (579L/kWh, 524L/kWh, 203L/kWh - prismatic, pouch, cylindrical respectively). For the LFP chemistry, the increase in off-gas generation of prismatic

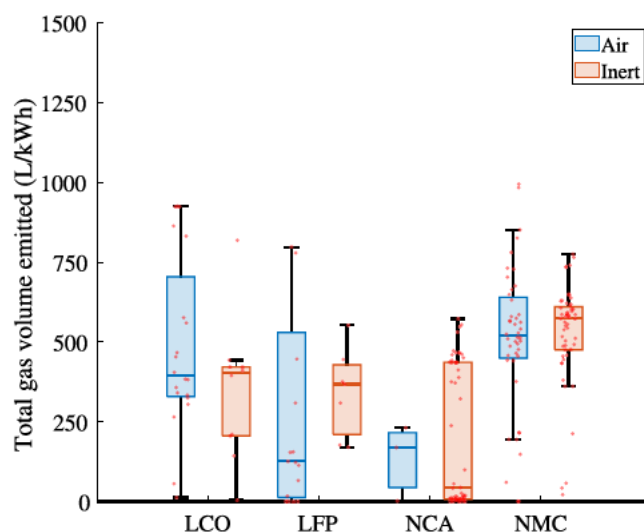


Fig. 6. Considering cell results only, the total amount of off-gas emitted from LIB failure at 100% SOC grouped by cell chemistry. (The number of values in each category are Form [Air, Inert]: LCO [20, 10], LFP [17, 7], NCA [3, 51] and NMC [44, 52].).

cells relative to cylindrical (368L/kWh - prismatic, 163L/kWh - cylindrical) is less than the LCO and NMC chemistries. Also LFP pouch cells produce an off-gas volume (126L/kWh) similar to cylindrical cells. Overall, from Figs. 5 and 6, it is likely that more off-gas is generated by prismatic, than pouch and finally cylindrical cells.

To corroborate the literature statement that NMC produces more off-gas than other chemistries (see Section 2.1), Fig. 6 presents the specific volume of off-gas production for LCO, LFP, NCA and NMC chemistries. Note that Fig. S4 shows outliers not present in Fig. 6 due to the scale used to improve clarity. Other chemistries have been recorded, see the compiled CSV data table in the supplementary materials, but are not presented due to lack of sufficient data. Fig. 6 shows, from the median of the data, that the NMC chemistry does produce more off-gas than other chemistries (LCO: 394 L/kWh, 403 L/kWh; LFP 126 L/kWh, 368 L/kWh; NCA 169 L/kWh, 44 L/kWh; NMC 519 L/kWh, 573 L/kWh - air and inert respectively). In an inert atmosphere it is likely that NMC cells produce more gas than other chemistries. However, in air there is a chance that NMC cells will produce the same amount of off-gas as LCO and LFP. Comparison to NCA chemistry in air is omitted due to the being only 3 data points for this set. Furthermore, given the available data, there is no definitive difference in gas volume production between the two atmospheres.

While Fig. 6 presents data at 100% SOC, Fig. 7 presents the off-gas volume for LCO, LFP and NMC chemistries over various SOC. This data is mostly of cells as only 2 data point in the 100% SOC group are for any other battery scale. Fig. S5 presents similar results for the additional chemistries LMO and NCA, but is not included in the main analysis due to the lack of data at lower SOC. From Fig. 7 it can be seen that there is a strong tendency for increased gas production at higher SOC for LCO and NMC chemistries, while for LFP the increase is less steep. Also, both LFP and LCO show that there is a possibility for more gas to be produced at 0% SOC than at 25% SOC. The median values of off-gas production are similar between LCO and NMC chemistries across SOC. While for LFP the median values are similar to LCO and NMC at SOC 25% and lower, but at higher SOC the LFP chemistry has a lower median. However, the LFP chemistry shows a large variation in off-gas volume at each SOC. As such, it is possible for them to produce as much off-gas as LCO and NMC at all SOC. Further, due to the variation in the NMC chemistry it is possible that they themselves could produce similar or more off-gas at lower SOC than at higher SOC. It should be noted that there are significantly less data points at SOC less than 100%, hence this should be a focus of the academic community.

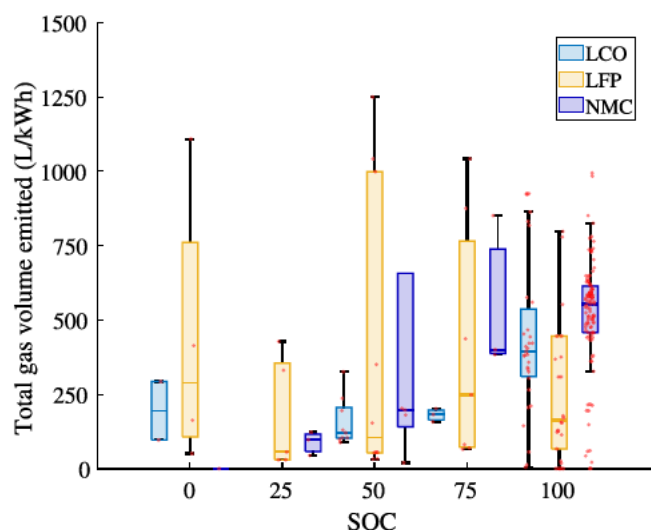


Fig. 7. The specific total amount of off-gas emitted from LIB failure at various SOC for air and inert atmosphere. (The number of values in each category are SOC % [LCO, LFP, NMC]: 0% [2, 4, 1], 25% [0, 5, 3], 50% [9, 10, 5], 75% [3, 7, 3] and 100% [31, 26, 98]).

For a clearer view of the median trends see Fig S6 which also presenting intermediate SOC as well as overcharge data. However, these additional SOC points typically only consist of one data entry, but from Fig S6 it can be seen there is a typical trend for LCO, LFP and NMC chemistries on overcharge to produce more off-gas than at 100% SOC. For the NCA chemistry there is negligible increase with overcharge.

It is of interest to determine if the severity of TR and off-gas production volume has changed over time with cell chemistry developments, i.e. with new additives or electrolyte compositions. Analysing the compiled data shows that on the whole there is no change in total specific off-gas production with time, see Fig. S7. LCO and NCA chemistries show an increase in the latter few years, but considering cell capacity, form, atmosphere and abuse type, no variables are determined to be influential. Hence, unknown cell material differences, differences in experimental setup or natural variation may be the cause.

4.3. Toxicity - HF and CO emissions

The total emission of HF and maximum emission rate is presented in Fig. 8, both of which increase with battery capacity. From this, it can be seen that the smallest cells release 0.01 g to 1 g of HF while larger cells release 0.1 g to 100 g and packs/EVs release kilograms of HF. The majority of cells studied are for LFP chemistry while NMC cells and EVs are studied to a lesser extent. The emission rates are less reported range from 10^{-4} g/s to 1 g/s from cell to EV scale (excluding HF release from air-conditioning).

The average HF emissions for LFP and NMC tests over various SOC is presented in Fig. 9 (note most of this data is related to cells as there is only one non-cell data point in the 100% SOC category). From this it is clear that the LFP chemistry typically releases over ten times more HF than the NMC chemistry normalised against battery capacity (50 g/kWh to 100 g/kWh for LFP vs <10 g/kWh for NMC). LFP cells show a slight downward trend in emissions as SOC increases (100 g/kWh at 0% SOC to 50 g/kWh at 100% SOC), due to higher TR temperatures reached (see Section 2.3), but NMC do not. However, due to the large variation in results there is little certainty that higher SOC will have lower emissions than lower SOC.

Note that there is limited data for LMO, NCA and LCO cells, although at 100% SOC LMO, NCA and NMC/LMO cells emit approximately 60 g/kWh, while LCO emit 15 g/kWh. LMO cells show a negative trend with SOC, while LCO do not show any trend.

Fig. 10 shows the maximum HF emission rate given SOC normalised against battery capacity. This shows that NMC cells have an increased rate with SOC (0.1 g/(s kWh) to 0.7 g/(s kWh)) while LFP do not. It also shows NMC cells have a much greater rate at larger SOC (0.7 g/(s kWh) for NMC vs 0.1 g/(s kWh) for LFP). This is attributed to the greater severity and strong SOC dependence of NMC failure.

The total emission of CO and maximum emission rate is presented in Fig. 11, showing both increase with battery capacity (most of the data is from cells, only four and two points at 25% and 100% SOC are not cells for LFP). Comparing Fig. 8 and Fig. 11 it can be seen that there is approximately an order of magnitude more CO emitted than HF given battery capacity, with rates also larger. A more detailed analysis of the CO emissions with SOC shows that at 100% SOC batteries with an NMC chemistry emit 10 times more CO specific to battery capacity than the LFP chemistry (172 g/kWh for NMC vs 19 g/kWh for LFP). Further, the CO emission of NMC batteries is two orders of magnitude greater than HF emissions, while for LFP the emissions are on the same order of magnitude. NMC batteries show a tendency to release more CO with increased SOC (10 g/kWh to 172 g/kWh for 25% to 100% SOC), while LFP batteries show a slight overall downward trend but there are unexpectedly low values at 25% and 75% SOC. This discussion has so far considered air and inert atmosphere data together. Fig S8 compares the results for LFP and NMC chemistries under both atmospheres and shows no statistical difference. As with the total volume (see Section 4.2), assuming composition is independent of atmosphere, the increase in total CO amount in NMC cells is both due to the chemistry and cell form leading to greater gas production.

4.4. Gas composition

A comparison of gas composition for each major chemistry over various SOC would be ideal. However, due to limited data only meaningful comparisons can be made for LFP and NMC at 100% and 50% SOC in air and 100% in an inert atmosphere. Results from tests in an inert atmosphere are considered as they simulate a scenario where oxygen is limited, such as in a sealed pack. Therefore, the gas composition before combustion can be determined, allowing for a truer assessment of off-gas hazards. Results for LFP and NMC in air at 100% SOC are presented in Fig. 13, other LFP and NMC results along with limited data for LCO and NCA is presented in Fig. S9 and Fig. S10. In these figures, *electrolytes* (Elect.) contains DMC, DEC, EMC and EC; *organic compounds* (org. comp.) contains ethanol, methanol, DME and methyl formate; *F containing compounds* (F comp.) is anything containing fluorine (including HF); and *other* is anything not listed in the graph or these categories. Note, due to a lack of compositional analysis tests in the literature concerning HF in LIB off-gas, only minimal data is available on HF as a fraction of the total gas. As such, there are fewer data points related to HF in the compositional analysis here than in the analysis of absolute HF production (Section 4.3).

For LFP at 100% in air (see Fig. 13), on average, the major component of the off-gas is H_2 (36%), then CO_2 (25%), CO (12%) and THC (11%). For NMC in similar conditions CO_2 (36%) is the major component, then CO (25%), H_2 (20%) and THC (12%). The greater CO and CO_2 generation by NMC is attributed to the tendency of NMC to lead to fires in air. Note, in previous works, none of the studies in air were set-up to record the emissions of electrolyte. Furthermore, there is no correlation of composition (percentage of species CO, CO_2 , H_2 or THC) with battery capacity, see Fig. S11.

Consider 50% SOC in air for LFP and NMC (see Fig. S9(a)) there is no meaningful difference in results when compared with 100% SOC (Fig. 13). Comparing atmospheres (see Fig. S9(b)), we see that there is a greater variation in CO_2 percentage for LFP and greater H_2 generation than in Fig. 13. These results also show that electrolyte vapour is emitted to a small percentage, as well as O_2 to a lesser extent. From Fig. 13 it can be seen that H_2 is a significant flammability/explosion concern as it is typically present in a greater proportion than the THC content.

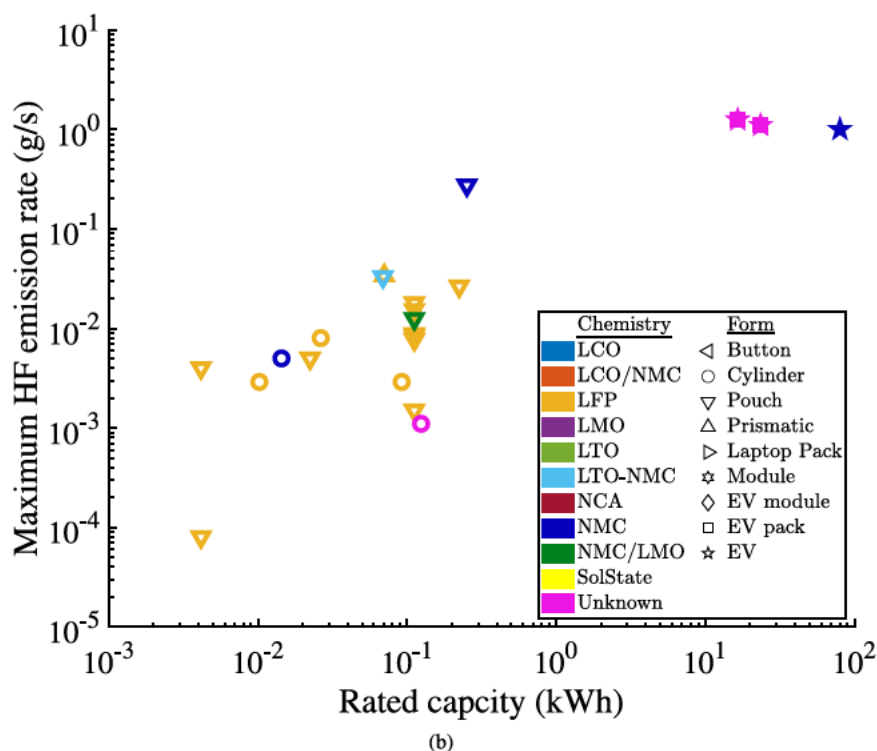
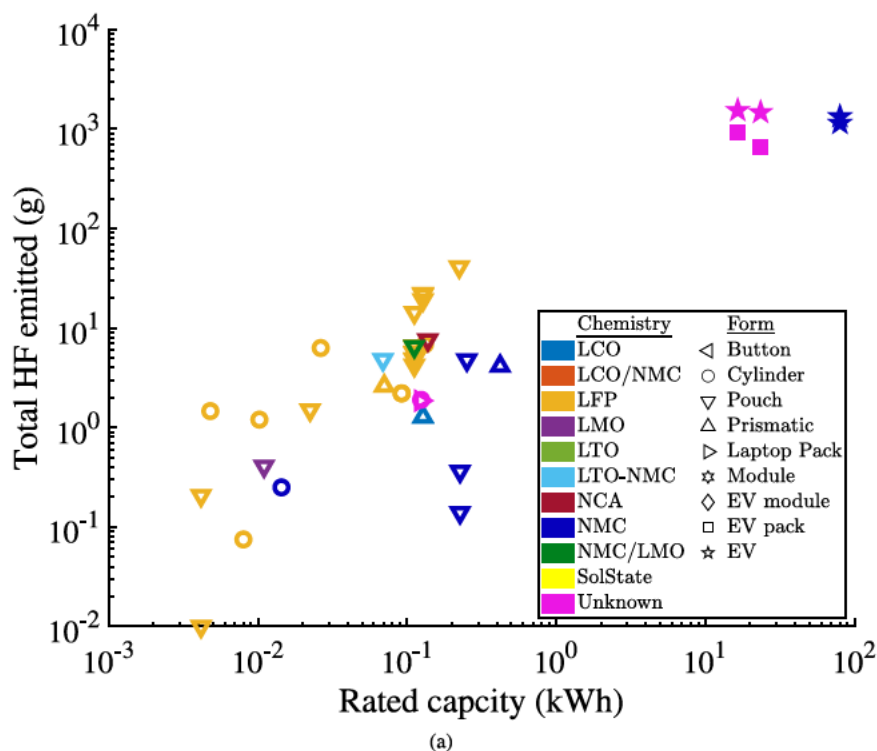


Fig. 8. HF emissions from LIB failure given rated battery capacity for batteries at 100% SOC considering all chemistries and form factors (a) total mass emitted, (b) maximum rate of emissions. Unfilled points are from tests conducted in air, grey filled points are from tests conducted in an inert atmosphere, coloured filled points are of an unknown SOC but assume 100% SOC for this plot.

It is also present in greater quantities for the LFP chemistry. The risk of H_2 content and also the role of cell chemistry on flammability is further discussed in Section 4.5.

As the emissions of gas species are interdependent, correlation plots are presented to understand the relation between the production of

each gas, see Fig. S12 and Fig. S13. However, there is no consistent behaviour between chemistries or atmospheric conditions. This is attributed to the unpredictability of combustion occurrence, and the effect the test set-up has on the availability of oxygen to promote combustion i.e. if the system is open or closed and the working volume.

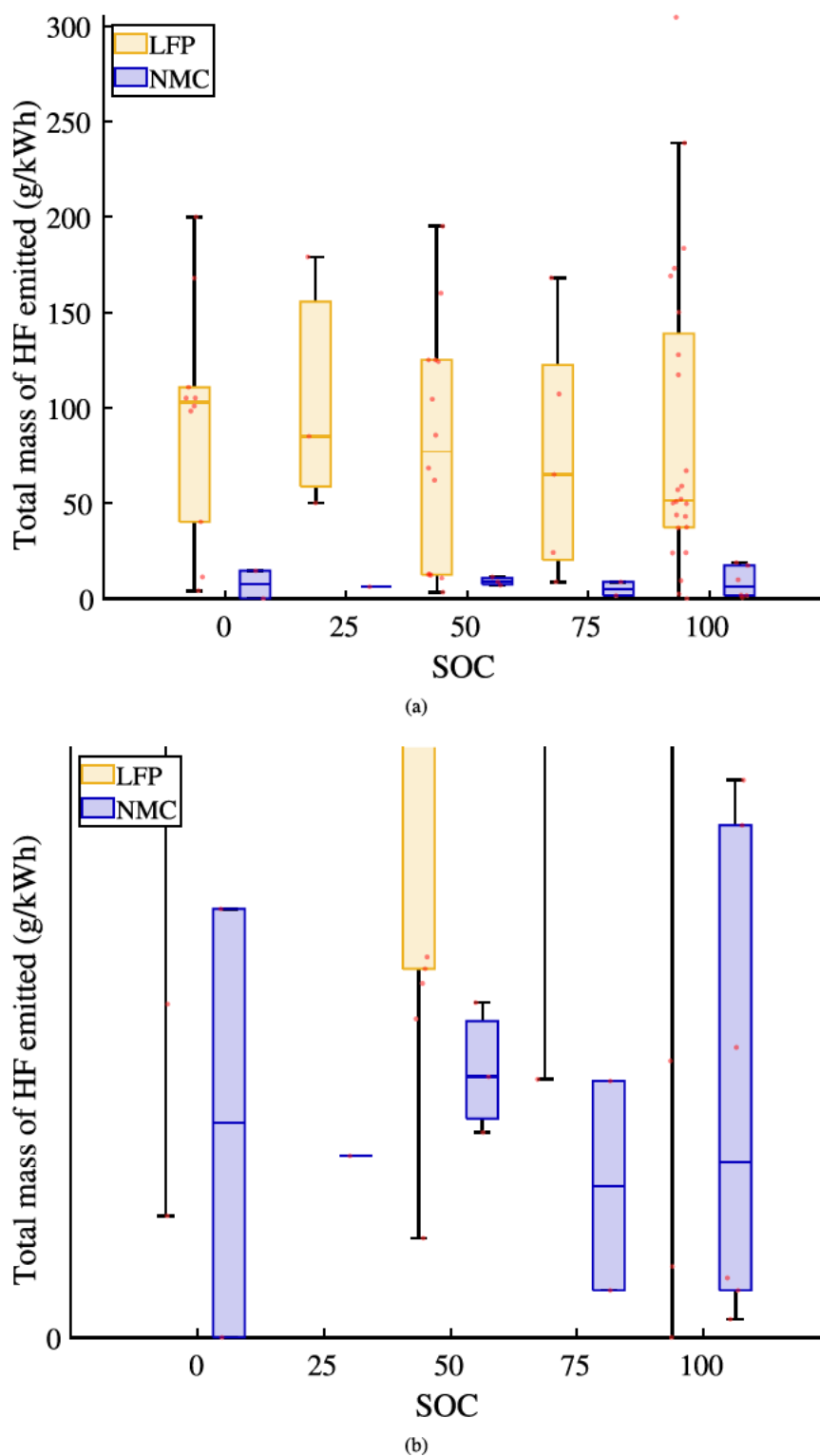


Fig. 9. (a) The specific total mass of HF emitted from LIB failure at various SOC in air, (b) enlargement of NMC data. (The number of values in each category are SOC % [LFP, NMC]: 0% [10, 2], 25% [3, 1], 50% [14, 3], 75% [5, 2] and 100% [24, 6].)

4.5. Discussion and recommendations

From Section 4.1 it can be noted that there is a lack of studies on the off-gassing of modules and packs. Addressing this in the academic community is important to understand how the behaviour of off-gassing and its composition changes with scale so that the change

in hazards with scale can be properly assessed. However, the literature does study cell form evenly between the three main types (cylindrical, pouch and prismatic) and focuses on the two chemistries mainly used in automotive sector NMC and LFP. As such the literature is useful for the industrial community. Although the elemental ratio of NMC cells is reported more in recent years, a third of papers still do not.

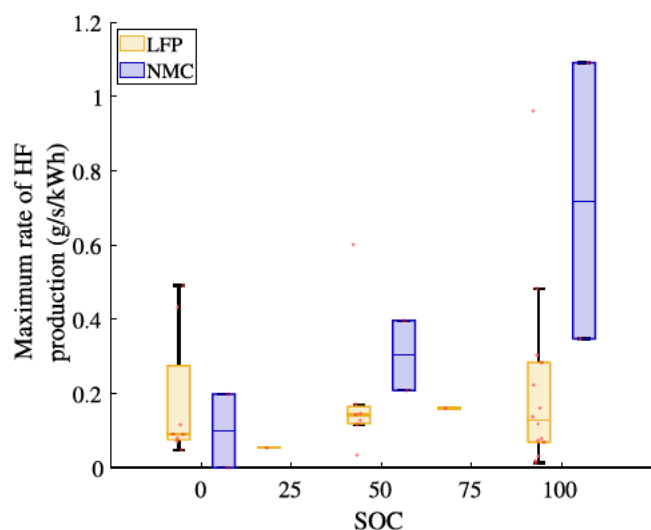


Fig. 10. The specific maximum rate of HF production from LIB failure at various SOC in air. (The number of values in each category are SOC % [LFP, NMC]: 0% [8, 2], 25% [1, 0], 50% [7, 2], 75% [1, 0] and 100% [14, 2].).

Hence, the academic community should encourage reporting of this to better understand the role of chemical composition of NMC on off-gas behaviour.

From the analysis in Section 4.2 it is clear that there is a strong positive correlation between off-gas volume and battery capacity across the entire range from Wh to kWh. However, the amount of gas produced specific to battery capacity is independent of battery capacity. NMC batteries do tend to produce more gas than other chemistries when considering all battery types. In general prismatic cells tend to produce more off-gas than pouch followed by cylindrical cells, even when considering chemistry. This could be caused by the large form factor having lower heat loss per unit volume leading to higher temperatures and increased decomposition. When considering separate cell forms, cylindrical LFP and NMC cells have similar medians and ranges, while LCO have a great median. For prismatic cell NMC produce less off-gas than LCO. However, there is limited data for cylindrical and prismatic LCO cells and also pouch and prismatic LFP cells which hinders comparison. Cylindrical cells and LFP chemistry show a slight tendency for greater variation in gas volume production.

As LFP are widely used in the automotive sector, more analysis of LFP pouch and prismatic cells at the scale of 10–100 Ah is crucial for hazard assessment between NMC and LFP batteries. Also, as the literature (see Section 2.6) has shown, it is difficult to scale cell TR behaviour to stack/module/pack TR behaviour. As such, comparisons of LFP and NMC batteries of a given capacity constructed of different cell forms need to be rigorously conducted to determine if there is any real difference in TR hazards, including gas hazards.

The amount of gas produced in air is not consistently more or less than that generated in an inert atmosphere, so the effect of the atmosphere on off-gas volume production is considered negligible. As is commonly suggested in the literature (Section 2.1), larger SOC typically leads to greater off-gas volumes while NMC LIBs generate more gas than LFP over all SOC. The variation within the data means that given a chemistry, a higher SOC may not lead to more gas, while LFP LIBs may produce as much as NMC LIBs. However, at SOC other than 100% and at overcharge there is significantly less data, typically less than 10 data points and only 1 for stages of overcharge (for each chemistry). Overcharge is a typical area of concern as it can occur undetected within battery operation. Hence it is critical to risk assessments to understand how the off-gas generation evolves with overcharge and should be a focus of the research community.

Table 3

Theoretical contaminated volume calculated from median HF and CO emissions for a 0.01 kWh battery.

Emissions at 100% SOC (g/kWh)		
Component i	NMC	LFP
CO	172	19
HF	6	52
Emissions at 0% SOC (g/kWh)		
Component i	NMC	LFP
CO	10 ^a	34
HF	7	102
Contaminated Volume (m ³)		
SOC	NMC	LFP
100%	115	355
0%	51	695

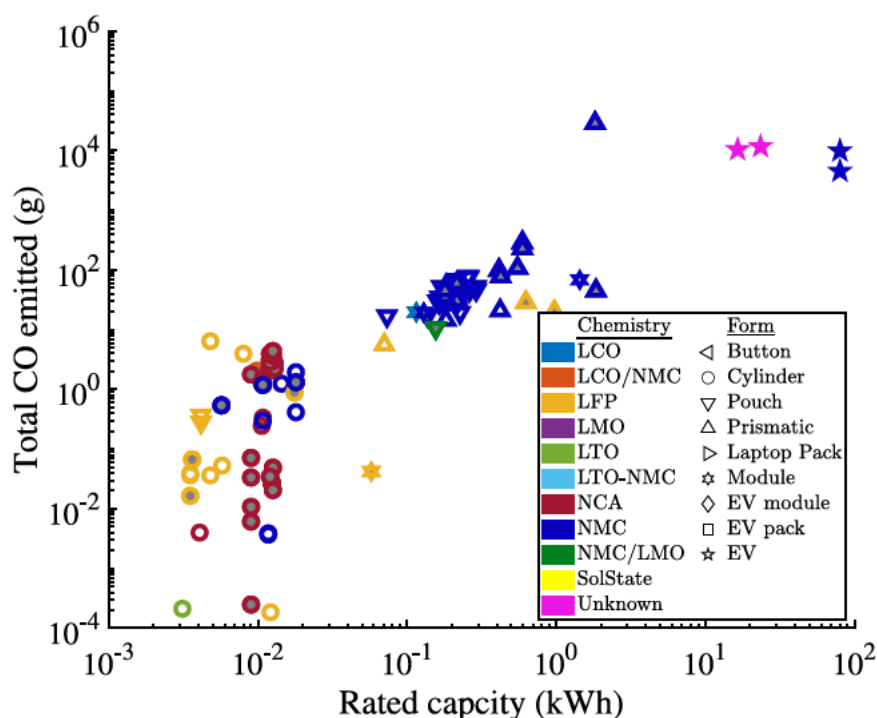
^a Estimated from data at 25% SOC.

The toxic component analysis in Section 4.3 shows HF release from LFP cells is greater than from NMC over various SOC, with a median in the range of 60 g/kWh to 100 g/kWh for LFP versus less than 10 g/kWh for NMC. However, the rate of HF release is greater for NMC cells than LFP cells at higher SOC, due to the increased severity of TR of NMC versus LFP. LFP show the trend of less HF production at higher SOC due to increased temperature of TR, while NMC show no trend due to relatively high TR temperatures at all SOC. The release of CO is greater for NMC cells compared to LFP cells. Over 0% to 100% SOC the release is 11 g/kWh to 172 g/kWh for NMC compared to 34 g/kWh to 19 g/kWh for LFP. Although several studies have investigated the absolute HF production, few have done so as a fraction of the overall composition of the off-gas (see Section 4.4). This should be a consideration for the academic community to determine if there are any correlations of the fraction of HF to other off-gas species to complement the correlations of SOC and cell chemistry analysed in this work.

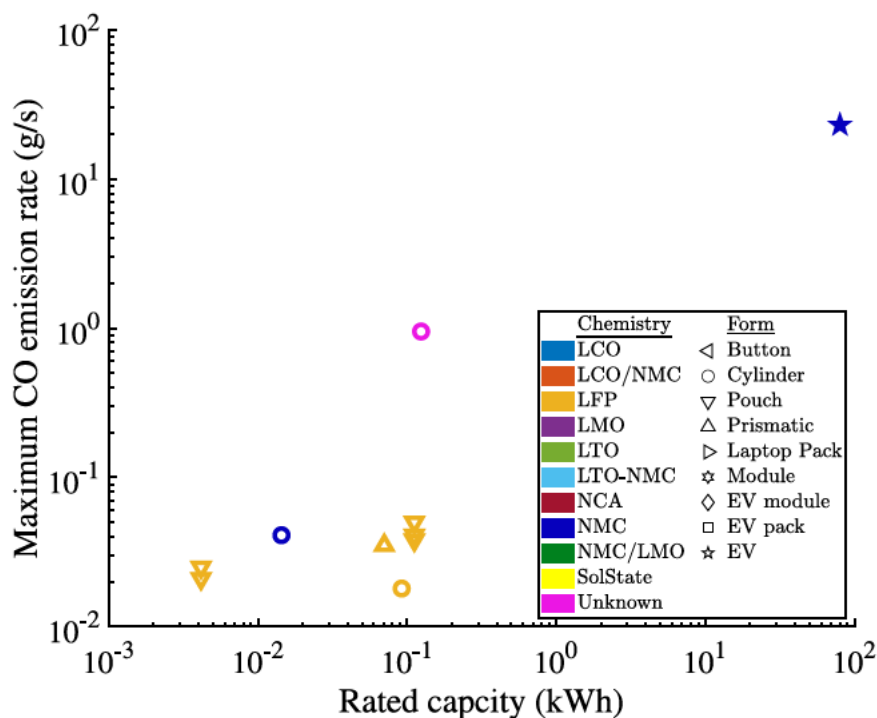
As the HF and CO production vary greatly with chemistry and SOC, the overall toxicity hazard has to be assessed considering the total amounts of all toxic components. This can be done by calculating the theoretical contaminated volume (m³) according to Eq. (1) [50]. Where m is the mass (mg) of the toxic component i which has an exposure limit value EL (mg/m³). The short term exposure limits (in the UK) for HF and CO are 1.5 mg/m³ and 23 mg/m³, respectively [54]. With this and the median HF and CO emissions for NMC and LFP at 0% and 100% SOC (see Figs. 9 and 12), Table 3 presents the calculated contaminated volume assuming a 0.01 kWh battery (this is the scale of a single cylindrical cell). Note median values are used as there are limited sources that reports both CO and HF emissions together for all four cases.

$$V_{\text{contaminated}} = \sum_{i=1}^n \frac{m_i}{EL_i} \quad (1)$$

From Table 3 it is shown that LFP batteries are significantly more toxic than NMC batteries when considering both major toxic components, especially at 0% SOC. This is due to the very low exposure limit of HF and the higher emissions of HF by LFP batteries. However, LFP batteries are nearly half as hazardous at 100% SOC than at 0% SOC as both HF and CO reduce at higher SOC. In comparison, NMC batteries are over twice as toxic at 100% SOC than 0% due to the large increase in CO emissions. Further, the lowest emissions from a single cell (approximately 0.01 kWh) are enough to full a single car garage or 20 ft shipping container. But at a large cell or module scale (where simultaneous burning is possible) contaminated volumes would be 10 and 100 times greater respectively. From this, LFP batteries can be said to be more toxic than NMC (in absolute terms) even though they produce on average less off-gas overall. However, the suffocation (from CO₂ emissions) and flammability hazards have to also be considered, discussed below.



(a)



(b)

Fig. 11. CO emissions from LIB failure given rated battery capacity for batteries at 100% SOC considering all chemistries and form factors (a) total mass emitted, (b) maximum rate of emissions. Unfilled points are from tests conducted in air, grey filled points are from tests conducted in an inert atmosphere, coloured filled points are of an unknown SOC but assume 100% SOC for this plot.

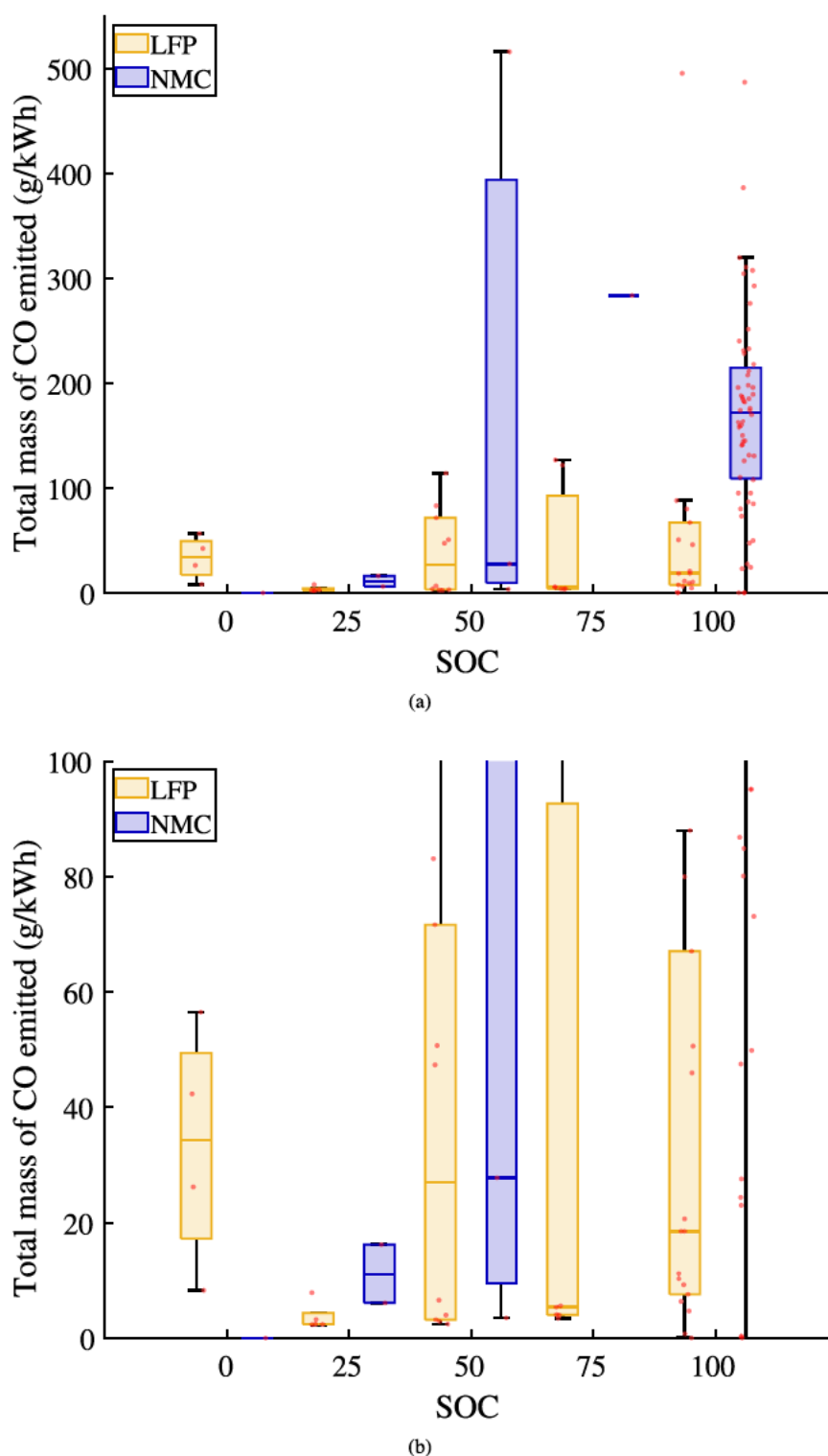


Fig. 12. (a) The specific total mass of CO emitted from LIB failure at various SOC in air, (b) enlargement of LFP data. (The number of values in each category are SOC % [LFP, NMC]: 0% [4, 1], 25% [5, 2], 50% [10, 3], 75% [7, 1] and 100% [18, 60].)

From the gas composition results of Section 4.4 it is shown there is a lack of data quantifying electrolyte solvent emissions, especially in air, as experiments are not set-up to detect it. This should be addressed as the electrolyte vapour is considered to be a significant flammability hazard. Further, tests set up should be assessed to ensure it does not affect results, i.e. open/closed system limiting combustion. From analysis of the off-gas composition, it is found that the LFP chemistry releases

more H_2 than NMC on average, while more CO is emitted by NMC with similar hydrocarbon contents in both. However, it is difficult to assess the overall flammability hazard from this data. Hence, to assess the flammability hazard of each chemistry the lower flammability limit (LFL) of the off-gas mixture is calculated. This is done according to the methods in Ref. [105] accounting for the dilution of the off-gas by the CO_2 generation. The LFL of each component (CO, H_2 and hydrocarbons

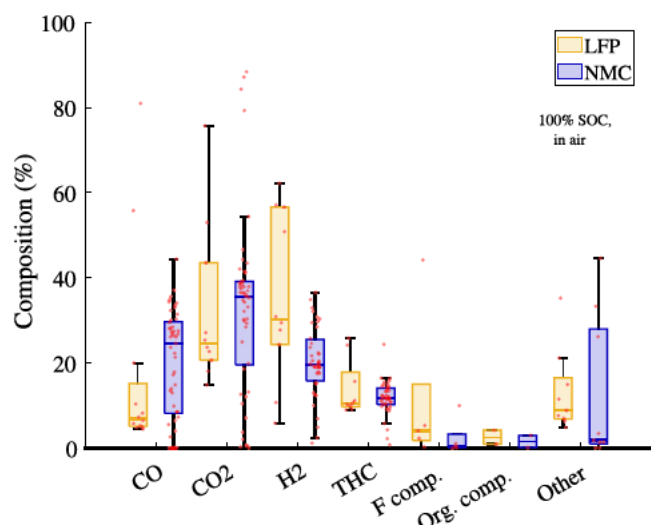


Fig. 13. Gas composition for LFP and NMC abused LIBs at 100% SOC in air. (The number of values in each category are Gas [LFP, NMC]: CO [12, 49], CO₂ [10, 49], H₂ [10, 39], THC [9, 39], F containing compounds [5, 5], organic compounds [2, 2] and other [9, 9].

present) are from data tables in Ref. [106]. Further detail of the LFL calculation methodology (including a table of LFL values for individual species) is given in the supplementary material.

Fig. 14 presents the calculated LFL. It can be seen that the LFP chemistry has a lower LFL (6.2%) compared to NMC (7.9% to 9.2%). This is attributed to LFP having a lower fraction of CO (which has a large LFL) and CO₂ (which dilutes the mixture), whilst having higher H₂ fractions, compared to NMC. The large range of LFL for NMC in air is attributed to the large range in CO and CO₂ production related to the variability in combustion of the off-gas leading to in/complete combustion products. Given the median LFL (Fig. 14) and gas volumes produced (Fig. 6) in an inert atmosphere then LFP batteries would breach their LFL in a volume 18% smaller than NMC cells (calculated assuming the gas is only comprised of CO₂, CO, H₂ and hydrocarbons). Hence, LFP batteries present a greater flammability hazard even though they (specifically cells) show less occurrence of flames.

As noted previously, CO₂ is not considered as a toxin, but in high quantities it presents a suffocation hazard. However, Fig. 15 shows that the contaminated volume from a 0.01 kWh battery (where the short term exposure limit of CO₂ is 27 400 mg/m³) is minimal. Even at the 1 kWh scale the hazardous volume would still be less than 10 m³, so the suffocation hazard is low relative to the toxicity hazard.

As noted above, LFP cells show a tendency for a lower LFL but this does not account for the presence of solvents (due to lack of available data). However, as shown by Fernandes et al. [48] up to 60% of the off-gas emission can be made up of electrolyte solvents when no venting fire occurs. This will also be true for other chemistries, however, the important factor here is that due to the lower TR temperature of LFP cells it is more likely that the vent gas is emitted without combustion. This results in a greater likelihood of vapour cloud emission and accumulation, leading to an increased risk of vapour cloud explosions for LFP batteries compared to NMC. This needs to be stressed in safety and risk assessments given the general belief that LFP cells are “safe” or “the safest” in public media [e.g.107–110]. This “safest” chemistry belief is based on typical abuse tests (overheat, penetration, etc [111]) due to LFP having lower maximum temperatures and heat generation or the absence of visible sparks and flames [112–114]. However, there are many instances of LFP-based EVs under TR and emitting vapour clouds [6,10], especially in the Chinese market where LFP dominates [115]. Additional to this is the emission of toxic substances that also present a further hazard. As such, it is unwise to categorise

the safety of a battery system based on the abuse test of cells that do not account for the explosion of the off-gas (and its toxicity) or the influence of the battery system design on failure behaviour. Therefore, there should be a focus within the battery community to provide a holistic assessment of battery safety considering stability and thermal, fire/explosion and toxicity hazards.

5. Conclusion

The off-gas from Li-ion battery TR is known to be flammable and toxic making it a serious safety concern of LIB utilisation in the rare event of catastrophic failure. As such, the off-gas generation has been widely investigated but with some contradictory findings between studies. However, no work has comprehensively analysed the available literature data to determine how the chemistry, SOC, scale/capacity and form affect gas volume production, toxicity and flammability. Hence, in this work we conducted a detailed meta-analysis of 60 papers to investigate the most influential parameters and the probable off-gas characteristics to determine what kind of battery would be least hazardous.

From the analysis, it is found that LFP and NMC batteries are the most studied, aligning with industries tendency towards these chemistries. The amount of gas produced scales linearly with capacity, however the specific gas production (L/kWh) shows no trend with capacity. NMC LIBs produce more off-gas than other chemistries (LCO 394 L/kWh, LFP 126 L/kWh, NCA 169 L/kWh, NMC 519 L/kWh), while prismatic cells also tend to generate larger off-gas volumes (cylindrical 156 L/kWh, pouch 502 L/kWh, prismatic 598 L/kWh). Also, a larger SOC does lead to greater average specific off-gas volumes specifically for NMC and LCO batteries, however there is significant variation in results.

While NMC batteries release more gas than LFP, LFP batteries are significantly more toxic than NMC ones in absolute terms. Toxicity varies with SOC, for NMC batteries the contaminated volume doubles from 0% to 100% SOC while for LFP in halves. The composition of off-gas on average is very similar between NMC and LFP cells, but LFP batteries have greater H₂ content while NMC batteries have a greater CO content. To assess the fire hazard the LFL limit of the off-gases is compared. The LFL for LFP and NMC are 6.2% and 7.9% (in an inert atmosphere) respectively. Given the LFL and the median off-gas volumes produced, LFP cells breach the LFL in a volume 18% smaller than NMC batteries. Hence LFP presents a greater flammability hazard even though they show less occurrence of flames in cell TR tests.

Also, from this work it is found that significant improvements in research can be made. To advance the understanding of off-gas further, for improved battery safety, it is recommended that:

1. the electrode elemental ratio and electrolyte composition of cells are reported to better compare NMC results and understand the affect of electrolytes;
2. LFP pouch and prismatic cells at the scale of 10–100 Ah to be studied to address the gap in data at this scale, so that like-for-like comparisons between high energy LFP and NMC cells can be made;
3. tests at module and pack level should be conducted to understand how the hazards scale with battery size;
4. the off-gas generation is studied at other SOC other than 100% and especially at overcharge to accurately determine how toxicity and flammability hazards vary; and
5. the experimental set up is designed to ensure the detection/quantification of common electrolyte solvents to determine when and how much is emitted as a vapour and hence determine the additional flammability hazards.

Finally, this work provides a critical resource to the battery community that can be used for the risk assessment of LIB TR fire, explosion and toxicity hazards. This is aided by supplying the compiled literature data in a raw format that is readable and editable to allow independent and ongoing analysis by interested/relevant parties.

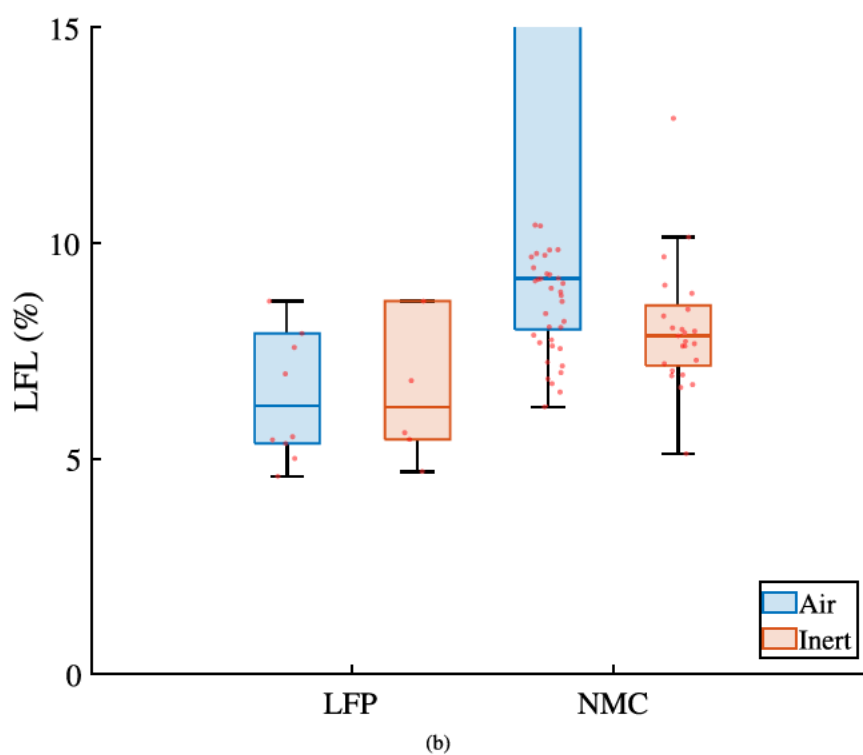
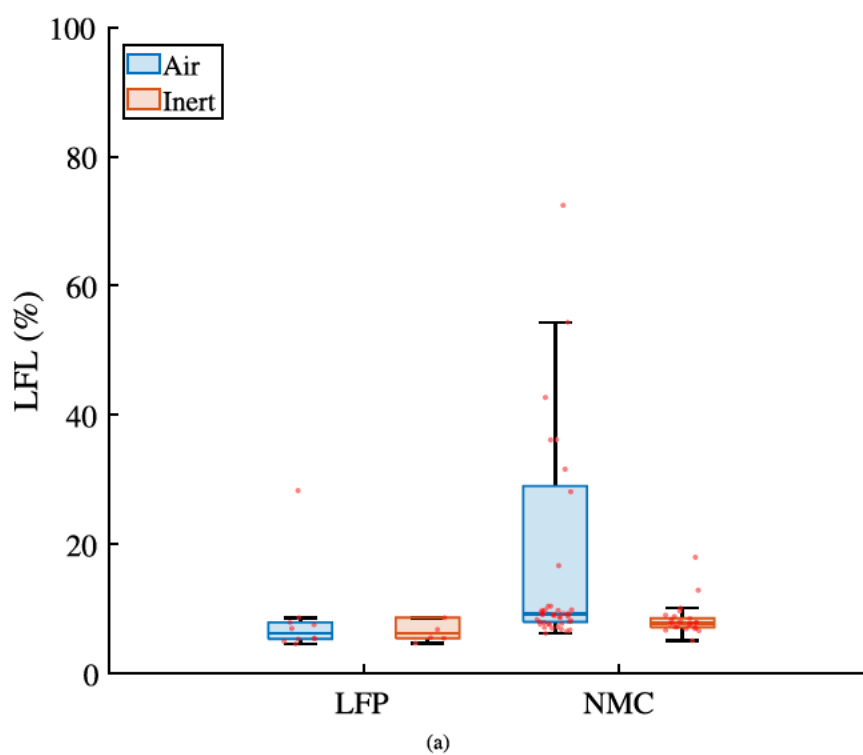


Fig. 14. (a) Calculated lower flammability limit considering CO, H₂ and hydrocarbons diluted by CO₂ (b) enlargement of data. (The number of values in each category are Chemistry [Air, Inert]: LFP [10, 6] and NMC [49, 25].)

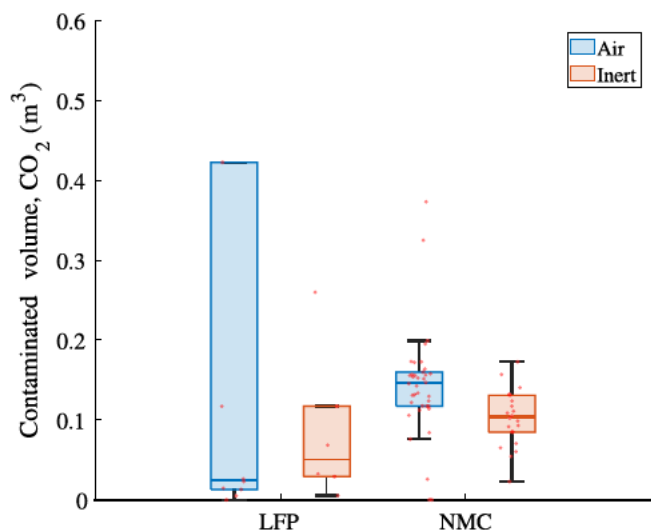


Fig. 15. Contaminated volume from CO₂ assuming 0.01 kWh battery. (The number of values in each category are Chemistry [Air, Inert]: LFP [10, 6] and NMC [38, 22].).

CRediT authorship contribution statement

Peter J. Bugryniec: Writing – review & editing, Writing – original draft, Visualization, Methodology, Investigation, Formal analysis, Data curation, Conceptualization. **Erik G. Resendiz:** Writing – review & editing, Data curation. **Solomon M. Nwophoke:** Writing – review & editing, Data curation. **Simran Khanna:** Writing – review & editing, Data curation. **Charles James:** Writing – review & editing, Data curation. **Solomon F. Brown:** Writing – review & editing, Supervision, Project administration, Funding acquisition, Conceptualization.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Data availability

The data is made available through the CSV files available as supplementary material.

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Appendix A. Supplementary data

Supplementary material related to this article can be found online at <https://doi.org/10.1016/j.est.2024.111288>.

References

- [1] B. Truchot, F. Fouillen, S. Collet, An experimental evaluation of toxic gas emissions from vehicle fires, *Fire Saf. J.* 97 (2018) 111–118, <https://dx.doi.org/10.1016/j.firesaf.2017.12.002>, URL <https://www.sciencedirect.com/science/article/pii/S0379711217307312>.
- [2] L.D. Mellert, U. Welte, M. Hermann, M. Kompatscher, X. Ponticq, V. Hagerbach, Electric mobility and road tunnel safety hazards of electric vehicle fires, 2018.
- [3] A. Lecocq, M. Bertana, B. Truchot, G. Mañair, Comparison of the fire consequences of an electric vehicle and an internal combustion engine vehicle, in: 2. International Conference on Fires in Vehicles, FIVE 2012, SP Technical Research Institute of Sweden, Borås, Chicago, United States, 2012, pp. 183–194, URL <https://ineris.hal.science/ineris-00973680>.

- [4] O. Willstrand, R. Bisschop, P. Blomqvist, A. Temple, J. Anderson, Toxic Gases from Fire in Electric Vehicles, Tech. Rep., RISE, 2020, URL <https://www.diva-portal.org/smash/get/diva2:1522149/FULLTEXT01.pdf>.
- [5] P.-J. Sturm, P.F. Leitner, D. Fruhwirt, S.F. Heindl, B. Kohl, O. Heger, R. Galler, R. Wenighofer, S. Krausbar, Brandauswirkungen von Fahrzeugen mit alternativen Antriebssystemen Tech. Rep., (Fire Effects of Vehicles with Alternative Drive Systems) 2021, [In German]. URL https://pure.tugraz.at/ws/portalfiles/portal/38365549/BRAFA_Ergebnisbericht_v1.0.pdf.
- [6] P. Sun, R. Bisschop, H. Niu, X. Huang, A review of battery fires in electric vehicles, *Fire Technol.* 56 (4) (2020) 1361–1410, <http://dx.doi.org/10.1007/s10694-019-00944-3>.
- [7] O. Willstrand, M. Pushp, P. Andersson, D. Brandell, Impact of different Li-ion cell test conditions on thermal runaway characteristics and gas release measurements, *J. Energy Storage* 68 (2023) 107785, <http://dx.doi.org/10.1016/j.est.2023.107785>, URL <https://www.sciencedirect.com/science/article/pii/S2352152X23011829>.
- [8] R. Tschirschwitz, C. Bernardy, P. Wagner, T. Rappsilber, C. Liebner, S.-K. Hahn, U. Krause, Harmful effects of lithium-ion battery thermal runaway: scale-up tests from cell to second-life modules, *RSC Adv.* 13 (2023) 20761–20779, <http://dx.doi.org/10.1039/D3RA02881J>.
- [9] M.B. McKinnon, S. DeCrane, S. Kerber, Four Firefighters Injured In Lithium-Ion Battery Energy Storage System Explosion - Arizona, Tech. Rep., Underwriters Laboratories Inc., UL Firefighter Safety Research Institute, Columbia, MD 21045, 2020, URL https://fsri.org/sites/default/files/2021-07/Four_Firefighters_Injured_In_Lithium_Ion_Battery_ESS_Explosion_Arizona_0.pdf.
- [10] South China Morning Post, Electric bus bursts into flames, sets nearby vehicles on fire in China, 2021, video URL <https://www.youtube.com/watch?v=T71cVhxGv4>.
- [11] Icelandic Transport Authority Conference, 1st international symposium on fire in electric storage at sea, 2023, URL <https://vimeo.com/830182462/aa852b93a7?share=copy>. (Accessed 30 November 2023).
- [12] H. Helgesen, S.H. ard, A.A. Langli, Study on Electrical Energy Storage for Ships: Battery Systems For Maritime Applications – Technology, Sustainability And Safety, Tech. Rep., European Maritime Safety Agency, 2020, URL <https://www.emsa.europa.eu/publications/item/3895-study-on-electrical-energy-storage-for-ships.html>.
- [13] F. Larsson, B. Mellander, Lithium-ion batteries used in electrified vehicles – General risk assessment and construction guidelines from a fire and gas release perspective, 2017, URL https://www.diva-portal.org/smash/record.jsf?dswid=2345&pid=diva2:1146859&c=1&searchType=SIMPLE&language=en&query=Lithium-ion+Batteries+used+in+Electrified+Vehicles+-+General+Risk+Assessment+and+Construction+Guidelines+from+a+Fire+and+Gas+Release+Perspective+af=&aq=&aq2=&aqe=&noOfRows=50&sortOrder=author_sort_asc&sortOrder2=title_sort_asc&onlyFullText=false&sf=all.
- [14] M.K.H. Hsieh, M.C. Lai, H.S.N. Sim, X. Lim, S.F.D. Fok, J. Joethy, T.Y. Kong, G.J.S. Lim, Electric scooter battery detonation: A case series and review of literature, *Ann. Burns Fire Disasters* 34 (2021) 264–276, URL <https://www.ncbi.nlm.nih.gov/pmc/articles/PMC8534310/>.
- [15] Electrical Safety First, Battery breakdown why are e-scooter and e-bike batteries exploding in people's homes and what can be done about it? 2023, URL https://www.electricalsafetyfirst.org.uk/media/syikuwb/esf_batterybreakdown_report_2023_v7_-_final.pdf. (Accessed 30 January 2024).
- [16] C. Meraner, Car park fires: A review of fire incidents, progress in research and future challenges, in: Proceedings of Seventh International Conference on Fires in Vehicles, Stavanger, Norway, 2023, pp. 7–23, URL <https://urn.kb.se/resolve?urn=urn:nbn:seri:diva-71489>.
- [17] A.R. Baird, E.J. Archibald, K.C. Marr, O.A. Ezekoye, Explosion hazards from lithium-ion battery vent gas, *J. Power Sources* 446 (2020) 227257, <http://dx.doi.org/10.1016/j.jpowsour.2019.227257>, URL <https://www.sciencedirect.com/science/article/pii/S0378775319312509>.
- [18] W. Li, S. Rao, Y. Xiao, Z. Gao, Y. Chen, H. Wang, M. Ouyang, Fire boundaries of lithium-ion cell eruption gases caused by thermal runaway, *iScience* 24 (2021) 102401, <http://dx.doi.org/10.1016/j.isci.2021.102401>, URL <https://www.sciencedirect.com/science/article/pii/S2589004221003692>.
- [19] T. Rappsilber, N. Yusfi, S. Krüger, S.-K. Hahn, T.-P. Fellingner, J. Krug von Nidda, R. Tschirschwitz, Meta-analysis of heat release and smoke gas emission during thermal runaway of lithium-ion batteries, *J. Energy Storage* 60 (2023) 106579, <http://dx.doi.org/10.1016/j.est.2022.106579>, URL <https://www.sciencedirect.com/science/article/pii/S2352152X22025683>.
- [20] Y. Miao, P. Hynan, A. von Jouanne, A. Yokochi, Current Li-ion battery technologies in electric vehicles and opportunities for advancements, *Energies* 12 (6) (2019) <http://dx.doi.org/10.3390/en12061074>, URL <https://www.mdpi.com/1996-1073/12/6/1074>.
- [21] W. Mrozik, M. Wise, N. Dickman, M. Ahmied, Z. Milojevic, P. Das, S. Lambert, P. Christensen, Abuse of lithium-ion batteries: emergence, composition, and toxicity of vapour cloud, in: Book of Abstracts Nordic Fire and Safety Days, 2022, pp. 86–87, <http://dx.doi.org/10.23699/sg7-kd69>, URL <https://www.diva-portal.org/smash/record.jsf?pid=diva2:1657152&dsid=8687>.
- [22] P. Bugryniec, Li-ion Battery Thermal Runaway Flowchart, 2023, <http://dx.doi.org/10.13140/RG.2.2.34249.01120>.

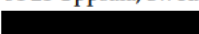
- [23] C. Essl, A.W. Golubkov, E. Gasser, M. Nachtnebel, A. Zankel, E. Ewert, A. Fuchs, Comprehensive hazard analysis of failing automotive lithium-ion batteries in overtemperature experiments, *Batteries* 6 (2) (2020) <http://dx.doi.org/10.3390/batteries6020030>, URL <https://www.mdpi.com/2313-0105/6/2/30>.
- [24] A.W. Golubkov, D. Fuchs, J. Wagner, H. Wilsche, C. Stangl, G. Fauler, G. Voitic, A. Thaler, V. Hacker, Thermal-runaway experiments on consumer Li-ion batteries with metal-oxide and olivin-type cathodes, *RSC Adv.* 4 (2014) 3633–3642, <http://dx.doi.org/10.1039/C3RA45748F>.
- [25] A.W. Golubkov, S. Scheikl, R. Planteu, G. Voitic, H. Wilsche, C. Stangl, G. Fauler, A. Thaler, V. Hacker, Thermal runaway of commercial 18650 Li-ion batteries with LFP and NCA cathodes – impact of state of charge and overcharge, *RSC Adv.* 5 (2015) 57171–57186, <http://dx.doi.org/10.1039/C5RA05897J>.
- [26] V. Somandepalli, K. Marr, Q. Horn, Quantification of combustion hazards of thermal runaway failures in lithium-ion batteries, *SAE Int. J. Altern. Powertrains* 3 (1) (2014) 98–104, <http://dx.doi.org/10.4271/2014-01-1857>.
- [27] S. Koch, A. Fill, K.P. Birke, Comprehensive gas analysis on large scale automotive lithium-ion cells in thermal runaway, *J. Power Sources* 398 (2018) 106–112, <http://dx.doi.org/10.1016/j.jpowsour.2018.07.051>, URL <https://www.sciencedirect.com/science/article/pii/S0378775318307687>.
- [28] A.W. Golubkov, R. Planteu, P. Krohn, B. Rasch, B. Brunnsteiner, A. Thaler, V. Hacker, Thermal runaway of large automotive Li-ion batteries, *RSC Adv.* 8 (2018) 40172–40186, <http://dx.doi.org/10.1039/C8RA06458J>.
- [29] D. Sturk, L. Rosell, P. Blomqvist, A. Ahlberg Tidblad, Analysis of Li-Ion battery gases vented in an inert atmosphere thermal test chamber, *Batteries* 5 (3) (2019) <http://dx.doi.org/10.3390/batteries5030061>, URL <https://www.mdpi.com/2313-0105/5/3/61>.
- [30] K.O.A. Amano, S.-K. Hahn, N. Butt, P. Vorwerk, E. Gimadiev, R. Tschirschwitz, T. Rappsilber, U. Krause, Composition and explosibility of gas emissions from lithium-ion batteries undergoing thermal runaway, *Batteries* 9 (6) (2023) <http://dx.doi.org/10.3390/batteries9060300>, URL <https://www.mdpi.com/2313-0105/9/6/300>.
- [31] H. Shen, H. Wang, M. Li, C. Li, Y. Zhang, Y. Li, X. Yang, X. Feng, M. Ouyang, Thermal runaway characteristics and gas composition analysis of lithium-ion batteries with different LFP and NCM cathode materials under inert atmosphere, *Electronics* 12 (7) (2023) <http://dx.doi.org/10.3390/electronics12071603>, URL <https://www.mdpi.com/2079-9292/12/7/1603>.
- [32] X. Yang, H. Wang, M. Li, Y. Li, C. Li, Y. Zhang, S. Chen, H. Shen, F. Qian, X. Feng, M. Ouyang, Experimental study on thermal runaway behavior of lithium-ion battery and analysis of combustible limit of gas production, *Batteries* 8 (11) (2022) <http://dx.doi.org/10.3390/batteries8110250>, URL <https://www.mdpi.com/2313-0105/8/11/250>.
- [33] D.J. Robles, J. Jeevarajan, Fire and gas characterization studies for lithium-ion cells and batteries, 2020, NASA Aerospace Battery Workshop. URL https://www.nasa.gov/sites/default/files/atoms/files/nabw20_fire_gas_char_studies_liion_cells_batt_djuarez-robles.pdf.
- [34] L. Yuan, T. Dubaniewicz, I. Zlochower, R. Thomas, N. Rayyan, Experimental study on thermal runaway and vented gases of lithium-ion cells, *Process Saf. Environ. Prot.* 144 (2020) 186–192, <http://dx.doi.org/10.1016/j.psep.2020.07.028>, URL <https://www.sciencedirect.com/science/article/pii/S0957582020316360>.
- [35] S. Hoelle, S. Scharner, S. Asanin, O. Hinrichsen, Analysis on thermal runaway behavior of prismatic lithium-ion batteries with autoclave calorimetry, *J. Electrochem. Soc.* 168 (12) (2021) 120515, <http://dx.doi.org/10.1149/1945-7111/ac3c27>.
- [36] C. Essl, A.W. Golubkov, A. Fuchs, Comparing different thermal runaway triggers for two automotive lithium-ion battery cell types, *J. Electrochem. Soc.* 167 (13) (2020) 130542, <http://dx.doi.org/10.1149/1945-7111/abbe5a>.
- [37] K.C. Abbott, J.E. Buston, J. Gill, S.L. Goddard, D. Howard, G. Howard, E. Read, R.C. Williams, Comprehensive gas analysis of a 21700 Li(Ni_{0.8}Co_{0.1}Mn_{0.1}O₂) cell using mass spectrometry, *J. Power Sources* 539 (2022) 231585, <http://dx.doi.org/10.1016/j.jpowsour.2022.231585>, URL <https://www.sciencedirect.com/science/article/pii/S0378775322005882>.
- [38] K.O.A. Amano, S.-K. Hahn, R. Tschirschwitz, T. Rappsilber, U. Krause, An experimental investigation of thermal runaway and gas release of NMC lithium-ion pouch batteries depending on the state of charge level, *Batteries* 8 (5) (2022) <http://dx.doi.org/10.3390/batteries8050041>, URL <https://www.mdpi.com/2313-0105/8/5/41>.
- [39] P. Liu, H. Sun, Y. Qiao, S. Sun, C. Wang, K. Jin, B. Mao, Q. Wang, Experimental study on the thermal runaway and fire behavior of LiNi_{0.8}Co_{0.1}Mn_{0.1}O₂ battery in open and confined spaces, *Process Saf. Environ. Prot.* 158 (2022) 711–726, <http://dx.doi.org/10.1016/j.psep.2021.12.056>, URL <https://www.sciencedirect.com/science/article/pii/S0957582021007321>.
- [40] M. Yang, M. Rong, J. Pan, Y. Ye, A. Yang, J. Chu, H. Yuan, X. Wang, Thermal runaway behavior analysis during overheating for commercial LiFePO₄ batteries under various state of charges, *Appl. Therm. Eng.* 230 (2023) 120816, <http://dx.doi.org/10.1016/j.applthermaleng.2023.120816>, URL <https://www.sciencedirect.com/science/article/pii/S1359431123008451>.
- [41] DNV GL, Maritime Battery Safety Joint Development project: Technical Reference for Li-ion Battery Explosion Risk and Fire Suppression, Tech. Rep., 2019, URL https://safety4sea.com/wp-content/uploads/2020/01/DNV-GL-Technical-Reference-for-Li-Ion-Battery-Explosion-Risk-and-Fire-Suppression-2020_01.pdf.
- [42] P. Liu, C. Liu, K. Yang, M. Zhang, F. Gao, B. Mao, H. Li, Q. Duan, Q. Wang, Thermal runaway and fire behaviors of lithium iron phosphate battery induced by over heating, *J. Energy Storage* 31 (2020) 101714, <http://dx.doi.org/10.1016/j.est.2020.101714>, URL <https://www.sciencedirect.com/science/article/pii/S2352152X20315516>.
- [43] L. Zhang, Q. Duan, X. Meng, K. Jin, J. Xu, J. Sun, Q. Wang, Experimental investigation on intermittent spray cooling and toxic hazards of lithium-ion battery thermal runaway, *Energy Convers. Manage.* 252 (2022) 115091, <http://dx.doi.org/10.1016/j.enconman.2021.115091>, URL <https://www.sciencedirect.com/science/article/pii/S019689042101267X>.
- [44] U. Bergström, A. Gustafsson, Lars Hägglund, C. Lejon, D. Sturk, T. Tengel, Vented Gases and Aerosol of Automotive Li-ion LFP and NMC Batteries in Humidified Nitrogen under Thermal Load, Tech. Rep., 2015, URL <https://www.msb.se/sitesassets/dokument/publikationer/english-publications/vented-gases-and-aerosol-of-automotive-li-ion-lfp-and-nmc-batteries-in-humidified-nitrogen-under-thermal-load.pdf>.
- [45] A. Nedjalkov, J. Meyer, M. Köhring, A. Doering, M. Angelmahr, S. Dahle, A. Sander, A. Fischer, W. Schade, Toxic gas emissions from damaged lithium ion batteries—Analysis and safety enhancement solution, *Batteries* 2 (1) (2016) <http://dx.doi.org/10.3390/batteries2010005>, URL <https://www.mdpi.com/2313-0105/2/1/5>.
- [46] Y. Zhang, H. Wang, W. Li, C. Li, Quantitative identification of emissions from abused prismatic Ni-rich lithium-ion batteries, *eTransportation* 2 (2019) 100031, <http://dx.doi.org/10.1016/j.etrans.2019.100031>, URL <https://www.sciencedirect.com/science/article/pii/S2590116819300311>.
- [47] Health and Safety Executive, EH40/2005 Workplace Exposure Limits, fourth ed., 2020, URL <https://www.hse.gov.uk/pubns/priced/eh40.pdf> ISBN 978 0 7176 6733 8.
- [48] Y. Fernandes, A. Bry, S. de Persis, Identification and quantification of gases emitted during abuse tests by overcharge of a commercial Li-ion battery, *J. Power Sources* 389 (2018) 106–119, <http://dx.doi.org/10.1016/j.jpowsour.2018.03.034>, URL <https://www.sciencedirect.com/science/article/pii/S0378775318302581>.
- [49] K. Zou, S. Lu, X. Chen, E. Gao, Y. Cao, Y. Bi, Thermal and gas characteristics of large-format LiNi_{0.8}Co_{0.1}Mn_{0.1}O₂ pouch power cell during thermal runaway, *J. Energy Storage* 39 (2021) 102609, <http://dx.doi.org/10.1016/j.est.2021.102609>, URL <https://www.sciencedirect.com/science/article/pii/S2352152X21003510>.
- [50] F. Diaz, Y. Wang, R. Weyhe, B. Friedrich, Gas generation measurement and evaluation during mechanical processing and thermal treatment of spent Li-ion batteries, *Waste Manag.* 84 (2019) 102–111, <http://dx.doi.org/10.1016/j.wasman.2018.11.029>, URL <https://www.sciencedirect.com/science/article/pii/S0956053X18307013>.
- [51] T. Shan, X. Zhu, Z. Wang, Understanding the boundary and mechanism of gas-induced explosion for lithium-ion cells: Experimental and theoretical analysis, *J. Energy Chem.* (2023) <http://dx.doi.org/10.1016/j.jechem.2023.07.029>, URL <https://www.sciencedirect.com/science/article/pii/S2095495623004291>.
- [52] Health and Safety Executive, General hazards of Carbon Dioxide <https://www.hse.gov.uk/carboncapture/carbondioxide.htm#:~:text=AstheconcentrationCO2,orcelsarsanddisplacedOxygen>.
- [53] ECHA, Search for Chemicals URL <https://echa.europa.eu/search-for-chemicals>.
- [54] IFA Institut für Arbeitsschutz der Deutschen Gesetzlichen Unfallversicherung, GESTIS International Limit Values, 2023, URL <https://limitvalue.ifa.dguv.de/>.
- [55] National Research Council (US) Committee on Toxicology, Emergency and continuous exposure limits for selected airborne contaminants: Volume 1, 1984, URL <https://www.ncbi.nlm.nih.gov/books/NBK208285/>.
- [56] A.O. Said, C. Lee, S.I. Stolarov, Experimental investigation of cascading failure in 18650 lithium ion cell arrays: Impact of cathode chemistry, *J. Power Sources* 446 (2020) 227347, <http://dx.doi.org/10.1016/j.jpowsour.2019.227347>, URL <https://www.sciencedirect.com/science/article/pii/S0378775319313400>.
- [57] T. Maloney, Solid state battery tests, 2017, URL https://www.fire.tc.faa.gov/pdf/systems/Nov17Meeting/Maloney-1117-Solid_State_battery_tests.pdf FAA Fire Safety, Systems Meeting.
- [58] Q. Zhang, J. Niu, Z. Zhao, Q. Wang, Research on the effect of thermal runaway gas components and explosion limits of lithium-ion batteries under different charge states, *J. Energy Storage* 45 (2022) 103759, <http://dx.doi.org/10.1016/j.est.2021.103759>, URL <https://www.sciencedirect.com/science/article/pii/S2352152X21014328>.
- [59] M. Karp, J. Sica, Flammable gas analysis UN lithium battery working group, in: International Aircraft Systems Fire Protection Working Group, FAA, 2021, URL <https://www.fire.tc.faa.gov/pdf/systems/April21Meeting/Karp-0421-UNGas.pdf>.
- [60] P. Ribière, S. Grugeon, M. Morcrette, S. Boyanov, S. Laruelle, G. Marlair, Investigation on the fire-induced hazards of Li-ion battery cells by fire calorimetry, *Energy Environ. Sci.* 5 (2012) 5271–5280, <http://dx.doi.org/10.1039/C1EE02218K>.
- [61] A. Lecocq, G.G. Eshetu, S. Grugeon, N. Martin, S. Laruelle, G. Marlair, Scenario-based prediction of Li-ion batteries fire-induced toxicity, *J. Power Sources* 316 (2016) 197–206, <http://dx.doi.org/10.1016/j.jpowsour.2016.02.090>, URL <https://www.sciencedirect.com/science/article/pii/S0378775316301975>.

- [62] T. Maloney, Lithium battery thermal runaway vent gas analysis composition and effect of combustion, 2015, URL <https://fire.tc.faa.gov/pdf/systems/May15Meeting/Maloney-0515-LithiumThermalRunaway.pdf> FAA presentation.
- [63] L. Rong, X. Cheng, Y. Fu, Effect of States of Charge on the Burning Behaviors of Lithium ion Batteries, in: 2019 9th International Conference on Fire Science and Fire Protection Engineering, ICFSFPE, 2019, pp. 1–5, <http://dx.doi.org/10.1109/ICFSFPE48751.2019.9055886>, URL <https://ieeexplore.ieee.org/document/9055886>.
- [64] Y. Peng, X. Zhou, Y. Hu, X. Ju, B. Liao, L. Yang, A new exploration of the fire behaviors of large format lithium ion battery, *J. Therm. Anal. Calorim.* 139 (2) (2020) 1243–1254, <http://dx.doi.org/10.1007/s10973-019-08459-3>.
- [65] Q. Zhang, J. Niu, J. Yang, T. Liu, F. Bao, Q. Wang, In-situ explosion limit analysis and hazards research of vent gas from lithium-ion battery thermal runaway, *J. Energy Storage* 56 (2022) 106146, <http://dx.doi.org/10.1016/j.est.2022.106146>, URL <https://www.sciencedirect.com/science/article/pii/S2352152X22021351>.
- [66] M. Yang, M. Rong, Y. Ye, A. Yang, J. Chu, H. Yuan, X. Wang, Comprehensive analysis of gas production for commercial LiFePO₄ batteries during overcharge-thermal runaway, *J. Energy Storage* 72 (2023) 108323, <http://dx.doi.org/10.1016/j.est.2023.108323>, URL <https://www.sciencedirect.com/science/article/pii/S2352152X23017206>.
- [67] E.P. Roth, C.J. Orendorff, How electrolytes influence battery safety, *Electrochem. Soc. Interface* 21 (2) (2012) 45, <http://dx.doi.org/10.1149/2.F04122if>.
- [68] J. Lamb, C.J. Orendorff, E.P. Roth, J. Langendorf, Studies on the thermal breakdown of common Li-ion battery electrolyte components, *J. Electrochem. Soc.* 162 (10) (2015) A2131, <http://dx.doi.org/10.1149/2.0651510jes>.
- [69] Y. Fernandes, A. Bry, S. de Persis, Thermal degradation analyses of carbonate solvents used in Li-ion batteries, *J. Power Sources* 414 (2019) 250–261, <http://dx.doi.org/10.1016/j.jpowsour.2018.12.077>, URL <https://www.sciencedirect.com/science/article/pii/S0378775318314289>.
- [70] P.J. Bugryniec, S. Vernuccio, S.F. Brown, Towards a micro-kinetic model of Li-ion battery thermal runaway — reaction network analysis of dimethyl carbonate thermal decomposition, *J. Power Sources* 580 (2023) 233394, <http://dx.doi.org/10.1016/j.jpowsour.2023.233394>, URL <https://www.sciencedirect.com/science/article/pii/S037877532300770X>.
- [71] Q. Wang, P. Ping, X. Zhao, G. Chu, J. Sun, C. Chen, Thermal runaway caused fire and explosion of lithium ion battery, *J. Power Sources* 208 (2012) 210–224, <http://dx.doi.org/10.1016/j.jpowsour.2012.02.038>, URL <https://www.sciencedirect.com/science/article/pii/S0378775312003989>.
- [72] X. Feng, M. Ouyang, X. Liu, L. Lu, Y. Xia, X. He, Thermal runaway mechanism of lithium ion battery for electric vehicles: A review, *Energy Storage Mater.* 10 (2018) 246–267, <http://dx.doi.org/10.1016/j.ensm.2017.05.013>, URL <https://www.sciencedirect.com/science/article/pii/S2405829716303464>.
- [73] D.H. Doughty, E.P. Roth, C.C. Crafts, G. Nagasubramanian, G. Henriksen, K. Amine, Effects of additives on thermal stability of Li ion cells, *J. Power Sources* 146 (1) (2005) 116–120, <http://dx.doi.org/10.1016/j.jpowsour.2005.03.170>, URL <https://www.sciencedirect.com/science/article/pii/S0378775305005057> Selected papers presented at the 12th International Meeting on Lithium Batteries.
- [74] F. Zheng, M. Kotobuki, S. Song, M.O. Lai, L. Lu, Review on solid electrolytes for all-solid-state lithium-ion batteries, *J. Power Sources* 389 (2018) 198–213, <http://dx.doi.org/10.1016/j.jpowsour.2018.04.022>, URL <https://www.sciencedirect.com/science/article/pii/S0378775318303653>.
- [75] D. Sturk, L. Hoffmann, A.A. Tidblad, Fire tests on E-vehicle battery cells and packs, *Traffic Inj. Prev.* 16 (sup1) (2015) S159–S164, <http://dx.doi.org/10.1080/15389588.2015.1015117>.
- [76] F. Larsson, P. Andersson, P. Blomqvist, B.-E. Mellander, Toxic fluoride gas emissions from lithium-ion battery fires, *Sci. Rep.* 7 (1) (2017) 10018, <http://dx.doi.org/10.1038/s41598-017-09784-z>.
- [77] Z. Huang, C. Zhao, H. Li, W. Peng, Z. Zhang, Q. Wang, Experimental study on thermal runaway and its propagation in the large format lithium ion battery module with two electrical connection modes, *Energy* 205 (2020) 117906, <http://dx.doi.org/10.1016/j.energy.2020.117906>, URL <https://www.sciencedirect.com/science/article/pii/S0360544220310136>.
- [78] T. Zhou, J. Sun, J. Li, S. Wei, J. Chen, S. Dang, N. Tang, Y. Zhu, Y. Lian, J. Guo, F. Zhang, H. Xie, H. Li, X. Qiu, L. Chen, Toxicity, emissions and structural damage from lithium-ion battery thermal runaway, *Batteries* 9 (6) (2023) <http://dx.doi.org/10.3390/batteries9060308>, URL <https://www.mdpi.com/2313-0105/9/6/308>.
- [79] F. Larsson, P. Andersson, B.-E. Mellander, Lithium-ion battery aspects on fires in electrified vehicles on the basis of experimental abuse tests, *Batteries* 2 (2) (2016) <http://dx.doi.org/10.3390/batteries2020009>, URL <https://www.mdpi.com/2313-0105/2/2/9>.
- [80] C.F. Larsson, P. Andersson, P. Blomqvist, B.-E. Mellander, Gas emissions from lithium-ion battery cells undergoing abuse from external fire, in: *Fires in Vehicles (FIVE) 2016 Conference Proceedings*, Vol. 2016, 2016, pp. 253–256, URL <https://research.chalmers.se/en/publication/243272>.
- [81] J. Sun, J. Li, T. Zhou, K. Yang, S. Wei, N. Tang, N. Dang, H. Li, X. Qiu, L. Chen, Toxicity, a serious concern of thermal runaway from commercial Li-ion battery, *Nano Energy* 27 (2016) 313–319, <http://dx.doi.org/10.1016/j.nanoen.2016.06.031>, URL <https://www.sciencedirect.com/science/article/pii/S2211285516302130>.
- [82] P. Andersson, P. Blomqvist, A. Lorén, F. Larsson, Investigation of Fire Emissions from Li-Ion Batteries, SP Technical Research Institute of Sweden, 2013, URL <https://www.diva-portal.org/smash/get/diva2:962743/FULLTEXT01.pdf>.
- [83] G. Siegemund, W. Schwertfeger, A. Feiring, B. Smart, F. Behr, H. Vogel, B. McKusick, Fluorine compounds, organic, in: *Ullmann's Encyclopedia of Industrial Chemistry*, John Wiley & Sons, Ltd, 2000, p. 461, http://dx.doi.org/10.1002/14356007.a11_349, arXiv:https://onlinelibrary.wiley.com/doi/pdf/10.1002/14356007.a11_349 URL https://onlinelibrary.wiley.com/doi/abs/10.1002/14356007.a11_349.
- [84] S. Ogunfuye, H. Sezer, A.O. Said, A. Simeoni, V. Akkerman, An analysis of gas-induced explosions in vented enclosures in lithium-ion batteries, *J. Energy Storage* 51 (2022) 104438, <http://dx.doi.org/10.1016/j.est.2022.104438>, URL <https://www.sciencedirect.com/science/article/pii/S2352152X22004613>.
- [85] N.P. Lebedeva, L. Boon-Brett, Considerations on the chemical toxicity of contemporary Li-ion battery electrolytes and their components, *J. Electrochem. Soc.* 163 (6) (2016) A821, <http://dx.doi.org/10.1149/2.0171606jes>.
- [86] Federal Emergency Management Agency, U.S. Department of Transportation, U.S. Environmental Protection Agency, Handbook Of Chemical Hazard Analysis Procedures, National Service Center for Environmental Publications (NSCEP), 1989, URL <https://nepis.epa.gov/Exec/ZipURL.cgi?Dockey=10003MK5.txt>.
- [87] A.A. Abd-El-Latif, P. Sichter, M. Kasper, T. Waldmann, M. Wohlfahrt-Mehrens, Insights into thermal runaway of Li-Ion cells by accelerating rate calorimetry coupled with external sensors and online gas analysis, *Batter. Supercaps* 4 (7) (2021) 1135–1144, <http://dx.doi.org/10.1002/batt.202100023>, arXiv:<https://chemistry-europe.onlinelibrary.wiley.com/doi/pdf/10.1002/batt.202100023> URL <https://chemistry-europe.onlinelibrary.wiley.com/doi/abs/10.1002/batt.202100023>.
- [88] M. Lammer, A. Königseder, V. Hacker, Holistic methodology for characterisation of the thermally induced failure of commercially available 18650 lithium ion cells, *RSC Adv.* 7 (2017) 24425–24429, <http://dx.doi.org/10.1039/C7RA02635H>.
- [89] M. Lammer, A. Königseder, P. Gluschitz, V. Hacker, Influence of aging on the heat and gas emissions from commercial lithium ion cells in case of thermal failure, *J. Electrochem. Sci. Eng.* 8 (1) (2018) 101–110, <http://dx.doi.org/10.5599/jese.476>, URL <https://pub.iapchem.org/ojs/index.php/JESE/article/view/476>.
- [90] M. Karp, Heat rate and vent gas analysis thermal runaway vent gas analysis of 18650 and pouch cell, in: 9th Triennial Fire and Safety Cabin Conference, 2019, URL https://www.fire.tc.faa.gov/2019Conference/files/Battery_V/KarpThermalRunawayAnalysis/KarpHeatRateandVentGasAnalysis.pdf.
- [91] C. Essl, A.W. Golubkov, A. Fuchs, Influence of aging on the failing behavior of automotive lithium-ion batteries, *Batteries* 7 (2) (2021) <http://dx.doi.org/10.3390/batteries7020023>, URL <https://www.mdpi.com/2313-0105/7/2/23>.
- [92] T. Ohsaki, T. Kishi, T. Kuboki, N. Takami, N. Shimura, Y. Sato, M. Sekino, A. Satoh, Overcharge reaction of lithium-ion batteries, *J. Power Sources* 146 (1) (2005) 97–100, <http://dx.doi.org/10.1016/j.jpowsour.2005.03.105>, URL <https://www.sciencedirect.com/science/article/pii/S0378775305005112> Selected papers presented at the 12th International Meeting on Lithium Batteries.
- [93] Q. Yuan, F. Zhao, W. Wang, Y. Zhao, Z. Liang, D. Yan, Overcharge failure investigation of lithium-ion batteries, *Electrochim. Acta* 178 (2015) 682–688, <http://dx.doi.org/10.1016/j.electacta.2015.07.147>, URL <https://www.sciencedirect.com/science/article/pii/S0013468615302140>.
- [94] K.C. Abbott, J.E. Buston, J. Gill, S.L. Goddard, D. Howard, G.E. Howard, E. Read, R.C. Williams, Experimental study of three commercially available 18650 lithium ion batteries using multiple abuse methods, *J. Energy Storage* 65 (2023) 107293, <http://dx.doi.org/10.1016/j.est.2023.107293>, URL <https://www.sciencedirect.com/science/article/pii/S2352152X23006904>.
- [95] C.M. Vendra, A.V. Shelke, J.E. Buston, J. Gill, D. Howard, E. Read, A. Abaza, B. Cooper, J.X. Wen, Numerical and experimental characterisation of high energy density 21700 lithium-ion battery fires, *Process Saf. Environ. Prot.* 160 (2022) 153–165, <http://dx.doi.org/10.1016/j.psep.2022.02.014>, URL <https://www.sciencedirect.com/science/article/pii/S0957582022001082>.
- [96] E.V. FireSafe, Mercedes Vito electric van in thermal runaway near Heathrow Airport, 2023, video URL <https://www.youtube.com/watch?v=m1ldMkwKLP4>.
- [97] R.W. Kennedy, K.C. Marr, O.A. Ezekoye, Gas release rates and properties from lithium cobalt oxide lithium ion battery arrays, *J. Power Sources* 487 (2021) 229388, <http://dx.doi.org/10.1016/j.jpowsour.2020.229388>, URL <https://www.sciencedirect.com/science/article/pii/S0378775320316724>.
- [98] C.C. Crafts, D.H. Doughty, J. McBreen, E.P. Roth, Advanced technology development program for lithium-ion batteries : thermal abuse performance of 18650 Li-ion cells, 2004, <http://dx.doi.org/10.2172/918751>, URL <https://www.osti.gov/biblio/918751>.
- [99] M. Karp, Lithium ion thermal runaway less than 2 wh lithium ion batteries, in: *International Aircraft Systems Fire Protection Working Group, FAA*, 2017, URL <https://www.fire.tc.faa.gov/pdf/systems/Nov17Meeting/Karp-1117-2WhBatteries.pdf>.

- [100] FAA Fire Safety, Solid state battery tests, 2017, URL https://www.fire.tc.faa.gov/pdf/systems/Nov17Meeting/Maloney-1117-Solid_State_battery_tests.pdf.
- [101] S. Chen, Z. Wang, J. Wang, X. Tong, W. Yan, Lower explosion limit of the vented gases from Li-ion batteries thermal runaway in high temperature condition, *J. Loss Prev. Process Ind.* 63 (2020) 103992, <http://dx.doi.org/10.1016/j.jlp.2019.103992>, URL <https://www.sciencedirect.com/science/article/pii/S0950423018309744>.
- [102] Z. Huang, X. Li, Q. Wang, Q. Duan, Y. Li, L. Li, Q. Wang, Experimental investigation on thermal runaway propagation of large format lithium ion battery modules with two cathodes, *Int. J. Heat Mass Transfer* 172 (2021) 121077, <http://dx.doi.org/10.1016/j.ijheatmasstransfer.2021.121077>, URL <https://www.sciencedirect.com/science/article/pii/S0017931021001800>.
- [103] J. Hynynen, O. Willstrand, P. Blomqvist, P. Andersson, Analysis of combustion gases from large-scale electric vehicle fire tests, *Fire Saf. J.* 139 (2023) 103829, <http://dx.doi.org/10.1016/j.firesaf.2023.103829>, URL <https://www.sciencedirect.com/science/article/pii/S0379711223000978>.
- [104] W.Q. Walker, Characterization of the GS YUASA 134 ah cell thermal runaway events with large format fractional thermal runaway calorimetry (L-FTRC), 2020, URL https://www.nasa.gov/sites/default/files/atoms/files/nabw20_char_gsy134_tr_l-ftrc_wwalker.pdf.
- [105] P. Bugryniec, S. Vernuccio, S. Brown, Predicting the evolution of flammable gases during li-ion battery thermal runaway using micro-kinetic modelling, in: A.C. Kokossis, M.C. Georgiadis, E. Pistikopoulos (Eds.), 33rd European Symposium on Computer Aided Process Engineering, in: *Computer Aided Chemical Engineering*, vol. 52, Elsevier, 2023, pp. 1077–1082, <http://dx.doi.org/10.1016/B978-0-443-15274-0.50172-4>, URL <https://www.sciencedirect.com/science/article/pii/B9780443152740501724>.
- [106] J.M. Kuchta, Investigation of Fire and Explosion Accidents in the Chemical, Mining, and Fuel-Related Industries - A Manual, *Techreport Bulletin 68*, U.S. Bureau of Mines, 1985.
- [107] Battery University, Lithium-ion safety concerns, 2022, URL <https://batteryuniversity.com/article/lithium-ion-safety-concerns>. (Accessed 30 January 2024).
- [108] Renogy, Are lithium ion batteries dangerous and what are the safest lithium batteries?, 2022, URL <https://www.renogy.com/blog/are-lithium-ion-batteries-dangerous-and-what-are-the-safest-lithium-batteries/>. (Accessed 30 January 2024).
- [109] LithiumWerks, Lithium iron phosphate: The safer chemistry, 2024, URL <https://lithiumwerks.com/products/technology/safety/>. (Accessed 30 January 2024).
- [110] relionbattery, How are LiFePO4 batteries safer than other lithium batteries?, 2024, URL <https://www.relionbattery.com/knowledge/how-are-lifepo4-batteries-safer-than-other-lithium-batteries>. (Accessed 30 January 2024).
- [111] V. Ruiz, A. Pfrang, A. Kriston, N. Omar, P. Van den Bossche, L. Boon-Brett, A review of international abuse testing standards and regulations for lithium ion batteries in electric and hybrid electric vehicles, *Renew. Sustain. Energy Rev.* 81 (2018) 1427–1452, <http://dx.doi.org/10.1016/j.rser.2017.05.195>, URL <https://www.sciencedirect.com/science/article/pii/S1364032117308250>.
- [112] P.J. Bugryniec, J.N. Davidson, D.J. Cumming, S.F. Brown, Pursuing safer batteries: Thermal abuse of LiFePO4 cells, *J. Power Sources* 414 (2019) 557–568, <http://dx.doi.org/10.1016/j.jpowsour.2019.01.013>.
- [113] D.H. Doughty, C.C. Crafts, FreedomCAR: Electrical Energy Storage System Abuse Test Manual for Electric and Hybrid Electric Vehicle Applications, *Tech. Rep.*, sAND2005-3123, Sandia National Laboratories, 2006, <http://dx.doi.org/10.2172/889934>.
- [114] Z. Wang, J. Yuan, X. Zhu, H. Wang, L. Huang, Y. Wang, S. Xu, Overcharge-to-thermal-runaway behavior and safety assessment of commercial lithium-ion cells with different cathode materials: A comparison study, *J. Energy Chem.* 55 (2021) 484–498, <http://dx.doi.org/10.1016/j.jechem.2020.07.028>, URL <https://www.sciencedirect.com/science/article/pii/S2095495620305234>.
- [115] International Energy Agency, Global EV Outlook 2023, *Tech. Rep.*, International Energy Agency, 2023, p. 57, URL <https://www.iea.org/reports/global-ev-outlook-2023>.

Review

Gas Emissions from Lithium-Ion Batteries: A Review of Experimental Results and Methodologies

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Abstract: Gas emissions from lithium-ion batteries (LIBs) have been analysed in a large number of experimental studies over the last decade, including investigations of their dependence on the state of charge, cathode chemistry, cell capacity, and many more factors. Unfortunately, the reported data are inconsistent between studies, which can be explained by weaknesses in experimental methodologies, the misinterpretation of data, or simply due to the comparison of datasets that build on different prerequisites. In the present work, the literature on gassing from battery components and battery cells is reported, with a focus on vent gas composition resulting from internal chemical processing in the battery and excluding studies where the gases are combusted after venting. The aim is to identify datasets of high quality that contribute to the advancement of our understanding of gas emissions from LIBs. Gas compositions from different stages in the gassing process are included, starting with the slow formation of gases during normal operation via mild thermal events to a thermal runaway (TR) with extensive gas production. Available published data are used to map gas quantity and composition from LIBs undergoing venting, with or without a TR, and to identify gaps in understanding and the need for further research.

Keywords: lithium-ion; battery vent gas; thermal runaway; gas analysis; battery fire; gas composition; electrolyte decomposition products; cell internal pressure build-up; thermal failure; thermal event



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1. Introduction

1.1. The Research Field

Research on safety related to lithium-ion batteries (LIBs) has rapidly increased over the last couple of decades, motivated by the continually expanding use of this battery type in electric vehicles (EVs) and a variety of electrical devices. An important aspect of LIB safety is the potential for the formation of flammable and toxic gases, which occur to a small extent during regular use but can accelerate as a result of internal LIB cell failure or external interference. To understand and mitigate unwanted gas formation, research on LIB failure has been expanding, spanning a wide range of studies from mechanistic studies on the degradation of cell materials to full-scale fire studies on EVs. An overview of the research field is given in Figure 1, where the main levels of the physical system under investigation are given in the horizontal direction: LIB cell components, followed by the cell level, and finally, the assembled battery. From the ordinate line, the degree of severity of conditions is clearly increasing, which can essentially be connected to temperature regimes: normal operation is around ambient temperatures, moderate heating is in the range of 80–150 °C, thermal runaway is characterised by rapid self-heating starting at around 150–220 °C and increasing to about 1000 °C, and finally, fire can occur at combustion temperatures. The mechanisms resulting in gassing in these regimes are partly different and require different experimental methodologies to be fully understood.

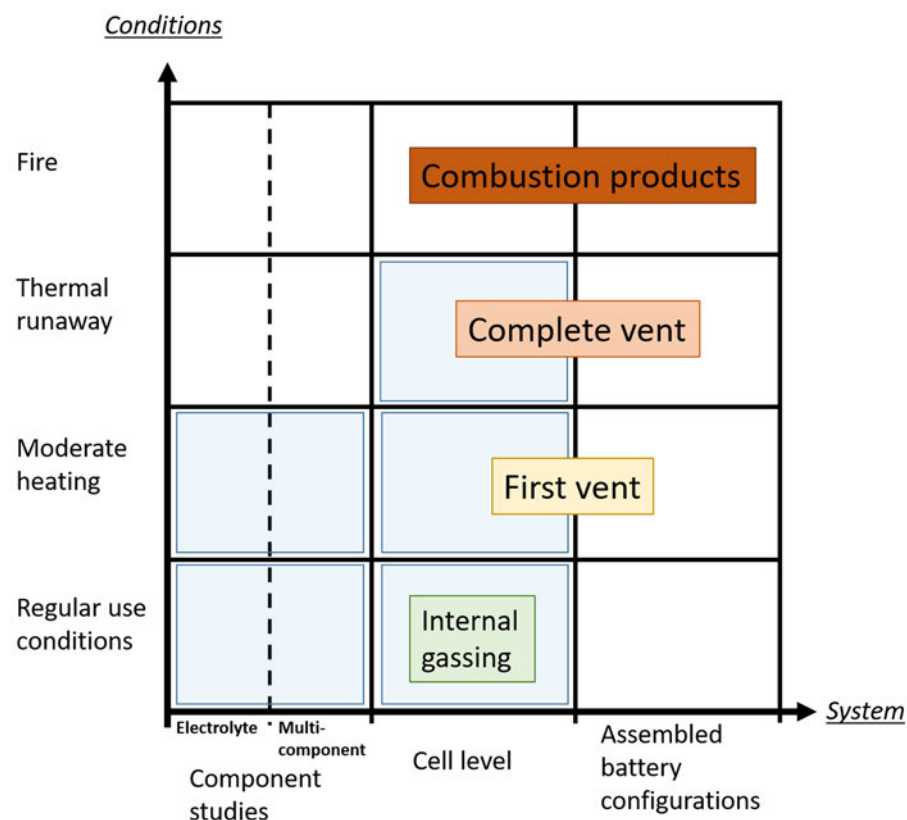


Figure 1. Schematic representation of the research field, where areas shaded in blue are the aspects focused on in the present review.

The literature includes studies on the composition of gases directly vented from the LIB cell before ignition and combustion occurs, as well as combustion products in case of a fire. In addition, selected components like electrolytes are studied separately to yield information on their contribution from different parts of the battery cell. The gassing process comprises different processes and phases, depending on multiple factors, such as the type of venting, temperature, cell chemistry (referring to cathode material), state of charge (SOC), etc., which are thought to determine the degree of chemical processing and the hazardousness of the vented gases in terms of flammability and toxicity.

The evolving research on LIB failure can be exemplified by a recent bibliometric analysis by Wang et al. [1], where keywords such as “battery” and “thermal hazard/risk/danger” were used to identify 707 publications in peer-review journals since 2010, with the majority produced since 2020. Keyword analysis in that work indicates an increasing tendency to use words like “fire” and “failure”. Keywords that were hardly used in the first part of the studied period but that emerged in the second half are “combustion”, “heat release rate”, and “fire hazard”, indicating an increased focus on these aspects. This is indicative of an increased focus on fire safety issues and the call for an improved fundamental understanding of fire and safety hazards related to LIBs.

Over the last decade, numerous review papers have been published, targeting various aspects of battery safety, such as heat release and gas formation and the mitigation of battery fires. An early review published already in 2012 by Wang et al. [2] was highly cited, but authors from the same group published a more extensive review in 2019 [3]. The later publication is the most extensive description available of the chemical mechanisms occurring inside the battery during abuse test conditions that are used to assess the safety performance of LIBs. That work also includes an overview of experimental studies of gases from abused LIBs, but it is not exhaustive on that point. An overview of gassing studies up until 2017 was presented as part of the experimental work by Fernandes et al. [4], and more recently, Qiu et al. [5] reviewed the research progress on LIB vent gas studies,

with a particular focus on the effects of ageing on gas production from thermal failure. Heat release from LIBs has been reviewed by Ghiji et al. [6], including both experimental and numerical studies to understand parametric factors on fire risk and heat release. The recent meta-analysis by Rappsilber et al. [7] points to the unfortunate fact that experimental studies by different research groups do not give a consistent picture of heat release and gassing from LIB. The discrepancies are thought to originate from the use of various experimental designs, analysis tools and procedures. Furthermore, although abuse studies give valuable information about how the battery may react in abnormal abuse conditions, their correlation to spontaneous fire incidents caused by the battery's internal failure is not straightforward. Thermal runaway (TR) caused by an internal short circuit fault in a cell, which may or may not propagate within the battery, is not necessarily a result of abuse. Hence, thermal runaway and thermal propagation failures have been notoriously challenging to replicate and study experimentally; a forced cell TR in a laboratory context evolves differently from a spontaneous TR in an LIB cell in a real application.

A review on off-gas composition and volume was recently published by Bugryniec et al. [8], with a particular focus on toxic compounds and total gas volumes. They included 60 published studies and outlined trends in gassing related to SOC, cell chemistry, and cell geometry (cylindrical, pouch, prismatic). A clear trend found was that the gas volume scaled linearly with cell capacity, but apart from that, there were significant variations in the results. One drawback in the meta-analysis was that experimental studies at various conditions were compared; for example, datasets both with and without combustion were included, which may explain a significant part of the inconsistencies.

As interpreted by the authors of the present work, experimental results are sometimes misinterpreted due to a lack of background information. Also, comparisons of different studies are made based on the assumption that they are targeting the same property or process. As an example, CO₂ concentrations from two studies of similar LIB cells can be different if one study measured the vented gas in an inert environment, while in the other study, the gases were ignited, and the products after combustion were quantified. In the present review, published research related to gassing from LIB cells is scrutinised, and we try to evaluate the understanding gained while identifying inconsistencies in test methodologies as well as pointing out weaknesses in the reviewed studies and recommending further research to strengthen scientific knowledge.

1.2. Brief Overview on LIB and Gassing

An LIB consists of solid and liquid material that is ideally stable over time but instead undergoes a continuous but slow degradation process referred to as “ageing” [9–11]. During ageing, the battery constituents undergo unwanted decomposition and chemical reactions, forming modest amounts of gas phase compounds that may eventually make the battery cell become pressurised and contribute to the irreversible swelling of the cell. Chemical reactions and decomposition are quite often temperature-dependent [11], which is the reason why batteries age faster in a warmer environment. The composition of gases in an aged battery has not been investigated to any significant extent, and it appears to often be assumed that the composition and hazardous properties of gas resulting from the ageing process are the same, or similar, to those occurring due to thermal, mechanical, and electrical abuse. This may explain the lack of published studies on LIB gassing due to ageing. Gas composition from aged LIB cells under abuse conditions has been shown to be different from that of fresh cells, which indicates that there may be some “chemical pre-processing” in an LIB cell during normal operation. This calls for further work on gas formation in LIB cells during ageing. The current state of knowledge on gas composition due to ageing is reviewed in Section 2 of the present work.

When a battery is exposed to abuse (thermal, electrical, or mechanical, as outlined by Ghiji et al. [6]), the rate of the internal chemical processing accelerates as a result of one or several factors: increased temperature; the positive and negative battery electrodes are in direct contact with each other and with the electrolyte; or the electrode is in direct

contact with external reactive materials like oxygen and humidity in the air. The resulting chemical reactions are exothermic and produce heat that further increases the temperature of the system and accelerates the chemical processing and breakdown of materials. In this work, we call the self-heating process of the battery a “thermal event”. Depending on the balance between heat production and the efficiency of heat dissipation to the surroundings, the thermal event may stop and self-quench as the battery cools down. However, if the generated heat cannot be dissipated at a sufficient rate, it may result in a “thermal runaway” (TR), which is a more severe thermal event where an uncontrolled sequence of exothermic chemical reactions results in rapid temperature increase and extensive gas production [3]. TR results in the extensive venting of the battery, potentially in an uncontrolled manner, and the ejected gases may ignite and cause a fire. In addition to gases, a venting battery will also expel solid material and fine particulate matter [12,13], which has implications for the health and overall risk assessment of LIB venting and fires.

The processes involved in a thermal event in an LIB have been explained comprehensively by, among others, Wang et al. [3] and Qiu et al. [5] and are just briefly outlined here. As the battery cell is overheated, the first step towards TR is commonly the breakdown of the SEI layer on the graphite anode, which occurs at about 90–120 °C. At these temperatures, volatile electrolyte components such as dimethyl carbonate (DMC) can start to evaporate and decompose. The SEI protects the graphite from direct contact with the electrolyte and is formed on the anode surface during cell formation. In the process, there is a release of stable lithium (Li) compounds, small hydrocarbons like ethylene (C₂H₄), carbon dioxide (CO₂) and oxygen molecules (O₂). As the SEI breaks down, the anode material can react with the electrolyte, and the electrolyte can also react with Li to form lithium carbonate (LiCO₃) and C₂–C₃ hydrocarbons. When the separator melts at temperatures in the range of 135–165 °C, the electrical contact between the anode and cathode can result in an ISC and, thereby, self-discharge. As the temperature increases further in the range of 150–310 °C, the cathode begins to decompose and release oxygen. As the temperature reaches 200 °C and higher, all components of the electrolyte start to decompose exothermically, upon which CO₂, hydrogen fluoride (HF), hydrocarbons, and various fluorinated hydrocarbon gases are formed. Oxygen released from the cathode material can react with the solvent or other hydrocarbon components, driving the production of final oxidation products CO₂ and H₂O, but also the intermediate hydrocarbons, CO and H₂, depending on the availability of O₂. For further details, we refer to the review by Wang et al. [3] and the references therein.

Gaseous species formed in LIB cells include CO₂, CO, O₂, H₂, CH₄, C₂H₄, and C₂H₆, higher hydrocarbons, including the carbonates in the electrolyte, and fluorinated compounds like HF and fluoromethane (CH₃F) [3]. These compounds are found in gases from aged batteries, from batteries that vented after a minor thermal event, but also from severe venting during a TR. The relative amount of these compounds varies notably depending on the degree of chemical processing, which is attributed to the effects of a range of parameters, such as SOC at the time of gas formation and venting, battery chemistry, age, battery geometry, and potentially a range of other factors. The composition of the gas mixture, as well as the total amount of gases being vented from a battery during TR, will affect the flammability and toxicity and, thus, the severity of a potential incident. To increase understanding of these factors, LIB cells have, to a large extent, been studied in different experimental configurations where a thermal event is triggered, and the gases and heat release are determined. An overview of the experimental methods is given in Section 3 of the present work. In addition to the research on complete LIB cells, studies have also been conducted on battery components, one by one and in combination. These studies provide a fundamental understanding of the processes occurring in an LIB, as explained in Section 4.

1.3. Scope of the Present Review

Experimental studies of gassing from LIBs can be divided into two categories: vent gas studies and combustion product studies. In vent gas studies, the composition of the gas mixture escaping the chemical system is investigated, while in combustion product

studies, gases are sampled during or after a fire. From the aspect of the fundamental understanding of the TR and the chemical processing in the battery, vent gas studies are of high importance since they reveal the types and amounts of gases being produced in the LIB. Hazard assessments require information about gas composition, gas volume, and, ideally, also initial conditions with respect to gas temperature, characterising a gas vent from an LIB. Using this information, the severity of an incident, for example, whether a fire is likely to start, can be evaluated. The interpretation of the mentioned data typically also requires background information about the LIB: cathode chemistry, electrolyte composition, SOC, cell geometry (pouch, prismatic, cylinder), cell capacity as well as, in the case of a battery pack, the cell configuration and connections inside the battery.

Experimental work often provides information on the vent gas composition target at one or several of the following levels:

1. Solvent studies address the properties of relevant carbonates, including their evaporation and decomposition characteristics. The value of these studies is understanding the stability of the different carbonates and their decomposition products. It is, however, important to note that properties like decomposition temperature may be altered depending on the chemical environment, and the results from solvent studies cannot be fully extrapolated to LIB conditions.
2. Electrolyte studies of mixtures of carbonate and lithium salt are needed to understand how the presence of salt affects the evaporation and decomposition of the electrolyte, which is important in the early steps of a thermal event.
3. The electrode material, in combination with the electrolyte in a half-cell, is studied to gain an understanding of the chemical interactions between these materials.
4. The composition of gases in a cell during regular use conditions provides information about the chemical environment at the onset of a thermal event, which is particularly important for aged cells.
5. The total volume of gas from LIB cells subject to internal failure caused by abuse conditions is important for an assessment of fire and explosion risks.
6. The composition of gas vented from LIB cells subject to internal failure caused by abuse conditions provides information that allows the assessment of flammability and toxicity.
7. Thermal evolution in LIB cells subject to failure caused by abuse conditions, including information about onset temperature for TR, is important to assess the risk of spread of TR in a battery pack.
8. Studies on several cells or a full pack are necessary to understand how findings from single-cell studies can be extrapolated to larger systems.
9. The composition of combustion gases from LIB fires provides information that enables risk assessment.

Unfortunately, it is practically impossible to study spontaneous internal failures not triggered by abuse in experimental settings since the occurrence of these events is very rare. A forced TR using different types of abuse triggers is currently the only available path, and it has been demonstrated that the choice of the TR method influences the cell response [14], depending on the accelerating effect of the chosen method as well as the difference in the localised vs. general activation of the TR reactions inside the cell.

In the present work, the main focus is on gas composition in vented gases from LIB cells, point 6 in the list above, but also the literature on points 2–4 is reviewed since they can provide supporting data in understanding the events.

Studies of assembled battery configurations at the pack level and complete devices such as electric vehicles (EVs) are dominated by combustion studies. While measurements of gases from evolved battery fires have been extensively reported in the literature, they are found to be of less importance for understanding the onset and different stages of an event and knowledge with the potential to mitigate or prevent future events. The correlation between the reaction mechanism and the related reaction products created by the test methods and the safety events and associated cell responses under real conditions

in the field are largely missing. To be able to assess the relevance of the experimental test results compared to different types of field responses to safety events, it is also critical to understand the level of abuse, e.g., how much external energy was added to the component/cell/battery assembly to produce the observed thermal event, since this is a measure of the experimental acceleration factor used. Experimental instrumentation can also be an accelerator if it results in uncharacteristic modifications of the test object; for example, it introduces alternative vent paths or oxygen ingress into the battery, enabling more intense fire events than would have been possible in the original battery design.

The motivation behind other published reviews on LIB vent gas research is commonly one of the following: (1) to understand the underlying mechanisms and ascertain their dependence on various parameters (SOC, energy density, age, etc.); (2) the quantification of factors related to heat release and gas release; and (3) the investigation of prevention and mitigation strategies. Ideally, the research data and conclusions should provide information that enables the development of successful safety strategies and the design of safe LIBs. To achieve these goals, the research needs to be designed in such a way that the produced data are not a mere investigation of a particular battery but that the results can be extrapolated to other conditions or, together with other datasets, provide valuable generalised information. The value of some studies is, unfortunately, hampered by a limited scope. In the review presented here, one important aim is to identify datasets of value for the overall understanding and quantification of vent gases from LIBs.

The present review summarises the current understanding of gas composition from different stages in the gassing process from LIBs, starting at ageing and the slow formation of gases, via mild thermal events to TR with extensive gas production. The published data are used to map gas quantity and composition from LIBs undergoing venting, with or without a TR, and to identify gaps in understanding and the need for further research.

The main discussion in this review is on major constituent gases (>1% of total gas volume) that contribute to the fire and explosion risks. CO, H₂, and hydrocarbons are the main drivers of combustion in the present context, and CO₂ plays an important role since it has an inerting effect. In addition, the potential presence of O₂ in the vented gases is discussed based on available knowledge and data. In this review, papers that make an exhaustive characterisation of minor components of importance to the toxicity of the off-gases are included, but no in-depth discussion on toxicity is provided. Hydrogen fluoride (HF) is, by some, considered an important toxic compound vented from LIBs, but it is not a player in a fire or explosion risk scenario and, therefore, has not been reviewed in an exhaustive manner. However, the experimental challenges related to the detection of HF and other fluorinated compounds are relevant for the discussion on experimental methodologies for vent gas studies and are discussed in this particular context.

2. Gassing During Normal Use Conditions

Gases are formed in an LIB cell already during the first cycling and the formation of the SEI layer when small amounts of CO₂ and hydrocarbons like C₂H₄ or CH₄ have been detected [15,16]. During regular use, a small amount of gas is produced by slow material degradation, and this is part of the ageing process. The ageing of the battery cell and the formation of gases is an important topic in terms of quality and durability [17], but it is also relevant in investigations of malfunctioning batteries since, as shown by Essl et al. [9], ageing affects the gas composition of a battery cell that experiences TR failure.

Several studies have determined the pressure or volume increase caused by cycling under different conditions within the normal usage range, but data are scarce regarding the composition of the gas mixture.

Scharf et al. [16] investigated gas production during cell formation in their effort to gain insights into the formation process and its temperature dependence in the range from 10 to 40 °C. The production of C₂H₄ is found to be important during cell formation with the build-up of the SEI layer when significant amounts of H₂ and CO are formed, with minor amounts (<1%) of O₂ and CO₂. Gases produced during formation are removed in

the final steps of cell production and will, thus, not affect the future of the cell, but the same type of chemical reactions continue at a low rate during the lifetime of the cell, and the same products can be expected to be formed.

Raghibi et al. [15] used gas chromatography (GC) to investigate both the composition and quantity of gases formed during the normal cycling of NMC pouch cells with four different carbonate solvents and two Li-salts to reveal the potential connection between gas formation and the choice of electrolyte components. They stress that the amount of free gas in the cell depends both on the volume of gas formed and the degree to which this gas is dissolved in the liquid electrolyte. While most gases are formed by the decomposition of the cyclic ethylene carbonate (EC) during cell formation and the establishment of the SEI layer, it was also found that the shorter and more volatile linear dimethyl carbonate (DMC) produces more gas than ethyl methyl carbonate (EMC). The solubility of the gases is largely affected by Li-salts, with lithium bis(fluorosulfonyl)imide-(LiFSI)-containing electrolytes taking up more gas than the ones with the more common lithium hexafluorophosphate (LiPF_6) [15].

An important finding by Raghibi et al. [15] was that the reactive gases O_2 and H_2 are dissolved in the electrolyte to a lesser extent than the more stable CO_2 and hydrocarbons. This means that the reactive gases are, to a larger extent, available to react with additional gases formed during a thermal event and, thus, contribute to the chemical processing and potential production of flammable gases in a thermal event.

From the information outlined above, it is concluded that the electrolyte constituents play a role in creating a reactive environment with O_2 available in a battery during normal use. The presence of O_2 and H_2 in aged cells may contribute to accelerated reactions if the battery cell is subject to abuse or other failure modes leading to a TR, and hence, different safety responses from aged compared to new batteries during testing conditions as well as in field incidents. Studies where thermal events were triggered in aged LIBs show that aged cells can have a different thermal response and different gas composition compared to fresh cells [9,18]. This calls for further studies on gas composition in cells during regular use conditions.

3. Methods Used in Li-Ion Battery Gassing Studies

Thermal events resulting in excess gassing from a LIB often originate from the spontaneous manifestation of internal cell faults but can also be a result of electrical, mechanical and thermal abuse, as explained in the review by Feng et al. [19]. Electrical abuse can be an external short circuit, overcharge, over-discharge or over-current. Mechanical abuse is crushing or the deformation or penetration of the battery cell with a nail or a rod. Thermal abuse can be either local with the initial heating of parts of a cell or involve the external heating of the full cell. The forced thermal runaway of a cell by applying some form of external abuse condition is currently the only feasible way to study battery thermal events. The most common method in experimental studies is the slow ($0.5\text{--}2.0\text{ }^\circ\text{C}/\text{min}$) uniform heating of the LIB cell, which allows for a well-controlled experiment. However, the slow homogenous heating of the entire cell is not the most relevant TR trigger condition when attempting to replicate real-world failure modes, which are characterised by rapid localised heating at and around the area of the cell directly affected by the fault or abuse condition, followed by the heterogeneous heating of the rest of the cell and potentially of neighbouring cells in a battery in the case of a propagating thermal event.

As recently outlined in a work by Willstrand et al. [14], the test methodology chosen can affect the results since the dynamics of the cell response depend on the TR trigger. Thus, a deeper understanding of the different test methods to assess their applicability to evaluate different aspects of TR behaviour is needed. In the following subsection, first, the controlled external heating experiments are outlined since they constitute a large part of the gassing studies and are performed using a range of dedicated methods. This is followed by information about rapid trigger methods like nail penetration. Finally, the common gas analysis methods are outlined.

The aim is not to make exhaustive reviews of the methods but to provide the reader with some overview information and to become acquainted with the pros and cons of methods commonly used and reported in the literature. An understanding of experimental methods and how the choice of methods may affect the cell's TR response will aid in the evaluation of some of the experimental studies presented in later sections.

3.1. Controlled Heating Experiments

Uniform and controlled heating is the most common way to trigger thermal events and TR in laboratory studies. The advantages are that the experiment is well-controlled, and it is relatively easy to perform and monitor thermal events. The experimental setups can broadly be divided into three categories: calorimeters, direct-contact uniform heating and radiative heating. These methods have a moderate heating rate in common, often in the range of 0.5 to 2.0 °C/min and starting from ambient. This means that it takes hours to bring the LIB cell or material to temperatures where the onset of electrolyte evaporation and chemical breakdown occur. With these methods, the tested systems can be expected to have enough time to equilibrate and reach homogeneous temperatures until the start of rapid self-heating (90–180 °C). Common methods are listed in Table 1, and some more details are provided in the text.

Table 1. Methods for controlled heating experiments.

Abbrev.	Name	Advantage
DSC	Differential scanning calorimetry	Provide an understanding of thermal properties in LIB cell materials
ARC	Accelerating rate calorimetry	Allow close monitoring of the thermal event
TC	Tewarson calorimeter	Can be applied to fire studies
CC	Cone calorimeter	Can be applied to fire studies
HP	Heating plates and heating tape giving thermal contact	Can be designed to fit any size and shape of LIB cell and easily combined with various diagnostics and methodologies
	Radiative heating	-

Calorimeters are used to measure the heat released or absorbed during a chemical reaction or physical change. It works by allowing the transfer of heat from the tested system to a surrounding medium where the heat can be measured. There are various types of calorimeters, each designed for specific applications and with different levels of precision. The technique for accelerating the rate of calorimetry (ARC) has been widely applied in studies of the LIB electrolyte [20–22] as well as on LIB cells [23–27]. The ARC increases the temperature in discrete steps, waits for the thermal transient to decay and then monitors the temperature of the cell for a fixed time duration. If the cell temperature does not exceed a threshold value, the process is repeated at the next temperature increment. If the cell temperature increases at a rate equal to or above the threshold value, the ARC switches to the exothermic mode, during which it closely matches the cell temperature, thus maintaining an adiabatic state. Differential scanning calorimetry (DSC) is a thermoanalytical technique used to measure the difference in heat flow into a substance and a reference material as a function of temperature or time. DSC has been used to correlate TR events to the thermal stability of battery components and is often used in combination with ARC [20,21,23,28]. Tewarson calorimeters are commonly used for measuring the specific heat capacity of materials at high temperatures and have been applied to measure the heat release rate in fire studies of LIB material [29]. A cone calorimeter can be used to measure the heat release rate, smoke production rate, and other key parameters during the combustion of materials, and is, therefore, particularly valuable in fire research [30–32]. Cone calorimeters can be

combined with combustion test chambers to measure the heat release rate, as performed by Huang et al. [33].

Uniform and controlled heating can be performed on large battery cells with electrically heated plates, as exemplified by Willstrand et al. [14]. Liu et al. [34] used a heating plate to simulate TR, which resulted in a fire later on. Lammer et al. [35] used an electric resistance furnace to heat the LIB cell. There were feedthroughs for thermocouples, an inert gas inlet, and a gas sampling station.

3.2. Rapid Trigger Methods

In safety testing, TR is often a result of a rapidly triggered ISC by applying external thermal or mechanical abuse (localised heating, crushing, or penetration). These events can be studied in laboratory environments, but the time scales are much shorter than for the slow thermal methods described above, and therefore, it is challenging to obtain reliable information on the dynamics.

Mechanical trigger methods have been explained in detail by Liu et al. [36], who explored compression, deformation, and penetration abuse. A significant difference was found between a local trigger, such as nail penetration, compared to an abuse condition applied uniformly to the cell, like the thermal methods, as presented in a previous section, and the consequences will vary depending on where in the cell and how the trigger is applied.

Willstrand et al. [14] used three methods for the rapid triggering of TR: a patented heater to locally increase the temperature to 50 °C/s; a nail penetration device; and an overcharge. The tests were performed in both open and closed setups to analyse the effect of ambient air on gas composition. It was shown that the choice of the trigger method did not significantly affect the amount or composition of gases produced.

3.3. Gas Analysis Methods

Gas analysis can be performed continuously to monitor the evolution of gases during venting, or on the total gas volume after venting, is finalised. Continuous monitoring presents a challenge in the experimental design since either optical access or gas sampling must be performed in the vicinity of the vent. The main methods used in the studies reported in this review are listed in Table 2, and the purpose of the present section is to outline the advantages and disadvantages of the methods.

Table 2. Gas analysis methods.

Abbrev.	Name	Advantage
FTIR	Fourier Transform Infrared	Non-intrusive detection of a wide range of components
GC	Gas chromatography	Allow the detection and quantification of a wide range of components
GC-MS	GC–mass spectrometry	In addition to GC capability, it includes structural information and is more sensitive
GC-FID	GC–Flame Ionisation Detector	Fast and simple analysis of organic components
OEMS	Online Electrochemical Mass Spectrometry	Allow the online monitoring of selected gases
Raman	-	Non-intrusive detection of a wide range of components
NMR	Nuclear Magnetic Resonance	Give detailed structural information and allow to follow kinetic reactions
IC	Ion Chromatography	Allow the quantification of inorganic F
	Wash bottle	Collection of HF and other inorganic F components

Many of the gas analysis techniques are quite versatile and can be used to detect a wide range of compounds. However, they need to be calibrated to accurately quantify compounds, which means that the researcher needs to know beforehand which compounds to target. It is common to target the components that are expected to occur in large fractions: CO, CO₂, H₂, CH₄, and C₂-hydrocarbons. Under some conditions, the electrolyte solvent (i.e., carbonates) is also present in significant amounts. This gives important information on the extent of chemical pre-processing that occurs inside the cell before venting since the carbonates are essentially unprocessed battery electrolyte solvent.

Fourier Transform Infrared Spectroscopy (FTIR) is used to identify organic and inorganic materials based on their absorption of infrared light [37]. FTIR works by measuring the interaction of infrared radiation with a sample, providing detailed information about its chemical composition and structure. FTIR has the advantage of being non-intrusive, and it can detect almost all compounds of relevance to LIB vent gases. Depending on the type of FTIR apparatus, it can provide spectra of all compounds in the gas mixture, which allows the identification of minor compounds with, for example, high toxicity.

Gas chromatography (GC) is a widely used analytical technique employed to separate, identify, and quantify the components of complex mixtures [38]. In GC, a gaseous sample is injected into a chromatographic column, where it interacts with a stationary phase. The components of the sample then separate based on their affinity for the stationary phase and their volatility. The species are identified by a detector, often a mass spectrometer when they reach the end of the column. Gas chromatography–mass spectrometry (GC-MS) combines the separation power of gas chromatography with the identification capabilities of mass spectrometry. In GC-MS, the effluent from the GC column is introduced into the mass spectrometer, where the separated components are ionised and fragmented. The resulting mass spectra provide information about the molecular weight and structure of the individual components, allowing for accurate identification and characterisation. LIB vent gas studies employing GC-MS commonly target a range of expected compounds, including CO, CO₂, H₂, small hydrocarbons, and O₂. For accurate quantification, the procedure is calibrated to the target compounds which are identified. There is also a possibility to identify minor compounds in the analysis where the targets are not predefined.

To find the absolute amount of gas formed during a TR experiment, it is common to measure pressure in a known volume and use the ideal gas law to calculate moles of gas [26,39–42].

GC is not suitable for the measurement of hydrogen fluoride (HF) and other fluorinated compounds, but instead, FTIR or wash bottle techniques are used. The wash bottle is conveniently combined with Ion Chromatography (IC) for the quantification of the total amount of fluoride ions in the sample.

The wash bottle technique is a practical method for determining the presence and concentration of HF but has some drawbacks. In a typical setup, a wash bottle containing a solution, often of a calcium salt-like calcium carbonate or calcium sulphate, is used. When HF vapour is present, it reacts with the calcium solution to form a white precipitate of calcium fluoride (CaF₂). The formation of this precipitate is a direct indication of the presence of HF. Quantitative determination can be achieved by measuring the amount of precipitate formed and relating it to the concentration of HF using stoichiometric calculations. A significant drawback of this method is that if there are other inorganic fluorinated compounds present in the environment that can decompose, hydrolyse, or otherwise react to release HF or fluoride ions, it leads to an overestimation of HF by this method.

4. Component Studies

While the battery cell is a complex system with multiple constituents that may interact in different ways depending on the temperature and chemical environment, it can be advantageous to increase our understanding of the interaction between a few key components. As an example, many studies have investigated only the electrolyte, its decomposition, heat formation, and product formation at various temperatures. Other studies have been

performed on half cells to reveal interactions between the electrolyte and the positive or negative electrode material during thermal events. In the following, extensive but contradictory literature on electrolyte decomposition is reviewed, followed by the more limited literature where cathodes are included. As outlined by, for example, Wang et al. [3], the anode material also plays an important role in the thermal events in LIB. However, the effects of different anode materials on gas composition have not been studied in a systematic way and are, therefore, not reviewed in the present work. For an outline of processes related to the anode, we refer to the work by Du Pasquier et al. [43].

4.1. Electrolyte Decomposition

In this section, the literature on gases originating from only the electrolyte at different conditions is reviewed, including the effect of different carbonates and accelerated degradation due to the Li-salt.

The common electrolytes consist of a mixture of two to four carbonates (commonly including EC, DEC, DMC, EMC, and PC) and Li-salt, which is most often LiPF_6 . The carbonates are flammable and toxic, and the salt and its potential products have varying degrees of toxicity. Understanding electrolyte decomposition is valuable for the evaluation of risks in case of electrolyte leakage during the handling of the battery or pure electrolyte evaporation as a result of mild mechanical abuse where the other cell components are not affected; for example, this includes some of the cases studied by Diaz et al. [44]. However, the full battery system includes several other parts that become chemically active at different temperatures, particularly the electrodes. This means that the reactions and products that are found in studies of electrolyte decomposition alone cannot be expected to mimic conditions in the battery cell. However, a fundamental understanding of the reactivity of the electrolyte is still valuable since it is part of the whole reaction scheme during a thermal event. Electrolyte decomposition may also have implications for the long-term storage of batteries. A study by Fernandes et al. [28] compared the results of electrolyte decomposition investigations to results from the overcharge abuse test of battery cells. They found a qualitative agreement, and from this, they concluded that part of gassing during the overcharge of LIB is a result of electrolyte decomposition. This was not explicitly discussed or shown in any of the other studies but implicitly assumed.

The electrolyte decomposition studies reviewed here include a few [45–49] at conditions similar to normal use or storage conditions at elevated temperatures but below the onset temperature of thermal events. These studies provide insights into the relevance of gassing as part of the ageing of batteries. The studies that target the response of the electrolyte to a thermal event commonly operate by ramping the temperature from ambient up to at least 300 °C, and sometimes as high as 650 °C, while monitoring the heat production and/or the composition of reaction products formed in the liquid or in the gas phase.

At a minimum, the following processes can occur when an electrolyte is heated beyond its point of stability:

1. Chemical reactions in the liquid, producing liquid or gaseous products, or solid residues are formed that may dissolve in the liquid or deposit.
2. The evaporation of the electrolyte.
3. The reaction between gas phase compounds and liquid, potentially generating new gas phase constituents.
4. The thermal decomposition of the electrolyte.
5. Further chemical reactions in the gas phase, between the gas phase electrolyte, electrolyte decomposition products, and gas phase products from the reaction with Li-salt.

Experimental studies on electrolytes may target one or several of the aspects listed above, and discrepancies in the literature can sometimes be attributed to a lack of understanding of which phenomena are studied and the interactions between chemical compounds in different phases.

An overview of the studies related to thermal decomposition and subsequent reactions of the electrolyte is presented in chronological order in Table 3. Here, the discussion of

the literature relates to several issues for which different studies have given different indications: (1) the presence of a direct chemical interaction between the solvent and Li-salt; (2) the effect of water (and other impurities); (3) the extent of formation of fluorinated products; and (4) the presence of an autocatalytic mechanism that contributes to carbonate decomposition. The presence of water or other impurities in the system is a very important factor since it potentially impacts reaction paths and kinetics and thus can lead to several of the noted discrepancies. In addition to these difficulties, mass spectroscopic methods have been reported to potentially have sampling problems with reactive species undergoing side reactions and transforming in the sampling lines or analyser [50], which may affect the results and conclusions.

Table 3. Studies of electrolyte decomposition. All cases contain LiPF₆, if not otherwise stated. Abbreviations in the methods column can be found in Tables 1 and 2.

Study	Electrolyte	Methods	Target Properties
Botte et al. (2001) [51]	EC, DMC	DSC	Thermal stability of liquid, 50–350 °C
Ravdel et al. (2003) [45]	EC, DEC, DMC, EMC	DSC, GC-MS, NMR	Characterisation of decomposition products in liquid and gas phases during heat ramping at 50–300 °C and storage at 85 °C
Gnanaraj et al. (2003) [20]	EC, DEC, DMC	DSC, ARC, NMR	Thermal stability of liquid electrolyte from 40 to 350 °C, and product characterisation in the liquid and gas phase
Campion et al. (2005) [48]	EC, DMC, DEC	NMR, GC-MS, SEC	Mechanistic study of liquid and its gas phase products after storage at 85–100 °C for 1–4 weeks
Kawamura et al. (2006) [47]	EC, DEC, DMC	H ₂ O after storage	Rate equation from the reaction of Li-salt with water at 0–30 °C for up to 50 h
Wang et al. (2006) [52]	EC, DEC, DMC, PC	C80-calorimeter	Thermal behaviour during heating to 300 °C, with and without Li-salt
Yang et al. (2006) [46]	EC, EMC, DMC, PC	TGA, FTIR	Characterisation of decomposition products in liquid and gas phases during heat ramping at 50–300 °C and storage at 85 °C
Wilken et al. (2012) [53]	EC, DMC	TGA, FTIR	Characterisation of gas phase products upon the heating of the electrolyte to 300 °C
Lekgoathi et al. (2013) [54]	Only LiPF ₆	TG, FTIR	Mechanistic study of decomposition of Li-salt with and without the presence of water
Wilken et al. (2013) [49]	EC, DMC	Raman, NMR	Gas and liquid characterisation during storage at 85 °C for up to 160 h
Okamoto (2013) [55]	EC	Ab initio calc.	Establish thermal decomposition mechanism in liquid
Yamaki et al. (2015) [56]	PC, EC, DMC	DSC	Determine rate equation during heat ramping to 450 °C
Tebbe et al. (2015) [57]	EC, various Li-Salts: LiPF ₆ , LiPOF ₄ , LiAsF ₆	DFT calc., liquid phase.	Investigate HF formation from Li-salt decomposition in EC
Lamb et al. (2015) [22]	DMC, EMC, EC, DEC	ARC	Gas production from the thermal decomposition of the solvents with and without Li-salt
Bertilsson et al. (2017) [58]	LP71, EC, PC, DEC, DMC, EA	TG, FTIR.	Emission of gases from solvents and electrolytes are characterised to 650 °C
Solchenbach et al. (2018) [50]	EC	TGA, OEMS	Thermal and oxidative decomposition of LiPF ₆ with and without EC to 350 °C
Fernandes et al. (2019) [28]	DMC, EMC, EC, PC	DSC	Thermal degradation of solvents to 300 °C
Liao et al. (2020) [59]	EC, DMC	Auto-clave, GC-MS	Gas emissions from the electrolyte is characterised for heating to 300 °C

An early disagreement in the literature was whether there are direct interactions between the Li-salt or its products with the carbonate solvents. According to the study by Botte et al. [51], there was no evidence of direct chemical interactions between the solvent mixture and LiPF_6 at elevated temperatures, but the presence of the salt lowered the onset temperature for carbonate decomposition. Ravdel et al. [45] used a similar method as Botte et al. but also analysed chemical products and showed that salt decomposes to form phosphorus pentafluoride (PF_5) and, indeed, reacts with carbonates and other hydrocarbons. At about the same time, Gnanaraj et al. [20] showed that fluorinated hydrocarbons were formed from the interaction of Li-salt with hydrocarbons. However, Yang et al. [46] and Wilken et al. [53] reported that no signs of decomposition products from carbonates were found in their studies.

The presence of water impurities is a factor that has been proposed to strongly affect the products formed in electrolyte decomposition. It is well known that LiPF_6 forms different products with or without water present [54], and the consequences for the electrolyte system have been studied by Kawamura et al. [47] and Campion et al. [48]. At storage conditions, the latter identified fluorophosphate products in the presence of water.

Another controversy concerns the potential catalytic effect of Li-salt and/or its decomposition products on carbonate degeneration, i.e., if compounds like, for example, phosphoryl fluoride (POF_3) regenerate to further degrade the carbonates.

There are conflicting reports on the importance of and the relative amounts of the reaction products POF_3 and PF_5 related to salt decomposition. While some studies report high concentrations, others report low occurrence or even the absence of these species. In some cases, it has been confirmed that the highly reactive fluorinated species were consumed by side reactions in the apparatus, thus misrepresenting the actual amounts formed [50]. Due to their high reactivity, the accurate quantification of species like HF, PF_5 and POF_3 is challenging and demands precise control of the test setup to avoid the method errors caused by unintended and uncontrolled side reactions with the surfaces in the test equipment.

4.2. Cathode Decomposition

The phrase “battery chemistry” commonly refers to the composition of the cathode material. The chemistry of the LIB cathode varies in performance, cost, and safety due to differences in material composition. The decomposition of the cathode material is a key factor in the internal chemical processing of the battery and the resulting reactivity of vent gases due to the production of O_2 that can initiate or accelerate oxidation processes. The layered oxide cathode materials, such as nickel–manganese–cobalt dioxide (NMC) and nickel–cobalt–aluminium dioxide (NCA), currently used in electric vehicle batteries decompose above the threshold temperatures in the range from about 150 °C to above 300 °C. Cathode material with a high nickel content have a higher specific capacity but poor thermal stability. The onset temperature is mainly a property of the cathode material, but it is, to some extent, also influenced by the chemical (electrolyte composition) and electrical environment. The decomposition mechanisms for different cathode materials and experimental studies reporting on the dependence of the electrolyte composition are comprehensively reviewed by Wang et al. [3] and will not be repeated here. For the analysis in the present work, however, it is relevant to know the thermal reactivity of each cathode material. Lithium–iron phosphate (LFP) cathodes likely produce very little or no oxygen at all, which is represented by the half-cell (cathode) chemistry with the lowest risk going into TR. The reactivity of the oxygen-producing cathodes is ranked as follows: $\text{LCO} > \text{NCA} > \text{NCM} > \text{LMO}$, where LCO and LMO are lithium–cobalt dioxide and lithium–manganese dioxide spinel, respectively. However, it is important to mention that Li-salt decomposition, the breakdown of the SEI and the chemical reactions between the graphite anode and the electrolyte start at lower temperatures than the cathode reactions, typically around 90 °C, but it can be lower [3,19,60]. Consequently, the initial thermal release and gassing are driven by the anode’s side, independently of the cathode. The heat released

from the anode-electrolyte reactions is sufficient to raise the cell temperature to levels where the oxygen-producing cathode materials start to break down.

An important fate of the O_2 released from the cathode is their subsequent reaction with the lithiated anode to produce heat. Oxygen can also react with the carbonates in the electrolyte, producing carbon oxides and hydrocarbons. No significant fraction of O_2 has been reported from studies of vented gases. There are at least two plausible explanations for the low presence of the vented gases: either O_2 is essentially consumed in the chemical reactions inside the cell before the venting occurs, or simply O_2 has not been a target compound for investigation in most studies, and, hence, not measured and reported.

5. Experimental Studies of Vent Gases

This section reviews the literature on the composition of vented gases from LIB cells. The criteria for the selection of publications for the analysis are (1) the vented gases have not been ignited, and (2) chemical composition has been determined, including at least the main gases CO , CO_2 , and H_2 . This means that, in this review paper, the focus is on vent gas compositions resulting from the chemical processing in the battery cells only, not gases formed in the combustion reactions outside the cell. A common way to ensure that no combustion occurs is to vent the cells into an inert environment of nitrogen gas. Some studies present results from experiments with and without combustion, and a few are ambiguous on whether combustion took place or not. The current mix of different types of studies found in the literature makes a direct comparison between results challenging and consequent conclusions about the correlation of gas emissions, both in terms of the species detected and their relative amounts, in vent gases as well as the influence of specific test parameters, e.g., SOC, potentially misleading, as will be shown below.

To enable the risk assessment of LIBs, the amount and composition of vent gases must be known, and the aim of this review is to extract information that will allow the fact-based prediction of vent gases, like battery chemistry, SOC, SOH, conditions of thermal event, etc. The studies vary a lot with regard to details reported on the LIB cell properties, the experimental conditions, and the results. Ideally, detailed information about cell material composition is provided; for example, electrode materials, the electrolyte composition, and the relative amounts of ingoing components. Regarding the experiments in detail, it is necessary to disclose the triggering method, as well as the SOC, at the start of the test. The results presented ideally include details on the temperature and heat release, mass loss from the LIB cell, the total amount of gas released, and gas composition including carbonates and fluorinated components. Many publications lack one or more parts of information, which makes it difficult to make comparisons between studies and to draw meaningful conclusions from the results.

The literature is collected and presented in tables in the following sections, with selected information of relevance to gas composition and other important aspects discussed further in the main text. Many studies present the total amount of gases formed, but some distinguish the so-called first vent. The first vent has some interesting characteristics that are outlined in the first subsection.

Comparatively few studies exist on venting following mechanical abuse, and these are found in the second subsection, with comments on the unique aspects related to mechanical abuse. The following sections summarise the works on electrical and thermal abuse, separated into individual tables for each cathode chemistry.

In the tables, the studied cell geometry is given, i.e., pouch, prismatic or cylindrical (Cyl.); among cylindrical cells, the numerals 18650, 26650, etc., refer to cell dimensions with the first two numbers indicating the diameter and the second two the length.

5.1. Characteristics of the First Vent

As outlined in the introduction, the thermal and chemical events resulting in gas release from LIBs proceed through a sequence of steps, with several occurring before the onset of what is defined as TR. The time during which the thermal process evolves varies de-

pending on LIB properties and the type and severity of the trigger event. Rapid penetration triggering leads to TR on very short timescales, with no preceding first vent [61]. But for experiments with a moderate heating rate [9,18,35,39,60–62] or gradual overcharging [4,61], gas formation occurs at low temperatures (80–160 °C) and the LIB cell vent and releases a limited volume of gas before the onset of uncontrolled TR. In some cases, the thermal events end with the first vent if the heat produced is effectively dissipated by surrounding materials and no heating above the TR threshold occurs. This has mainly been seen in LFP cells. Characteristic parameters in studies of first-vent events are summarised in Table 4. In all studies, the evolution of the thermal event and overall gas formation were monitored, but the determination of the gas composition of the first vent was not performed in all studies, as noted in the final column.

Table 4. Studies where the first vent is detected. If the number of cells is not specified in the second column, the testing was performed on the single-cell level.

	LIB Type	Heating Rate, °C/min	SOC	T at Time of Venting, °C	Gas Volume 1st Vent/Total Volume, %	Main Gaseous Components
Abbott et al. (2022) [63]	NMC Cyl. 18650	~20	5–100	157–125		CH ₄
Essl et al. (2021) [9]	NMC pouch, aged and fresh	2	100	~120	3–7	Gas composition of 1st vent not determined
Essl et al. (2020) [60]	NMC/LMO pouch	0.33–0.39	0, 30, 100	120–130	6	DEC, CO ₂ , H ₂ O
Essl et al. (2020) [61]	NMC pouch	2	100	121	4	Gas composition of 1st vent not determined
		-	OC	56		
	NMC prismatic	2	100	138	2	
		-	OC	66		
Fernandes et al. (2018) [4]	LFP Cyl. 26650	-	OC	80	0.7	DMC, EMC, C ₂ H ₄ , CO ₂ , CO
Lammer et al. (2018) [18]	NCA Cyl. 18650, aged cells	0.5	100	116–139	2–4	CO ₂ , C ₂ H ₂ (aged), (solvent not monitored)
Lammer et al. (2017) [35]	NCA Cyl. 18650	0.5	100	130	<3	CO ₂ (solvent not monitored)
Golubkov et al. (2015) [39]	LFP Cyl. 18650	2	0–100 and OC	130–150 but 80 at OC	-	Gas composition of 1st vent not determined
	NCA Cyl. 18650					
Golubkov et al. (2014) [62]	LFP Cyl. 18650	2	0–100 and OC	195	~7	Gas composition of 1st vent not determined
	NMC Cyl. 18650			168		
	LCO/NMC Cyl. 18650			149		

The first vent can be detected as a pressure or gas volume increase, depending on the nature of the experimental apparatus. While in the majority of gassing studies, gas sampling is performed after the main venting is completed (to determine the composition of all vented gases), there are only a few studies where the gas is continuously monitored using FTIR or regularly sampled to a GC-MS to enable the identification of the main constituents at different times in the venting process [4,18,35,60]. Since the gases in the first vent only comprise the gases formed by chemical processing in the cell before the onset of TR, it is likely that the composition may show similarities to gases formed during the ageing process of the cell, which are mentioned in Section 2 on gases produced during

regular use conditions. Thus, the first vent, as investigated in experimental studies, is potentially useful to increase understanding and predict gases that may be released at the battery's end-of-life. It can also provide complementary information when interpreting TR gas data from aged cells compared to pristine cells, as at least initially, the gases emitted at the onset of TR from aged cells are likely already present in the cell and not as direct products of the TR event itself.

Among the early detection strategies for LIB failure, the detection of CO, CO₂, and hydrocarbons has been suggested [64]. This strategy relies on the first vent preceding the main vent. Considering the experimental evidence that under some circumstances, the timescale of a TR in real applications is so short that a minor first vent is not detected, such detection strategies cannot be expected to reliably predict every LIB failure beforehand.

The experiments presented in Table 4 include the detection of a first vent, in which the gas volume is in the range of about 1–7% of the total gassing when the first vent was followed by a main vent. When overcharge was used to trigger TR, the venting occurred at or below 80 °C (measured at the surface of the cell). For temperature triggers, the first vent occurred at temperatures from 120 to 195 °C, with the NMC and NCA cells in the lower part of this range. The venting seemed to occur after the onset of SEI breakdown but before the separator melted completely and a full ISC developed in the cell. The internal production of O₂ gas from the cathode material was initiated at higher temperatures and was not expected in the first vents, which was confirmed in the studies that were instrumented to detect O₂ [35,60].

Regarding the identification of the composition of the first vent, all studies in which species detection was performed indicated CO₂ as an important or even dominant species. In the studies by Lammer et al. [18,35], CO₂ was identified as the main constituent, but caution has to be exercised in the interpretation of the results since the apparatus did not allow the detection of carbonate solvents. In the studies where solvents were monitored, carbonates were identified as the main components [4,60]. In the study of cells aged by cycling and storage, Lammer et al. [18] concluded that, while new cells mainly released CO₂ in the first vent, the aged cells had a large share of short hydrocarbons. Calendar-aged cells, in particular, had significant amounts of C₂H₂, which was attributed to gases formed during the ageing processes and not in direct response to the experimental trigger.

All studies except one were performed at 100% SOC or higher; however, the only study with a lower SOC indicated that the gas volume of the first vent was not dependent on SOC [60]. In conclusion, the first vent is most likely the outcome of mainly the evaporation of the solvent and decomposition of solvents with low decomposition temperatures.

5.2. Mechanical Abuse Tests

The mechanical abuse of EV batteries commonly involves the penetration or deformation of the battery pack by impact. There is a range of mechanical trigger methods used to test batteries, including compression, bending, and penetration [36]. Penetration is considered to be the most severe condition since it results in an instantaneous short circuit [19]. The immediate effects of mechanical abuse on the battery at the cell level can be short-circuiting between the positive and negative electrodes, resulting in the discharge of electrical energy and/or the rupturing of the cell wall, leading to the leakage of the electrolyte. Experimental studies have been conducted to investigate the thermal effects and energy release resulting from mechanical abuse. For an overview of thermal events after mechanical abuse, we refer to Feng et al. [19]. Examples of studies include rapid nail penetration [65], slow nail penetration [66] and abuse with a blunt rod [67]. In the present work, the focus is on the gas release, for which a few experimental studies with nail penetration as the trigger have been performed; these are summarised in Table 5.

The effect of penetration is the instantaneous release of gases [68], but in some laboratory tests, nail penetration has been shown to produce significantly smaller gas volumes compared to thermal abuse [44] or overcharge abuse [61]. However, as noted by Feng et al. [19], the common nail penetration tests show a lack of reproducibility. For nail

penetration setups, the sealing of the nail hole to avoid creating an alternative route for smoke and gas to escape is important as this may impact the internal chemical processing inside the cell and, hence, the gas composition. Sealing information is not always explicitly available in publications and may explain some of the differences in gas compositions between studies.

The most comprehensive gas composition investigations from mechanical abuse are the nail penetration tests by Essl et al. [61]. The tests were performed in a sealed chamber with a nitrogen atmosphere, with a 42CrMo4 nail of 3 mm diameter and cone angle of 60°, pushed 8 mm perpendicularly into the cell. It immediately triggered gas release and heating above 700 °C for both pouch and prismatic cells. There was only one venting, starting when the cell was penetrated. Approximately equal amounts of CO and CO₂ were emitted, based on volume, and close to the same volume of H₂, with smaller amounts of C₂H₄, CH₄, and H₂O. From the experimental description, it is not clear whether the venting occurred at the site of penetration or through the cell safety valve (for prismatic cells, pouch cells do not have a dedicated valve).

Table 5. Summary of experimental studies targeting vent gases from LIB after nail penetration. If the number of cells is not specified in the second column, the testing was performed on the single-cell level. Gases are listed in order of relative amount, starting with the most abundant compound.

Reference	Type	SOC, %	Electrolyte	Gases	Detection Method
Willstrand et al. (2023) [14]	NMC811 Prismatic	100	-	No gas detection for this trigger method	NDIR, FTIR
Hoelle et al. (2021) [69]	NMC111 Prismatic	100	EC:DMC:EMC 1:1:1	H ₂ , CO, CO ₂ , C ₂ H ₄ , CH ₄	GC
Essl et al. (2020) [61]	NMC622 Prismatic	100	EC:EMC 1:1	CO ₂ , H ₂ , CO, C ₂ H ₄ , CH ₄ , H ₂ O	FTIR, GC
	NMC622 Pouch		EC:DMC:EMC 2:3:3	CO ₂ , CO, H ₂ , C ₂ H ₄ , CH ₄ , H ₂ O	
Diaz et al. (2019) [44]	LCO Cyl. 18650	0–100	DEC, DMC, EC, EMC, PC	Carbonates, CO, HF, H ₂ O	FTIR online, IC
	NMC Cyl. 18650	100			
	LFP Cyl. 1865)	100			
	LCO Pouch	100			
Koch et al. (2018) [68]	NMC Pouch	100	-	Hydrocarbons/CO not specified	Gas sensors
Nedjalkov et al. (2016) [70]	NMC Pouch	>100	EMC, EC	Carbonates, CO, HF, CO ₂ , aromatic hydrocarbons	GC-MS, QMS, IC, offline analysis

The study by Diaz et al. [44] included mechanical and thermal (see Section 5.4) abuse mimicking recycling. For nail penetration, several cases showed no TR and only small amounts of gas production. It was concluded that, in these cases, electrolyte boiling off was the main gassing mechanism.

Koch et al. [68] did not give any details on the gas composition but focused on the dynamics of the event. Their aim was to investigate sensors, and the main takeaway for the context of the present review is that they detected gas venting instantaneously at penetration and modest surface temperature increases of about 150 to 300 °C.

Hoelle et al. [69] triggered 50 NMC and NCA cells of various capacities to the TR and found that the vented mass of gas had a linear dependence on cell capacity (Ah). They

only performed vent gas analysis on one NMC cell, which expelled about 30 vol% each of CO, CO₂ and H₂, respectively. The remaining 10% consisted of nearly equal amounts of the small hydrocarbons CH₄ and C₂H₄, which seems reasonable, considering that the electrolyte solvent contained similar amounts of methyl and ethyl groups.

Since the data are scarce, no significant trends were revealed, but one observation worth mentioning is that high temperatures were not reached in a majority of the tests. In the studies by Diaz et al. [44] and Nedjalkov et al. [70], carbonates and other large hydrocarbon fragments were detected, which indicated limited chemical processing in the battery before gas release. The gases are, for the most part, evaporated electrolytes. The study by Nedjalkov et al. [70] is the only one monitoring the release of heavier hydrocarbons, for example, benzene, and is relevant for the assessment of toxicity.

To understand the gas composition resulting from pure mechanical abuse, more experimental data are needed. The thermal effects of mechanical abuse were strongly dependent on the surroundings, for example, if the triggered cell was part of a battery pack. The mechanical abuse resulted in heat and gas release, and if the heat was not dissipated in the system, it triggered further heating and a full-scale TR event, which could propagate to surrounding cells in the case of a battery pack.

5.3. Electrical Abuse Tests

Overcharging is a severe form of abuse where the cell is forced into abnormal electrochemical conditions, and the battery is filled with excess electrical energy, which normally is not present and needs to be consumed. This triggers exothermic side reactions and results in the decomposition of cell materials, producing gases [19]. Minor overcharge conditions contribute to accelerated ageing, whereas more extensive overcharge leads to immediate cell failure. In EVs and battery installations with a well-designed battery management system, electrical abuse conditions, such as overcharge, overcurrent, and over-discharge, are effectively prevented, and for EVs, effective protection against electrical abuse is part of globally regulated safety requirements for homologation [71].

Over-discharge leads to the over-delithiation of the anode, the decomposition of the SEI, and the production of CO and CO₂ [19]. This will induce increased pressure, cell swelling, and a changed chemical environment in the cell, which will influence subsequent chemical processes.

Experimental studies reporting gas composition from overcharged LIB cells are summarised in Table 6. To the best of the authors' knowledge, there is no published report on gas composition analysis as a result of the over-discharge TR trigger.

Table 6. The summary of experimental studies targeting vent gases from Li-ion batteries after overcharge. If the number of cells is not specified in the second column, the testing was performed on single-cell level.

Reference	Cathode	SOC, %	Electrolyte	Gases	Detection Method
Willstrand et al. (2023) [14]	NMC811 Prismatic	128, 131	-		NDIR, FTIR
Cai et al. (2021) [64]	NMC Prismatic	213	-	CO ₂	Gas sensor
Yuan et al. (2015) [72]	NMC Prismatic	100–200	-	CO, CO ₂ , CH ₄ , C ₂ H ₄ , C ₂ H ₆	GC-MS
Essl et al. (2020) [61]	NMC622 Prismatic	147	EC:EMC 1:1	CO, CO ₂ , H ₂ , CH ₄ , C ₂ H ₄ , H ₂ O	FTIR, GC
	NMC622 Pouch	146	EC:DMC:EMC 2:3:3		

5.4. Thermal Abuse Tests

As mentioned in Section 3.1, slow and uniform heating is the most commonly used among the thermal abuse trigger methods, but there are a few studies that use localised and/or rapid heating. Appropriate information about the heating method and rate is given in the summary tables when such data are provided in the original publication.

This section is organised with subsections for different cathode chemistries, and the vent gas experiments are summarised in tables. Please note that the focus is on relative gas composition. All studies report on increasing absolute gas amounts with increasing SOC, but the focus of our analysis is on the relative amounts since it defines the flammability and explosion characteristics of the gas mixtures.

Some experimental data are plotted in figures to enable the discussion of trends. Cathode chemistry and SOC are the two factors generally considered to determine gas volume and composition. However, it is highly likely that gas composition is also dependent on other cell properties like the electrolyte composition and external factors, such as the amount of added energy transferred to the LIB cell during the abuse testing. Cell geometry (prismatic, pouch, cylindrical) may also play an important role since heat transfer and dynamics can vary depending on cell geometry and size.

5.4.1. LFP Batteries

Lithium–Iron Phosphate (LFP) batteries are claimed to represent a superior safety profile, long cycle life, and stable thermal characteristics compared to cells with layered oxide cathode materials, such as NMC and NCA. This is due to the chemical stability of the olivine structure. Unfortunately, LFP offers a lower energy density compared to other lithium-ion chemistries, but thanks to their long cycle life with many charge–discharge cycles, they are suitable for applications in electric vehicles, renewable energy storage, and grid stabilisation.

Experimental studies reporting on vent gases from LFP cells are collected in Table 7, and major gaseous components are plotted in Figures 2 and 3. All plotted data are for cylindrical cells with relatively low capacity, which means it is a fairly homogeneous dataset. However, heating has been conducted with different methods and heating rates, which may result in different dynamics and, therefore, also different gas production.

Table 7. Vent gas studies targeting LFP batteries. If the number of cells is not specified in the second column, the testing was performed on single-cell level.

Reference	Cell Geometry	Capacity, Ah	Electrolyte	SOC, %	Abuse Condition	Detection Method
Yang et al. (2023) [73]	Cyl 18650	1.5	DMC:EMC 1:1.9	25, 50, 75, 100	Electric heater, 19 W	GC, GC-FID
		1.1	DMC:EMC:EC 1.9:2.8:1			
		1.8	DMC:EMC:EC 1:1.9:1.4			
		1.1	DMC:EMC:EC 1.0:2.8:1			
Robles and Jeevarajan (2020) [74]	Cyl	5.5	DMC, EMC, ratio unknown	100	10 °C/min	FTIR, GC-MS
Yuan et al. (2020) [26]	Cyl 26650	3.8	-	100	ARC: 4.9 kW DSC: 10 °C/min	GC
Sturk et al. (2019) [75]	5 pouch cells	7	-	100	Heating (rate not given)	FTIR
Gully (2019) [76]	Cyl. 26650	1.5	-	50, 75, 100, OC	Heating (rate not given)	FTIR
		2.5				

Table 7. Cont.

Reference	Cell Geometry	Capacity, Ah	Electrolyte	SOC, %	Abuse Condition	Detection Method
Diaz et al. (2019) [44]	Cyl. 18650	1.1	DEC, DMC, EMC, PC. Ratios not known.	100	Heating (rate not given)	FTIR
Maloney (2016) [77]	Cyl. 26650	-	-	100	Heating (rate not given)	NDIR (CO, CO ₂), GC-FID (THC), O ₂ detection
Bergström et al. (2015) [78]	Pouch	-	-	-	Heating (rate not given)	GC-MS
Golubkov et al. (2015) [39]	Cyl. 18650	1.1	DMC, EMC, EC, PC, MPC	0, 50, 100, OC	Heating (rate not given)	GC
Golubkov et al. (2014) [62]	Cyl. 18650	1.1	DMC:EMC:EC:PC, 4:2:3:1	100	Heating, 1.5–3.5 °C/min	GC-MS

A clear trend seen in Figure 2 is that CO concentrations are low, below 10% for most cases, including both inert and air environments. There is no significant increase in CO with increasing SOC. CO₂ concentrations, as well as H₂ concentrations (Figure 3), are greatly scattered between the datasets, but within each dataset, the trend is decreasing CO₂ with increasing SOC. The hydrocarbons are generally below 20%, but for the few datasets with a higher hydrocarbon fraction, the CO and CO₂ levels are low, indicating a limited chemical oxidation.

Among the studies listed in Table 7, the ones by Golubkov et al. [39,62] are considered the most reliable in terms of gas composition due to the use of a good experimental design in an inert environment.

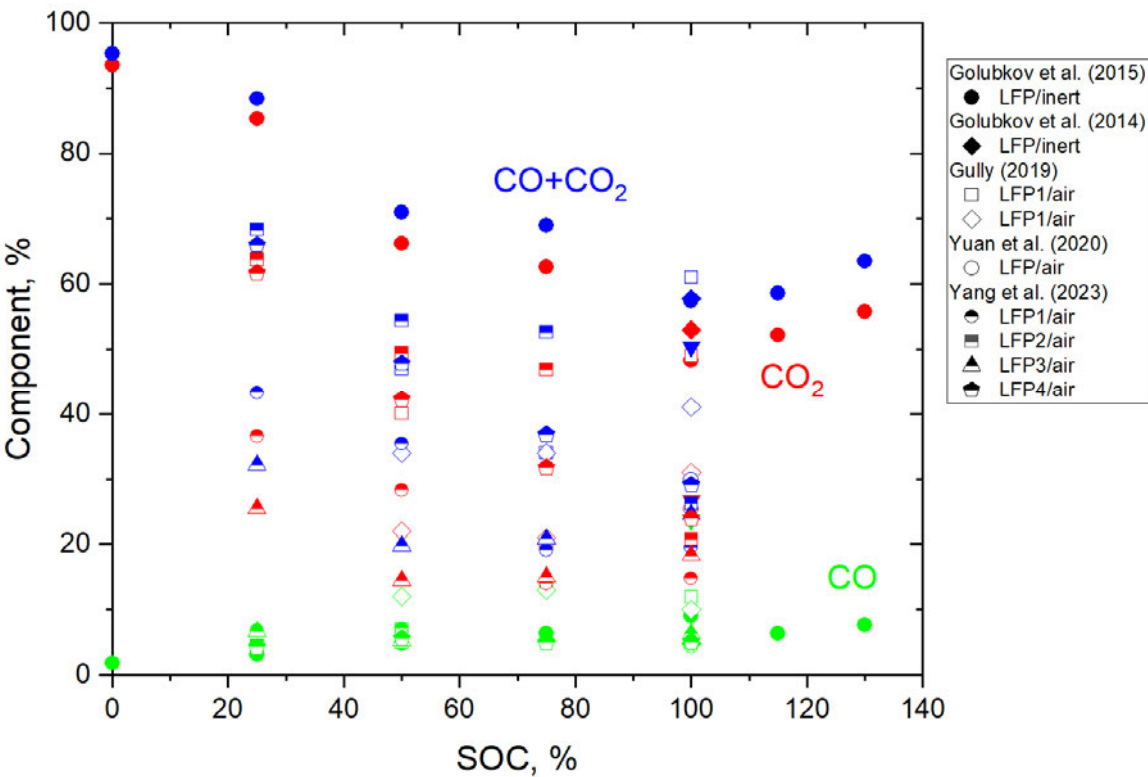


Figure 2. Fraction of CO and CO₂, and the sum of the two, vs. SOC for LFP cells. Filled symbols represent experiments performed in an inert environment. (The mentioned references are Golubkov et al. [39,62]; Gully [76]; Yuan et al. [26]; Yang et al. [73]).

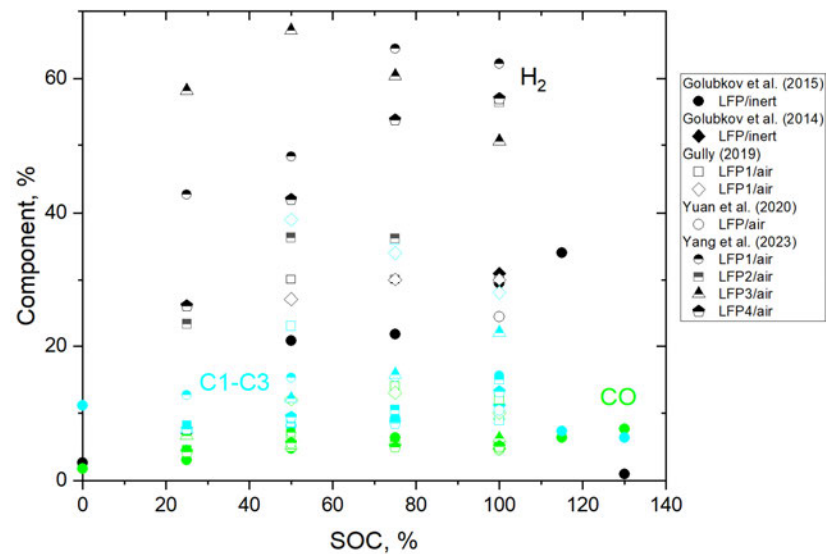


Figure 3. Fractions of CO and H₂, and the sum of C1–C3 hydrocarbons, vs. SOC for LFP cells. Filled symbols represent experiments performed in an inert environment. (The mentioned references are Golubkov et al. [39,62]; Gully [76]; Yuan et al. [26]; Yang et al. [73]).

5.4.2. NMC Batteries

Nickel–Manganese–Cobalt (NMC) batteries are used in applications requiring a balance of power and longevity and higher energy densities compared to LFP batteries. This makes NMC batteries particularly suitable for EVs and portable electronics, where energy density and performance are critical. NMC cells are quite well studied, as evident from Table 8, where the vent gas studies are collected. Studies have been performed on cylindrical, prismatic, and pouch cells over a wide range of capacities.

Table 8. Vent gas studies targeting NMC batteries. If the number of cells is not specified in the second column the testing was performed on the single-cell level.

Reference	Cell Geometry	Capacity, Ah	Electrolyte	SOC, %	Abuse Condition	Detection Method
Zhang et al. (2024) [79]	Cyl 18650	2.5	DEC:EMC:EC	30, 50, 70, 100, 120	Heating/ARC, rate not specified	GC-FID
	Prismatic	157	-	25, 50, 75, 100	Uniform heating, 1 °C/min	
Willstrand et al. (2023) [14]	Prismatic	157	-	50, 100	Uniform heating, 20 °C/min	Off-line analysis with NDIR, GC
	Prismatic	157	-	100	Local heater, rate not specified	
	Prismatic	157	-	>100	Overcharge	
Amano et al. (2023) [80]	Pouch	2.5	EC:DMC:DEC:EMC	30–100		FTIR
Amano et al. (2022) [81]	Pouch	10	DMC:EC:EMC:PC	30–100	Heating plug, rate not specified	FTIR
		10, 32			Heating plate, rate not specified	
Abbott et al. (2022) [63]	Cyl 21700	5	-	25, 100	Heating, ~20 °C/min	MS. Gas analyser CO
Essl et al. (2021) [9]	Pouch	60	EC:EMC 1:1	100	Heating, rate not specified	FTIR
Cai et al. (2021) [64]	Prismatic	-	-	0–213	Heating, rate not specified	NDIR, only CO ₂

Table 8. Cont.

Reference	Cell Geometry	Capacity, Ah	Electrolyte	SOC, %	Abuse Condition	Detection Method
Robles and Jeevarajan (2020) [74]	Prismatic	25	DMC, EMC, proportions unknown	100	Heating, 10 °C/min	FTIR, GC-MS
Yuan et al. (2020) [26]	Cyl. 18650	3.2	-	100	ARC: 4.9 kW DSC: 10 °C/min	GC
Essl et al. (2020) [61]	Pouch	60	EC:EMC 1:1	100	Heating, 2 °C/min	FTIR, GC
	Prismatic	60	EC:DMC:EMC 2:3:3	100	Heating, 2 °C/min	FTIR, GC
Chen et al. (2020) [82]	5 Cyl. 18650	-	-	60, 100	Uniform heating, rate not specified	GC-MS
	7 Cyl. 18650	-	-	100		
Gully (2019) [76]	Pouch	63	-	50–100, overcharge	Heating, rate not specified	FTIR
Diaz et al. (2019) [44]	Cyl. 18650	2.6	DEC, DMC, EMC, PC	100	Heating, rate not specified	FTIR
Koch et al. [68]	41 Pouch 10 Prismatic	20–81	EC, DMC, DEC, EMC	100	Heating, 1 °C/min	GC-WLD
Bergström et al. (2015) [78]	Pouch	-	-	-	Heating, rate not specified	GC-MS
Golubkov et al. (2014) [62]	Cyl. 18650	1.5	DMC:EMC:EC:PC, 7:1:1:1	100	Heating 2 °C/min	GC
Maloney (2016) [77]	Cyl. 26650	-	-	100	Heating, rate not given	NDIR (CO, CO ₂), GC-FID (THC), O ₂ detection

Figures 4 and 5 include data from studies covering a range of SOC for CO/CO₂ and CO/H₂/C1–C3, respectively. A general trend is increasing concentrations of reactive components and decreasing concentrations in CO₂ with increasing SOC. The capacity of the prismatic LIB cell studied by Willstrand et al. [14] was more than an order of magnitude larger than that of the cylindrical cells studied by Zhang et al. [79] and Abbott et al. [63].

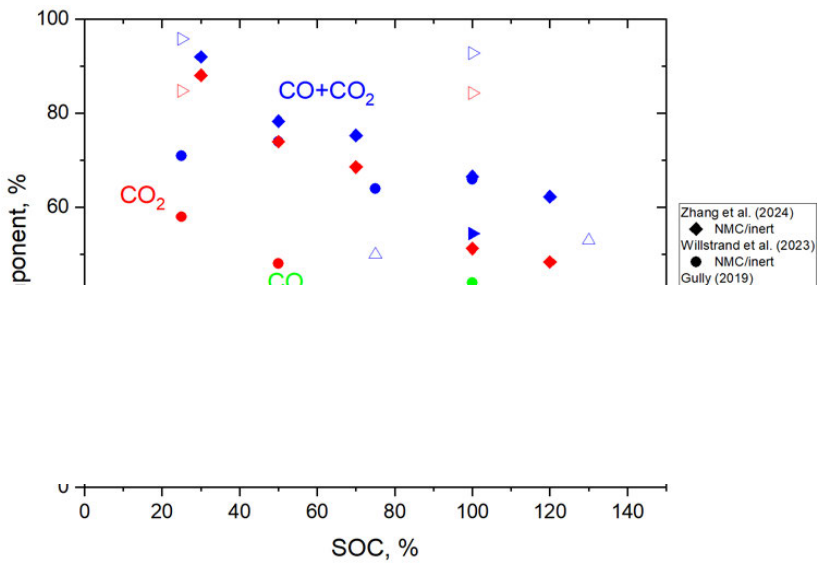


Figure 4. Fractions of CO and CO₂ and the sum of the two vs. SOC for NMC cells. Filled symbols represent experiments performed in an inert environment. (The mentioned references are Zhang et al. [79]; Willstrand et al. [14]; Gully [76]; Abbott et al. [63]).

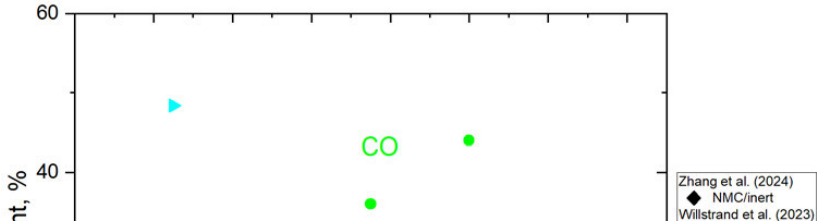


Figure 4. Fractions of CO and CO₂, and the sum of the two vs. SOC for NMC cells. Filled symbols represent experiments performed in an inert environment. (The mentioned references are Zhang et al. [79]; Willstrand et al. [14]; Gully [76]; Abbott et al. [63]).

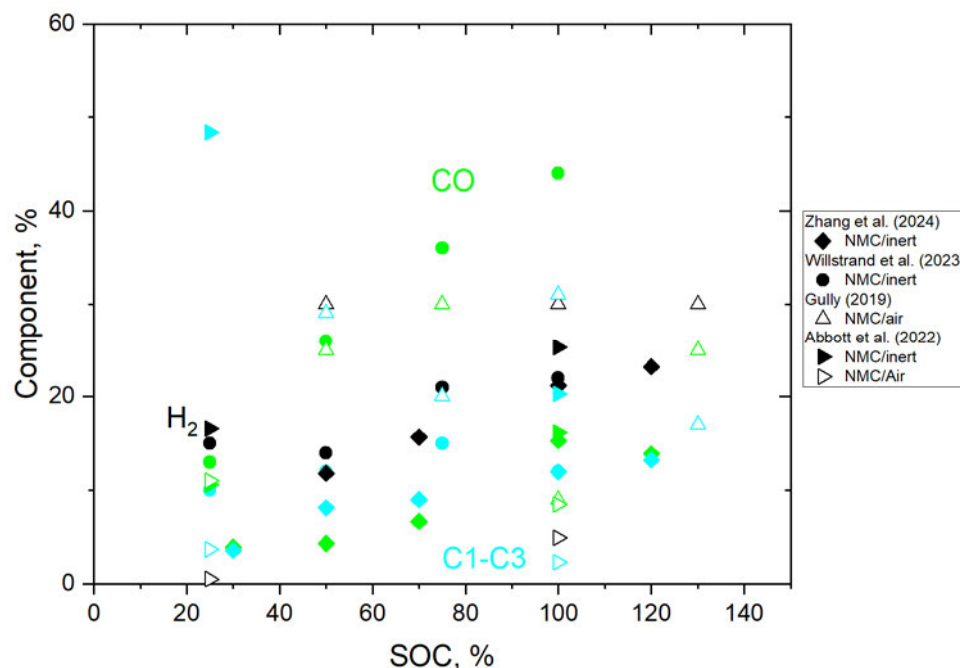


Figure 5. Fractions of CO and H₂, and the sum of C1–C3 hydrocarbons vs. SOC for NMC cells. Filled symbols represent experiments performed in an inert environment. (The mentioned references are Zhang et al. [79]; Willstrand et al. [14]; Gully [76]; Abbott et al. [63]).

5.4.3. **NCA Batteries** While most works focus on major constituents, the study by Diaz et al. [44] focused on toxic gases and is therefore valuable for the interpretation of toxicity based on minor substances. NCA batteries have a high specific energy and are common in high-performance applications, particularly in premium EVs and aerospace technologies, due to their excellent energy density and durability. The toxic components detected were DC, EC, PC, DMC, HCl, CO, Acrolein, COF₂, HF, and formaldehyde.

NCA LIB cells have been studied less than LFP and NMC cells, and as seen in Table 9, the only cell geometry studied is cylindrical 18650 cells with relatively low capacity. Heating in these experimental studies has been uniform and at a moderate rate, which indicates that the results may be comparable. Also, all studies employed GC techniques for gas measurements of CO, CO₂, H₂, and small hydrocarbons of CH₄, C₂H₄, and C₂H₆, but excluding electrolyte species. Several studies targeted the detection of O₂, but it was not found.

The first two studies, published in 2004 and 2006 by Crafts et al. [83] and Abraham et al. [27], respectively, reported experiments where the cells were heated to 150 or 160 °C and then gas samples were extracted. Abraham et al. reported that for an NCA cell subject to slow heating, self-heating started at 84 °C. They allowed heating to proceed up to about 150 °C, then quenched the cells and investigated gas composition and structural changes in the cell materials. Details on the structural changes are outside the scope of the present work but are valuable. Measurements were performed on gases that was extracted from swelled, non-ruptured cells and on the gases that were vented to the atmosphere. The non-vented gas had a significantly higher CO content, which could indicate that, for the vented gases, CO₂ was produced from CO and was further oxidised as the gas came in contact with ambient air. Another finding was that there was quite a large difference between cells quenched at 150 °C and 160 °C; the additional 10 °C led to more than a doubling of the gas volume, indicating that significant internal processing occurred in the cell at this temperature.

The later studies in Table 9 allowed heating to higher temperatures, and the cells went into TR. The experimental setups were designed to maintain an inert atmosphere, which prevented the hot gases from combusting upon venting. Golubkov et al. [39] covered the widest range of SOC, including overcharge.

Table 9. Vent gas studies targeting NCA batteries. If the number of cells is not specified in the second column, the testing was performed on the single-cell level.

Reference	Cell Geometry	Capacity, Ah	Electrolyte	SOC, %	Abuse Condition	Detection Method
Abbott et al. (2023) [84]	Cyl. 18650	~3	-	100	Heating, rate not specified, ARC	GC-MS, inert and air atm.
Lammer et al. (2018) [18]	Cyl. 18650	3.35, 2.5	-	100	Heating, 0.5 °C/min, furnace	GC
Lammer et al. (2017) [35]	Cyl 18650	3.2, 3.5	-	100	Heating, 0.5 °C/min, furnace	GC
Golubkov et al. (2015) [39]	Cyl 18650	3.35	DMC, EMC, EC, PC, MPC	0–100, OC	Heating, rate not specified	GC
Abraham et al. (2006) [27]	Cyl 18650	~1	EC:EMC 3:7 (wt)	100	Heat to 150 °C, ~0.02 °C/min, ARC	GC-MS Not inert atm.
Crafts et al. (2004) [83]	Cyl 18650	1.1	EC:EMC 3:7 (wt)	60, 100	Heating, 0.02 °C/min, ARC	GC-MS Not inert atm.

Two studies were published by Lammer et al. in 2017 [35] and 2018 [18]. In the later study, they investigated three NCA batteries with different characteristics that gave different gas compositions. Lammer et al. [18] argued that there are differences in other properties, like separator material and thickness, that could affect the gas composition, but the effects of these factors have not been studied. The gases expelled at TR are more similar for the different NCA cells compared to an NMC cell from the same study and contain high levels of H₂ and CO.

Figure 6 presents CO and CO₂ concentrations and the sum of the two for the studies on NCA cells listed in Table 9. The study by Golubkov et al. [39] is the only one covering the full range of SOC. This study is of high quality with good experimental design and excellent characterisation of important aspects of the experiment and is, thus, considered reliable. An important trend was that at a low SOC, CO₂ dominated, while CO increased and dominated at a SOC greater than 60%. However, the sum of the CO and CO₂ was consistent at about 70% for all cells. The other studies performed in an inert atmosphere, all at SOC 100%, showed some scattering but no clear deviation from the relationship between CO and CO₂. Figure 7 represents flammable components, including CO, H₂, and the sum of C1–C3 hydrocarbons for the studies in Table 9. Here, the results produced in an inert atmosphere are in quite good agreement, while the ones in the air deviate. The difference between studies in an air environment can likely be attributed to the fact that there were various amounts of oxidiser present and different time scales for the oxidation chemistry to take place. The extent of external oxidation will depend on the volume of air that the vent gases are expanded into and the time before the measurement is conducted. Also, the dynamics of the venting process may be affected since the entrainment of the air into the venting plume can result in a different extent of oxidation.

5.4.4. NMC/LMO Batteries

Nickel–Manganese–Cobalt oxide/Lithium–Manganese oxide (NMC/LMO) batteries are a hybrid type of LIB that combines the advantages of both NMC and LMO chemistries. By blending these two materials, the battery benefits from the high energy density of NMC and the safety and stability of LMO. The two available studies are summarised in Table 10.

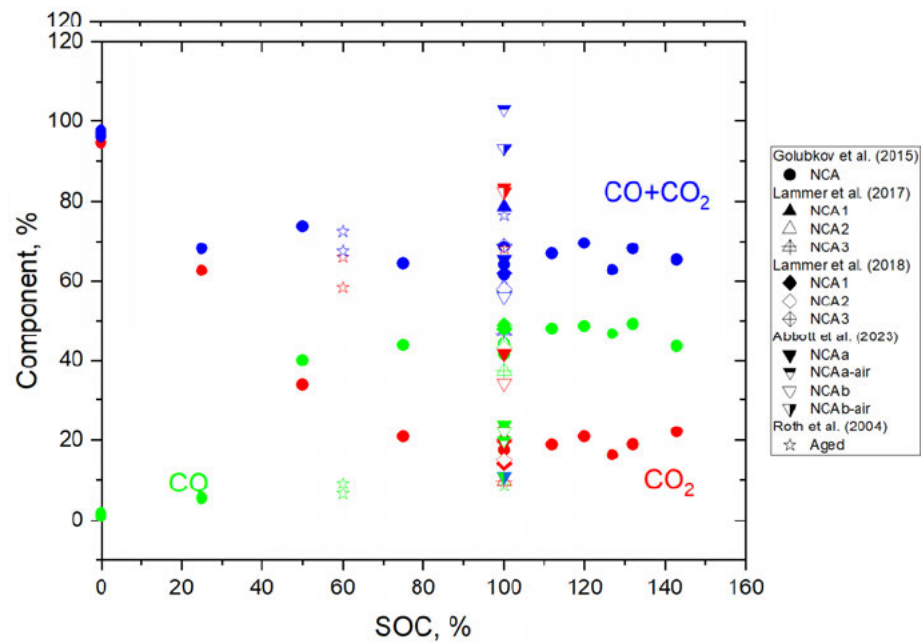


Figure 6. Fraction of CO and CO₂, and the sum of the two vs. SOC for NCA cells. (The mentioned references are Golubkov et al. [39]; Lammer et al. [18,35]; Abbott et al. [84]; Roth et al. [24]).

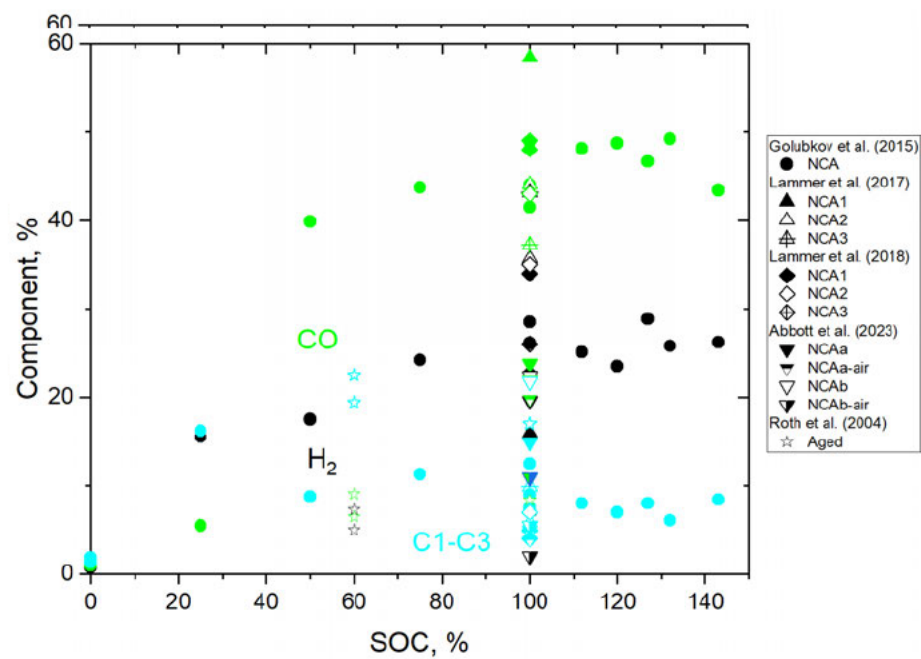


Figure 7. Fraction of CO and H₂, and the sum C1–C3 hydrocarbons vs. SOC for NCA cells. (The mentioned references are Golubkov et al. [39]; Lammer et al. [18,35]; Abbott et al. [84]; Roth et al. [24]).

Table 10. Vent gas studies targeting NMC/LMO batteries. If the number of cells is not specified in the second column, the testing was performed on the single-cell level.

Reference	Cell Geometry	Capacity, Ah	Electrolyte	SOC, %	Abuse Condition	Detection Method
Sturk et al. (2019) [75]	5 pouches	14	-	100	Heating, rate not specified	FTIR
Essl et al. [60]	Pouch	41	EC:DEC:DMC 48:48:4	100, 30, 0	Heating, rate not specified	FTIR, GC

The study by Essl et al. [60] on pouch cells was also conducted in an inert environment and gave a low relative concentration of CO compared to studies on other battery chemistries. Also, low H₂ fractions were detected by Essl et al. [60]. The study seemed to be well-designed and may be an indicator that NMC/LMO vent gases are relatively harmless. However, more studies are needed to establish this trend.

Sturk et al. [75] detected only the components HF and CO₂, but is a valuable contribution to the understanding of the dynamics of venting.

5.4.5. LCO Batteries

Lithium–Cobalt oxide (LCO) batteries are one of the earliest and most widely used types of LIB, particularly in consumer electronics like smartphones, laptops, and cameras. The cathode material in LCO batteries is composed of cobalt oxide, which provides a high energy density, allowing for compact and lightweight battery designs with substantial energy storage.

Even though LCO batteries are widely used, there are few studies dedicated to the determination of gas composition resulting from thermal abuse. The three available studies are summarised in Table 11.

Table 11. Vent gas studies targeting LCO batteries. If the number of cells is not specified in the second column, the testing was performed on the single-cell level.

Reference	Cell Geometry	Capacity, Ah	Electrolyte	SOC, %	Abuse Condition	Detection Method
Kennedy et al. (2021) [85]	Pouch	10	EC:DMC, 1:1	50, 75, 100	Fast uniform heating	GC-TCD
	5 Pouches	10		50, 75, 100		
	5 Pouches	18.5		100		
Diaz et al. (2019) [44]	Cyl. 18650	3	DEC, DMC, EMC, PC. Ratios not known.	0, 50, 100	Uniform heating (rate not stated)	FTIR
	Pouch	2.5		100		FTIR
Maloney (2016) [77]	Cyl. 18650	2.6	-	10–100	Uniform heating (rate not stated)	NDIR (CO, CO ₂), GC-FID (THC), O ₂ detection

The studies by Maloney [77] and Diaz et al. [44] both include cylindrical LIBs with similar capacity and covering a range of SOC, but apart from that, the information from the publications is quite different with Diaz et al. focusing on toxic products and Maloney on combustible gases. Maloney saw a rather constant THC (total hydrocarbon) level throughout, while the concentrations of combustible CO and H₂ increased from about 40 and 60% SOC, respectively. A weakness in this study is that the quantified gases (CO, CO₂, H₂, THC) only add up in the range of 20 to 80% of the total gas volume, which means there is a large number of other gas species that were not identified. This, in combination with a lack of information on the LIB cell and experimental procedure, limited the use of the data for the assessment of gas composition from LCO batteries.

The recent work by Kennedy et al. [85] focused on the quantification of the gas release rate, but they also reported on gas composition for several SOC. This is a well-designed study that characterises the LIB cell composition and experiments both on single pouch cells and arrays of five cells. Overall, there were no large differences in the relative concentrations of the measured components (CO, CO₂, H₂ and small hydrocarbons) over the studied range of SOC. In comparison to other cell chemistries, the authors commented on the fact that they measured lower CO and higher CO₂, but this may be a result of the fact that they performed the measurements in an air environment while comparing them to studies in an inert atmosphere.

The study by Kennedy et al. [85] is recommended as the best available study on LCO LIB, including valuable results and discussion on the differences between single cells and

arrays. However, since the data on gas composition from venting LCO cells is so scarce, no conclusions can be made on potential differences compared to other cell chemistries.

6. Discussion

In this section, selected topics of relevance for the assessment of the quality of vent gas studies are discussed.

6.1. Influence of Gas Analysis Methods

The most common gas analysis techniques to study vent gases from LIB are FTIR and GC, with the latter often used in combination with mass spectrometry. A common characteristic of these methods is that they can be used for the screening of all hydrocarbons and many other relevant components. Characterisation, including minor constituents like benzene, toluene, and styrene, has been performed by Nedjalkov et al. [70], Gully [76], and Zhang et al. [42], and including amines and nitriles by Sun et al. [86].

For the assessment of fire and explosion risks, it is sufficient to detect major products of CO, CO₂, H₂, and small hydrocarbons. In some cases, electrolytes and other oxygenated hydrocarbons are also relevant if they are present in quantities around a few percent of the total gas volume or more. It is most common to present the gas composition as a percentage of the total gas volume. This introduces a potential source of error if all major constituents are undetected. It is, for example, plausible that the electrolyte solvent was indeed a significant contributor to the total gas volume in a few studies where it was not monitored. If this is true, the presented results indicate too-high concentrations of the reported components. When evaluating the results, it is, therefore, important to understand how the reported percentages are derived, if they build on a true quantification, or are just a presentation of the relative amounts of the selected components that were measured in a specific experiment.

Although HF is not the focus of the present review, it is still relevant to comment on the challenges in accurately detecting and quantifying this compound [87] and, indeed, all reactive fluorinated compounds. Independent of the analysis method, the first challenge was the sampling distance or sampling time and the potential for interactions between the reactive compound and the sampling system. In studies where HF was targeted but undetected, it was likely that it was absorbed on surfaces or was converted to a more stable compound in the sampling system [22,75]. An unexpectedly low HF concentration was, thus, likely a result of the removal of HF in the system. The over-prediction of HF can also occur, for example when the wash bottle technique is used, due to the fact that all F⁻, i.e., inorganic fluorine, will form a precipitate. Therefore, wash bottle results should be interpreted as “total inorganic F” [75]. The accurate determination of HF can be performed in situ using, for example, FTIR detection. But, as discussed by Sturk et al. [75], FTIR results can also be erroneous if it is performed offline. The spread in data, by several orders of magnitude, presented in the review of HF concentrations by Bugryniec et al. [8] is likely a result of both the over- and underestimation, as explained above.

6.2. Presence of O₂ in the Vented Gases

As outlined previously, O₂ is produced during the decomposition of cathode material of most LIB types at temperatures in the vicinity of the threshold temperature for TR. It has been speculated that oxygen escaping in the vent gases will sustain a flame, which can be tested by simply calculating the amount of oxygen needed to combust the different gases potentially vented.

Golubkov et al. [39] used a simplified calculation to establish that about 6% of the produced gas volume from the cathode material of an over-charged NCA cell can be O₂. Based on reaction kinetics, an amount this small will be completely consumed by chemical reactions inside the LIB cell. Possible reaction paths for oxygen produced inside the cell can be to react with the lithiated anode, resulting in significant heat production and a range of reactions with the electrolyte solvents [3].

O₂ has been a target component in several experimental studies of vent gas composition using GC [39,60–62] but was not detected. A few percent of O₂ was measured in vent gases from a single LCO cell at 100% SOC but not from an array of five cells in the same study under similar conditions [85]. The authors suggest that the time frame for internal chemical processing is shorter in the single cell since it cools faster; therefore, O₂ escape unreacted.

Willstrand et al. [88] performed a series of tests where O₂ was measured for 70 Ah and 157 Ah cells at 75% and 100% SOC, where two cases were triggered by a slow heating ramp and the rest used a rapid trigger method. The cells that were brought to TR by slow heating did not show O₂ in the vented gases. The cells that were triggered by a rapid trigger method released measurable amounts of O₂. The lower capacity (70 Ah) of cells had 3–6% O₂ in the vent gases when the SOC was 75%, but for both the fully charged 70 Ah cells and the high capacity (157 Ah) cell at 75%, the concentrations were about 1% or less.

To summarise, the majority of the published gas studies show that no oxygen is vented from LIB cells, and the theoretical analysis of the reaction chemistry strongly suggests that the oxygen is consumed before venting. The few experiments that indeed have O₂ in the vent gas are for single cells with relatively low capacities and at a lower SOC. The described cases have a common denominator in that the temperature after TR triggering is lower and, thus, the internal chemical processing is slower. In the case of a real EV field fire, the scenario is similar to the laboratory cases where no oxygen was detected compared to the ones where oxygen was detected. Furthermore, based on a pure combustion consideration, even the highest amount of oxygen that was detected, about 6%, cannot realistically be expected to sustain a fire. It can, therefore, be concluded that it is very unlikely that LIB fires can be sustained by the internally released oxygen present in the gas mixture and that open flame fire relies on the availability of atmospheric oxygen. However, the oxygen produced by the cell may be sufficient to feed a low-intensity smouldering that may flare up if and when ambient oxygen from the atmosphere becomes available.

6.3. The Potential Role of Solvent Composition

Carbonates commonly used as electrolyte solvents differ in important properties like boiling temperature, decomposition temperature, and combustion characteristics. When it comes to combustion, it is also worth noting that the oxygen present in the carbonate molecules can act as oxidisers without having to first form O₂ and that the carbonates are generally combustible at lower external oxygen concentrations compared to hydrocarbons. A number of papers have reported the flammability limits [89,90], ignition delay time [91] and laminar flame speeds [92,93] of relevant carbonates. These works are important when attempting to assess the risk related to the leakage of carbonates as a result of cell punctures, electrolyte spills during manufacturing, or when handling end-of-life LIBs, for example, during recycling. However, to understand the processes occurring during internal chemical processing in an LIB cell during a thermal event, understanding the evaporation and decomposition conditions of the electrolyte solution is more important. In addition, chemical reactions of carbonates with other cell components or their products following decomposition drive exothermic chemistry, resulting in excess heat production during a thermal event.

The boiling points for the common carbonates at ambient conditions are in the range from 91 °C for the smallest carbonate DMC to 248 °C for the cyclic EC [94]. The expected environment inside an LIB cell is different from the ambient, which impacts the electrolyte solvent characteristics. For example, the pressure inside the cell is higher inside the cell and slowly increases due to normal ageing, which will shift the boiling temperature to higher values, whereas the presence of Li-salt may decrease the stability and initiate decomposition at lower temperatures than ambient temperatures. However, the trends in boiling point and stability are still relevant to consider. A mild thermal event can start at a temperature as low as 69 °C, starting with the breakdown of the SEI layer, followed by heat production. The temperature window for a mild thermal event and transition to a severe thermal

runaway will then stretch up to above 200 °C, with the onset temperature for TR varying for different cell chemistries. Considering the boiling temperature of the carbonates, it is clear that some can turn to the gas phase early in the heating, while others stay in the liquid phase. This implies that the amount of material in the gas phase and, thus, the internal pressure in the cell can be dependent on the carbonate composition, with a more rapid pressure increase when the carbonates with low boiling temperatures like DMC or EMC (109 °C) are the dominant electrolyte solvent species. The order of boiling points for four common carbonates are as follows: DMC < EMC < DEC < EC. However, the trends for decomposition are different and complicate the situation.

The decomposition of the carbonates is partly dependent on the presence of Li-salt. As shown by Lamb et al. [22], for temperatures up to about 450 °C, the linear carbonates DMC and EMC do not decompose in the studied temperature range if no Li-salt is present. With 1.2 M of LiPF₆, EMC shows signs of thermal decomposition already at 175 °C, while DMC is not affected. EC and DEC decompose thermally both in the absence and presence of Li-salt, but the decomposition of DEC occurs at a temperature of approximately 100 K lower with salt, while the onset temperature is essentially unchanged for EC. The reactivity of the carbonates in the presence of Li-salt, based on the analysis of the decomposition onset temperature and gas volume, has a trend opposite to that of the boiling point, with EC being the most reactive, followed by DEC, EMC, and DMC.

Yet another factor that needs to be considered for carbonate reactivity is the reaction with O₂, which takes place at temperatures high enough for oxygen to be released from the cathode. The cyclic EC reacts rather slowly with O₂, and the thermal decomposition is likely the most important degradation route over a broad temperature range. Among the linear carbonates, DEC tends to be the most reactive towards O₂ since it has longer carbon chains (two ethyl groups) compared to EMC (one ethyl and one methyl group) and the least reactive DMC (two methyl groups).

The difference in hydrocarbon chain length in the carbonates will have implications for the production of small hydrocarbons and, hence, their relative amounts during electrolyte decomposition and reactions.

6.4. Relevance of Measured CO and CO₂ Concentrations

CO and CO₂ are two of the major products in the battery vent gases and can be produced via several different routes during the internal chemical processing in a LIB thermal event. The two compounds originate from hydrocarbon oxidation, and since the main hydrocarbon content in an LIB cell is the carbonates in the electrolyte, one can expect that the amount and composition of carbonates will affect their total amount. However, the availability of chemical reaction partners in the oxidation process, originating from the electrode materials, will affect the relative amounts of CO and CO₂.

CO₂ is a product of the direct decomposition of the carbonate electrolytes, and as explained in the previous section, carbonates have a different propensity to decompose. CO₂ is also produced when the SEI layer degrades and as a result of reactions of electrode materials. In the presence of O₂ produced inside the cell from the cathode material, carbonates can produce either CO or CO₂ depending on the ratio of O₂ to the carbonate. Lower O₂ concentrations result in incomplete oxidation and more CO production. Another mechanism that can increase CO concentration is when intercalated Li at the anode reduces CO₂ and carbonates to CO.

The processes for CO and CO₂ production, to a large extent, occur simultaneously as the temperature in the LIB cell increases and the chemical composition evolves. It is outside the scope of the present work to investigate the details of the chemical dynamics, but it is relevant to point out that since the reactions have different temperature dependencies, their relative importance changes over time. The degree of chemical pre-processing before the venting starts will affect the ratio, and therefore, there will be differences between, for example, slow thermal abuse that allows significant chemistry to happen inside the LIB

cell before venting starts, and rapid penetration, which is a local event which immediately brings the cell chemicals in contact with external air.

Experimental studies on vent gas composition present scattered results for relative amounts of CO and CO₂. However, a large number of studies are hampered by limitations in experimental design and gas analysis, and the results are potentially not reliable. The relevant quantity for the assessment of risk for fire and explosion is the composition of the gas as it is vented, while unfortunately, many studies report data representing gas mixtures that have been subject to oxidation external to the battery. Recent reviews include data from different study designs. First, CO/CO₂ from experiments where combustion occurred after gas release are not relevant to compare to fresh vent gases. However, even in cases where there was no combustion, the oxidation of CO to CO₂ likely occurred in the presence of O₂ and other gas-phase oxidisers if the environment was not inert. Hence, only the CO/CO₂ ratio measured in an inert atmosphere can be considered representative of the vent gas composition.

Since the conversion of CO to CO₂ after venting is suspected in many studies, we also chose to plot the sum of the two components in Figures 2, 4 and 6. It is seen that the sums of the components are in better agreement between the studies compared to the ratio of the individual gases.

6.5. Dependence on SOC and Cathode Chemistry

There is consensus in the literature that gas compositions from LIBs venting vary depending on cathode chemistry and SOC, but unfortunately, the experimental data are scattered, with studies pointing at different trends. As revealed in the present work, experimental design, gas analysis techniques, and the interpretation of data affect the results. The consequence of this is that different studies are not comparable because they are performed with different prerequisites. In the present work, an effort was made to identify datasets that represent vent gases prior to combustion and without effects from the external environment. Despite this, the data show significant variability, but there are some trends that can be identified.

From the data selected in this study for LFP, NMC, and NCA LIBs, as plotted in Figures 2–7, it can be concluded that over the full range of SOC, the relative concentration of inert CO₂ decreases and the more reactive components, including CO, increase as SOC increases. There is a weak trend of increasing CO levels at a higher SOC range, above about 75%. Some trends connected to the cathode chemistries are seen, for example, that LFP produce low concentrations of CO. However, the trends should be seen as indications, not as proof, due to the limited data available.

An example which indicates that cathode chemistry is not necessarily the main factor affecting the gas composition is the study on NCA cylindrical LIB cells by Lammer et al. This study shows that although the cells have the same cathode chemistry and an almost identical nominal capacity, the gas compositions are quite different in repetitive tests. The most extreme difference is found in the relative amount of H₂ in the released gases, which ranges from about 18 to 42%. Considering that this difference is seen in the same study with identical experimental methodology, it is one of the most striking examples that cathode chemistry may not be a strong determining factor for gas composition.

7. Conclusions and Outlook

In the present work, experimental studies on the composition of vented gases from LIBs were reviewed. As discussed in detail in previous sections, the use of different experimental designs and measurement methods hamper the potential to draw general conclusions from the datasets. In some cases, the usefulness of studies is limited not by the quality of the study but by the lack of detailed explanation of samples or methodologies. The repeatability and reproducibility of test results rely on good experimental planning and the control of all test parameters. Test results in currently available publications on LIB

gassing offer a scattered picture, and their comparison is challenging due to differences in how the experiments were conducted as well as what data were collected and reported.

The datasets from vent gas studies are valuable for the assessment of fire safety and toxicity and can contribute to the development of mitigation strategies. An experimental investigation of LIB venting ideally reports on mass loss and total gas volume and provides complete speciation with the accurate quantification of flammable substances. In addition, to reveal the sources of gases, it is highly beneficial if the tested LIB cells are completely characterised, including the weight of active and passive materials and the composition of the electrolyte. For multi-cell assemblies and battery-level tests, information about battery layout and cell connections, as well as quantified amounts of other flammable materials present in the battery that may contribute to the gaseous emissions, is needed to make the correct assessment of the released gases. The experimental studies on venting presented in Section 5 of the present work are summarised in Table 12 as a guide for the level of information given in the published studies.

Table 12. Summary of data collected in different studies. Abbreviations: UH—uniform heating, P—penetration, OC—over-charge. Legend: Trig.—trigger method; Char.—complete characterisation of battery; Mass.—mass loss; TGV—total gas volume; Spec.—complete speciation; Quant.—quantification. (x) in the speciation column means that speciation is extensive but not complete on the main compounds.

	Trig.	Char.	Mass	TGV	Spec.	Quant.	Comments
Zhang et al. (2024) [79]	UH	-	x	x	x	-	Speciation up to C2
Yang et al. (2023) [73]	UH	-	x	x	x	x	-
Willstrand et al. (2023) [14]	UH	-	x	x	x	-	-
Abbott et al. (2023) [84]	UH	-	x	x	x	-	-
Amano et al. (2023) [80]	UH	-	x	x	(x)	x	No H ₂ detection
Abbott et al. (2022) [63]	UH	-	-	x	x	x	-
Amano et al. (2022) [81]	UH	-	x	x	(x)	x	No H ₂ detection
Cai et al. (2021) [64]	OC	-	-	-	-	-	-
Essl et al. (2021) [9]	UH	x	x	x	x	x	Influence of ageing
Hoelle et al. (2021) [69]	P	-	-	-	-	-	-
Kennedy et al. (2021) [85]	UH	x	x	x	x	x	-
Essl et al. (2020) [61]	UH, P, OC	x	x	x	x	x	-
Essl et al. (2020) [60]	UH	x	x	x	x	x	-
Robles and Jeevarajan (2020) [74]	UH	-	-	x	x	x	Upscaling; electrolyte composition not known
Yuan et al. (2020) [26]	UH	-	-	x	-	x	All gases (carbonates) not detected
Sturk et al. (2019) [75]	UH	-	-	x	x	-	Good speciation; incomplete quantification
Gully (2019) [76]	UH, OC	-	-	x	(x)	x	No H ₂ detection
Diaz et al. (2019) [44]	UH, P	-	x	x	x	-	Detection of carbonates and F-compounds; electrolyte composition not known
Fernandes et al. (2018) [4]	OC	x	x	x	x	x	Complete characterisation of LIB
Koch et al. (2018) [68]	UH	-	x	x	x	x	Many cells evaluated together
Koch et al. (2018) [68]	P	-	x	x	-	x	Modules
Lammer et al. (2018) [18]	UH	-	-	x	-	x	Various storage scenarios
Lammer et al. (2017) [35]	UH	-	-	x	-	-	-
Gully (2019) [76]	UH	-	-	x	x	x	-

Table 12. Cont.

	Trig.	Char.	Mass	TGV	Spec.	Quant.	Comments
Maloney (2016) [77]	UH	-	-	x	-	x	Performed upscaling experiment; SOC range
Nedjalkov et al. (2016) [70]	P	x	x	x	-	-	
Bergström et al. [78]	UH	-	-	-	x	x	Focus on speciation/toxicology; SOC not given
Yuan et al. (2015) [71]	OC	x	-	-	(x)	-	
Golubkov et al. (2015) [39]	UH	x	x	x	-	x	Characterisation of LIB; electrolyte not detected
Golubkov et al. (2014) [62]	UH	x	x	x	-	x	-
Abraham et al. (2006) [27]	UH	x	-	x	-	x	-
Crafts et al. (2004) [83]	UH	x	-	x	x	x	-

While new and improved experimental studies on LIB cell TRs are essential for increased understanding, it is also important to increase the fundamental understanding of the underlying chemical and physical processes. A substantially increased understanding at the molecular level can be gained from computational studies, and dedicated experiments can be performed on the component level. The outcome from such research then needs to be aggregated into computational models, taking different aspects into account and investigating their interdependence. Groundbreaking work in the modelling of the thermal runaway in LIB cells has recently been published by García et al. [95,96], and for further detail, we refer to these works and the references therein.

The present work, unfortunately, focuses more on the differences between published studies than on similarities since we believe that it is necessary to bring to the common understanding that general conclusions cannot be drawn from the bulk of the studies reporting on the gas composition of vent gases from LIBs. To mitigate this, the present review has highlighted some important factors to consider in future experiments to improve comparability:

1. Detailed information about the test object, e.g., the cell type and size, cell chemistry, and composition of materials, including electrolyte solvent composition;
2. The TR trigger condition to determine the severity of the abuse;
3. Clarity regarding what gases (e.g., vent gases, combustion gases or a mix) are studied;
4. The timing and method of gas measurement (e.g., direct measurement or delayed measurement on sampled gases) as this affects the chemical history of the gases due to the reactivity of some species;
5. Controlled gas environment, e.g., turbulent or stationary conditions, inert or air atmosphere, and gas chamber volume;
6. When evaluating the gas measurements, it is also important to consider the possibility of other oxidants in addition to O₂, and it is recommended to perform a mass balance to the greatest extent possible in order to verify the consistency of results.

Another aspect that needs to be highlighted is the interpretation of TR trigger methods. TR is triggered in order to simulate internal short circuit conditions and evaluate cell and battery safety performance for this fault condition. However, it is important to remember that all current test methods rely on forced thermal runaway abuse triggers that introduce some level of acceleration compared to spontaneous internal short circuit events, either in the form of added energy or by creating multiple shorting conditions inside the triggered cell.

When choosing the trigger method, it is important to consider its relevance to the real-world condition it aims to simulate. For the analysis of the heat production and the temperature increase and acceleration of chemical reactions, it is important to consider if this is an effect of energy transferred to the system (via the trigger) or originating from the system. The TR trigger method affects the test results since gassing reactions depend on the

dynamics of the chemical reactions, especially the speed of the reaction. Slow TR initiation, such as the homogeneous heating of the entire cell, favours higher relative amounts of CO₂, whereas fast triggers, like nail penetration or high-power localised heaters, typically lead to incomplete oxidation and higher relative concentrations of CO.

Finally, we would like to stress that the test level influences the TR dynamics, and single-cell results cannot be extrapolated linearly to multi-cell configurations due to the higher temperatures realised in the latter case, which directly impacts the chemical processing and rates of reaction. For highly reactive species, such as HF and POF₃, the presence of additional surfaces in multi-cell assemblies and batteries reduces the actual gas concentrations due to rapid adsorption and reaction. Vent gas studies on multi-cell configurations are few, and we encourage the research community to take on the challenge of studying these to a larger extent.

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References

1. Wang, X.; Liu, H.; Pan, K.; Huang, R.; Gou, X.Q.; Qiang, Y.J. Exploring thermal hazard of lithium-ion batteries by bibliometric analysis. *J. Energy Storage* **2023**, *67*, 107578. [\[CrossRef\]](#)
2. Wang, Q.S.; Ping, P.; Zhao, X.J.; Chu, G.Q.; Sun, J.H.; Chen, C.H. Thermal runaway caused fire and explosion of lithium ion battery. *J. Power Sources* **2012**, *208*, 210–224. [\[CrossRef\]](#)
3. Wang, Q.S.; Mao, B.B.; Stolarov, S.I.; Sun, J.H. A review of lithium ion battery failure mechanisms and fire prevention strategies. *Prog. Energy Combust. Sci.* **2019**, *73*, 95–131. [\[CrossRef\]](#)
4. Fernandes, Y.; Bry, A.; de Persis, S. Identification and quantification of gases emitted during abuse tests by overcharge of a commercial Li-ion battery. *J. Power Sources* **2018**, *389*, 106–119. [\[CrossRef\]](#)
5. Qiu, M.M.; Liu, J.H.; Cong, B.H.; Cui, Y. Research Progress in Thermal Runaway Vent Gas Characteristics of Li-Ion Battery. *Batteries* **2023**, *9*, 411. [\[CrossRef\]](#)
6. Ghiji, M.; Edmonds, S.; Moinuddin, K. A Review of Experimental and Numerical Studies of Lithium Ion Battery Fires. *Appl. Sci.* **2021**, *11*, 1247. [\[CrossRef\]](#)
7. Rappsilber, T.; Yufi, N.; Kruger, S.; Hahn, S.K.; Fellingner, T.P.; von Nidda, J.K.; Tschirschwitz, R. Meta-analysis of heat release and smoke gas emission during thermal runaway of lithium-ion batteries. *J. Energy Storage* **2023**, *60*, 106579. [\[CrossRef\]](#)
8. Bugrynec, P.J.; Resendiz, E.G.; Nwophoke, S.M.; Khanna, S.; James, C.; Brown, S.F. Review of gas emissions from lithium-ion battery thermal runaway failure—Considering toxic and flammable compounds. *J. Energy Storage* **2024**, *87*, 111288. [\[CrossRef\]](#)
9. Essl, C.; Golubkov, A.W.; Fuchs, A. Influence of Aging on the Failing Behavior of Automotive Lithium-Ion Batteries. *Batteries* **2021**, *7*, 23. [\[CrossRef\]](#)
10. Broussely, M.; Biensan, P.; Bonhomme, F.; Blanchard, P.; Herreyre, S.; Nechev, K.; Staniewicz, R.J. Main aging mechanisms in Li ion batteries. *J. Power Sources* **2005**, *146*, 90–96. [\[CrossRef\]](#)
11. Thompson, L.M.; Harlow, J.E.; Eldesoky, A.; Bauer, M.K.G.; Cheng, J.H.; Stone, W.S.; Taskovic, T.; McFarlane, C.R.M.; Dahn, J.R. Study of Electrolyte and Electrode Composition Changes vs Time in Aged Li-Ion Cells. *J. Electrochem. Soc.* **2021**, *168*, 020532. [\[CrossRef\]](#)
12. Claassen, M.; Bingham, B.; Chow, J.C.; Watson, J.G.; Chu, P.B.; Wang, Y.; Wang, X.L. Characterization of Lithium-Ion Battery Fire Emissions-Part 2: Particle Size Distributions and Emission Factors. *Batteries* **2024**, *10*, 366. [\[CrossRef\]](#)
13. Claassen, M.; Bingham, B.; Chow, J.C.; Watson, J.G.; Wang, Y.; Wang, X.L. Characterization of Lithium-Ion Battery Fire Emissions-Part 1: Chemical Composition of Fine Particles (PM_{2.5}). *Batteries* **2024**, *10*, 301. [\[CrossRef\]](#)
14. Willstrand, O.; Pushp, M.; Andersson, P.; Brandell, D. Impact of different Li-ion cell test conditions on thermal runaway characteristics and gas release measurements. *J. Energy Storage* **2023**, *68*, 107785. [\[CrossRef\]](#)

15. Raghibi, M.; Xiong, B.K.; Phadke, S.; Anouti, M. Role of the electrolyte in gas formation during the cycling of a Gr//NMC battery as a function of temperature: Solvent, salt, and ionic liquid effect. *Electrochim. Acta* **2020**, *362*, 137214. [\[CrossRef\]](#)
16. Scharf, J.; von Lüdgers, C.; Matysik, F.M.; Misiewicz, C.; Wandt, J.; Berg, E.J. Gas evolution in large-format automotive lithium-ion battery during formation: Effect of cell size and temperature. *J. Power Sources* **2024**, *603*, 234419. [\[CrossRef\]](#)
17. Palacin, M.R. Understanding ageing in Li-ion batteries: A chemical issue. *Chem. Soc. Rev.* **2018**, *47*, 4924–4933. [\[CrossRef\]](#) [\[PubMed\]](#)
18. Lammer, M.; Königseder, A.; Glushtitz, P.; Hacker, V. Influence of aging on the heat and gas emissions from commercial lithium ion cells in case of thermal failure. *J. Electrochem. Sci. Eng.* **2018**, *8*, 101–110. [\[CrossRef\]](#)
19. Feng, X.N.; Ouyang, M.G.; Liu, X.; Lu, L.G.; Xia, Y.; He, X.M. Thermal runaway mechanism of lithium ion battery for electric vehicles: A review. *Energy Storage Mater.* **2018**, *10*, 246–267. [\[CrossRef\]](#)
20. Gnanaraj, J.S.; Zinigrad, E.; Asraf, L.; Gottlieb, H.E.; Sprecher, M.; Schmidt, M.; Geissler, W.; Aurbach, D. A detailed investigation of the thermal reactions of LiPF₆ solution in organic carbonates using ARC and DSC. *J. Electrochem. Soc.* **2003**, *150*, A1533–A1537. [\[CrossRef\]](#)
21. Gnanaraj, J.S.; Zinigrad, E.; Asraf, L.; Gottlieb, H.E.; Sprecher, M.; Aurbach, D.; Schmidt, M. The use of accelerating rate calorimetry (ARC) for the study of the thermal reactions of Li-ion battery electrolyte solutions. *J. Power Sources* **2003**, *119*–121, 794–798. [\[CrossRef\]](#)
22. Lamb, J.; Orendorff, C.J.; Roth, E.P.; Langendorf, J. Studies on the Thermal Breakdown of Common Li-Ion Battery Electrolyte Components. *J. Electrochem. Soc.* **2015**, *162*, A2131–A2135. [\[CrossRef\]](#)
23. Maleki, H.; Deng, G.P.; Anani, A.; Howard, J. Thermal stability studies of Li-ion cells and components. *J. Electrochem. Soc.* **1999**, *146*, 3224–3229. [\[CrossRef\]](#)
24. Roth, E.P.; Doughty, D.H. Thermal abuse performance of high-power 18650 Li-ion cells. *J. Power Sources* **2004**, *128*, 308–318. [\[CrossRef\]](#)
25. Doughty, D.H.; Roth, E.P.; Crafts, C.C.; Nagasubramanian, G.; Henriksen, G.; Amine, K. Effects of additives on thermal stability of Li ion cells. *J. Power Sources* **2005**, *146*, 116–120. [\[CrossRef\]](#)
26. Yuan, L.M.; Dubaniewicz, T.; Zlochower, I.; Thomas, R.; Rayyan, N. Experimental study on thermal runaway and vented gases of lithium-ion cells. *Process Saf. Environ. Prot.* **2020**, *144*, 186–192. [\[CrossRef\]](#)
27. Abraham, D.P.; Roth, E.P.; Kostecki, R.; McCarthy, K.; MacLaren, S.; Doughty, D.H. Diagnostic examination of thermally abused high-power lithium-ion cells. *J. Power Sources* **2006**, *161*, 648–657. [\[CrossRef\]](#)
28. Fernandes, Y.; Bry, A.; de Persis, S. Thermal degradation analyses of carbonate solvents used in Li-ion batteries. *J. Power Sources* **2019**, *414*, 250–261. [\[CrossRef\]](#)
29. Eshetu, G.G.; Bertrand, J.P.; Lecocq, A.; Grugeon, S.; Laruelle, S.; Armand, M.; Marlair, G. Fire behavior of carbonates-based electrolytes used in Li-ion rechargeable batteries with a focus on the role of the LiPF₆ and LiFSI salts. *J. Power Sources* **2014**, *269*, 804–811. [\[CrossRef\]](#)
30. Liu, C.C.; Shen, W.Y.; Liu, X.Z.; Chen, Y.J.; Ding, C.; Huang, Q. Research on thermal runaway process of 18650 cylindrical lithium-ion batteries with different cathodes using cone calorimetry. *J. Energy Storage* **2023**, *64*, 107175. [\[CrossRef\]](#)
31. Zhong, G.B.; Mao, B.B.; Wang, C.; Jiang, L.; Xu, K.Q.; Sun, J.H.; Wang, Q.S. Thermal runaway and fire behavior investigation of lithium ion batteries using modified cone calorimeter. *J. Therm. Anal. Calorim.* **2019**, *135*, 2879–2889. [\[CrossRef\]](#)
32. Fu, Y.Y.; Lu, S.; Li, K.Y.; Liu, C.C.; Cheng, X.D.; Zhang, H.P. An experimental study on burning behaviors of 18650 lithium ion batteries using a cone calorimeter. *J. Power Sources* **2015**, *273*, 216–222. [\[CrossRef\]](#)
33. Huang, Z.H.; Zhao, C.P.; Li, H.; Peng, W.; Zhang, Z.; Wang, Q.S. Experimental study on thermal runaway and its propagation in the large format lithium ion battery module with two electrical connection modes. *Energy* **2020**, *205*, 117906. [\[CrossRef\]](#)
34. Liu, P.J.; Liu, C.Q.; Yang, K.; Zhang, M.J.; Gao, F.; Mao, B.B.; Li, H.; Duan, Q.L.; Wang, Q.S. Thermal runaway and fire behaviors of lithium iron phosphate battery induced by over heating. *J. Energy Storage* **2020**, *31*, 101714. [\[CrossRef\]](#)
35. Lammer, M.; Königseder, A.; Hacker, V. Holistic methodology for characterisation of the thermally induced failure of commercially available 18650 lithium ion cells. *RSC Adv.* **2017**, *7*, 24425–24429. [\[CrossRef\]](#)
36. Liu, B.H.; Jia, Y.K.; Yuan, C.H.; Wang, L.B.; Gao, X.; Yin, S.; Xu, J. Safety issues and mechanisms of lithium-ion battery cell upon mechanical abusive loading: A review. *Energy Storage Mater.* **2020**, *24*, 85–112. [\[CrossRef\]](#)
37. Guerrero-Pérez, M.O.; Patience, G.S. Experimental methods in chemical engineering: Fourier transform infrared spectroscopy-FTIR. *Can. J. Chem. Eng.* **2020**, *98*, 25–33. [\[CrossRef\]](#)
38. Laajimi, H.; Galli, F.; Patience, G.S.; Schieppati, D. Experimental methods in chemical engineering: Gas chromatography-GC. *Can. J. Chem. Eng.* **2022**, *100*, 3123–3144. [\[CrossRef\]](#)
39. Golubkov, A.W.; Scheikl, S.; Planteu, R.; Voitic, G.; Wiltsche, H.; Stangl, C.; Fauler, G.; Thaler, A.; Hacker, V. Thermal runaway of commercial 18650 Li-ion batteries with LFP and NCA cathodes—Impact of state of charge and overcharge. *RSC Adv.* **2015**, *5*, 57171–57186. [\[CrossRef\]](#)
40. Golubkov, A.W.; Planteu, R.; Krohn, P.; Rasch, B.; Brunnsteiner, B.; Thaler, A.; Hacker, V. Thermal runaway of large automotive Li-ion batteries. *RSC Adv.* **2018**, *8*, 40172–40186. [\[CrossRef\]](#)
41. Koch, S.; Fill, A.; Birke, K.P. Comprehensive gas analysis on large scale automotive lithium-ion cells in thermal runaway. *J. Power Sources* **2018**, *398*, 106–112. [\[CrossRef\]](#)

42. Zhang, Y.; Wang, H.; Li, W.; Li, C. Quantitative identification of emissions from abused prismatic Ni-rich lithium-ion batteries. *eTransportation* **2019**, *2*, 100031. [\[CrossRef\]](#)
43. Du Pasquier, A.; Disma, F.; Bowmer, T.; Gozdz, A.S.; Amatucci, G.; Tarascon, J.M. Differential scanning calorimetry study of the reactivity of carbon anodes in plastic Li-ion batteries. *J. Electrochem. Soc.* **1998**, *145*, 472–477. [\[CrossRef\]](#)
44. Diaz, F.; Wang, Y.F.N.; Weyhe, R.; Friedrich, B. Gas generation measurement and evaluation during mechanical processing and thermal treatment of spent Li-ion batteries. *Waste Manag.* **2019**, *84*, 102–111. [\[CrossRef\]](#) [\[PubMed\]](#)
45. Ravdel, B.; Abraham, K.M.; Gitzendanner, R.; DiCarlo, J.; Lucht, B.; Campion, C. Thermal stability of lithium-ion battery electrolytes. *J. Power Sources* **2003**, *119*, 805–810. [\[CrossRef\]](#)
46. Yang, H.; Zhuang, G.V.; Ross, P.N. Thermal stability of LiPF₆ salt and Li-ion battery electrolytes containing LiPF₆. *J. Power Sources* **2006**, *161*, 573–579. [\[CrossRef\]](#)
47. Kawamura, T.; Okada, S.; Yamaki, J. Decomposition reaction of LiPF₆-based electrolytes for lithium ion cells. *J. Power Sources* **2006**, *156*, 547–554. [\[CrossRef\]](#)
48. Campion, C.L.; Li, W.T.; Lucht, B.L. Thermal decomposition of LiPF₆-based electrolytes for lithium-ion batteries. *J. Electrochem. Soc.* **2005**, *152*, A2327–A2334. [\[CrossRef\]](#)
49. Wilken, S.; Treskow, M.; Scheers, J.; Johansson, P.; Jacobsson, P. Initial stages of thermal decomposition of LiPF₆-based lithium ion battery electrolytes by detailed Raman and NMR spectroscopy. *RSC Adv.* **2013**, *3*, 16359–16364. [\[CrossRef\]](#)
50. Solchenbach, S.; Metzger, M.; Egawa, M.; Beyer, H.; Gasteiger, H.A. Quantification of PF₅ and PO₂ from Side Reactions of LiPF₆ in Li-Ion Batteries. *J. Electrochem. Soc.* **2018**, *165*, A3022–A3028. [\[CrossRef\]](#)
51. Botte, G.G.; White, R.E.; Zhang, Z.M. Thermal stability of LiPF₆-EC: EMC electrolyte for lithium ion batteries. *J. Power Sources* **2001**, *97–98*, 570–575. [\[CrossRef\]](#)
52. Wang, Q.S.; Sun, J.H.; Yao, X.L.; Chen, C.H. C80 calorimeter studies of the thermal behavior of LiPF₆ solutions. *J. Solut. Chem.* **2006**, *35*, 179–189. [\[CrossRef\]](#)
53. Wilken, S.; Johansson, P.; Jacobsson, P. Infrared spectroscopy of instantaneous decomposition products of LiPF₆-based lithium battery electrolytes. *Solid State Ion.* **2012**, *225*, 608–610. [\[CrossRef\]](#)
54. Lekgoathi, M.D.S.; Vilakazi, B.M.; Wagener, J.B.; Le Roux, J.P.; Moolman, D. Decomposition kinetics of anhydrous and moisture exposed LiPF₆ salts by thermogravimetry. *J. Fluor. Chem.* **2013**, *149*, 53–56. [\[CrossRef\]](#)
55. Okamoto, Y. Ab Initio Calculations of Thermal Decomposition Mechanism of LiPF₆-Based Electrolytes for Lithium-Ion Batteries. *J. Electrochem. Soc.* **2013**, *160*, A404–A409. [\[CrossRef\]](#)
56. Yamaki, J.; Shinjo, Y.; Doi, T.; Okada, S.; Ogumi, Z. The Rate Equation of Decomposition for Electrolytes with LiPF₆ in Li-Ion Cells at Elevated Temperatures. *J. Electrochem. Soc.* **2015**, *162*, A520–A530. [\[CrossRef\]](#)
57. Tebbe, J.L.; Fuerst, T.F.; Musgrave, C.B. Mechanism of hydrofluoric acid formation in ethylene carbonate electrolytes with fluorine salt additives. *J. Power Sources* **2015**, *297*, 427–435. [\[CrossRef\]](#)
58. Bertilsson, S.; Larsson, F.; Furlani, M.; Albinsson, I.; Mellander, B.E. Lithium-ion battery electrolyte emissions analyzed by coupled thermogravimetric/Fourier-transform infrared spectroscopy. *J. Power Sources* **2017**, *365*, 446–455. [\[CrossRef\]](#)
59. Liao, Z.H.; Zhang, S.; Zhao, Y.K.; Qiu, Z.J.; Li, K.; Han, D.; Zhang, G.Q.; Habetler, T.G. Experimental evaluation of thermolysis-driven gas emissions from LiPF₆-carbonate electrolyte used in lithium-ion batteries. *J. Energy Chem.* **2020**, *49*, 124–135. [\[CrossRef\]](#)
60. Essl, C.; Golubkov, A.W.; Gasser, E.; Nachtebel, M.; Zankel, A.; Ewert, E.; Fuchs, A. Comprehensive Hazard Analysis of Failing Automotive Lithium-Ion Batteries in Overttemperature Experiments. *Batteries* **2020**, *6*, 30. [\[CrossRef\]](#)
61. Essl, C.; Golubkov, A.W.; Fuchs, A. Comparing Different Thermal Runaway Triggers for Two Automotive Lithium-Ion Battery Cell Types. *J. Electrochem. Soc.* **2020**, *167*, 130542. [\[CrossRef\]](#)
62. Golubkov, A.W.; Fuchs, D.; Wagner, J.; Wiltsche, H.; Stangl, C.; Fauler, G.; Voitic, G.; Thaler, A.; Hacker, V. Thermal-runaway experiments on consumer Li-ion batteries with metal-oxide and olivin-type cathodes. *RSC Adv.* **2014**, *4*, 3633–3642. [\[CrossRef\]](#)
63. Abbott, K.C.; Buston, J.E.H.; Gill, J.; Goddard, S.L.; Howard, D.; Howard, G.; Read, E.; Williams, R.C.E. Comprehensive gas analysis of a 21700 Li(Ni_{0.8}Co_{0.1}Mn_{0.1}O₂) cell using mass spectrometry. *J. Power Sources* **2022**, *539*, 231585. [\[CrossRef\]](#)
64. Cai, T.; Valecha, P.; Tran, V.; Engle, B.; Stefanopoulou, A.; Siegel, J. Detection of Li-ion battery failure and venting with Carbon Dioxide sensors. *Etransportation* **2021**, *7*, 100100. [\[CrossRef\]](#)
65. Lamb, J.; Orendorff, C.J.; Steele, L.A.M.; Spangler, S.W. Failure propagation in multi-cell lithium ion batteries. *J. Power Sources* **2015**, *283*, 517–523. [\[CrossRef\]](#)
66. Huang, S.; Du, X.N.; Richter, M.; Ford, J.; Cavalheiro, G.M.; Du, Z.J.; White, R.T.; Zhang, G.S. Understanding Li-Ion Cell Internal Short Circuit and Thermal Runaway through Small, Slow and In Situ Sensing Nail Penetration. *J. Electrochem. Soc.* **2020**, *167*, 090526. [\[CrossRef\]](#)
67. Lamb, J.; Orendorff, C.J. Evaluation of mechanical abuse techniques in lithium ion batteries. *J. Power Sources* **2014**, *247*, 189–196. [\[CrossRef\]](#)
68. Koch, S.; Birke, K.P.; Kuhn, R. Fast Thermal Runaway Detection for Lithium-Ion Cells in Large Scale Traction Batteries. *Batteries* **2018**, *4*, 16. [\[CrossRef\]](#)
69. Hoelle, S.; Scharner, S.; Asanin, S.; Hinrichsen, O. Analysis on Thermal Runaway Behavior of Prismatic Lithium-Ion Batteries with Autoclave Calorimetry. *J. Electrochem. Soc.* **2021**, *168*, 120515. [\[CrossRef\]](#)
70. Nedjalkov, A.; Meyer, J.; Koehring, M.; Doering, A.; Angelmahr, M.; Dahle, S.; Sander, A.; Fischer, A.; Schade, W. Toxic Gas Emissions from Damaged Lithium Ion Batteries-Analysis and Safety Enhancement Solution. *Batteries* **2016**, *2*, 5. [\[CrossRef\]](#)

71. Yuan, Q.F.; Zhao, F.G.; Wang, W.D.; Zhao, Y.M.; Liang, Z.Y.; Yan, D.L. Overcharge failure investigation of lithium-ion batteries. *Electrochim. Acta* **2015**, *178*, 682–688. [CrossRef]
72. Global Technical Regulation No. 20 Electric Vehicle Safety (EVS). 2018. Available online: <https://unece.org/fileadmin/DAM/trans/main/wp29/wp29wgs/wp29gen/wp29registry/ECE-TRANS-180a20e.pdf> (accessed on 12 October 2024).
73. Yang, M.; Rong, M.; Pan, J.; Ye, Y.; Yang, A.; Chu, J.; Yuan, H.; Wang, X. Thermal runaway behavior analysis during overheating for commercial LiFePO₄ batteries under various state of charges. *Appl. Therm. Eng.* **2023**, *230*, 120816. [CrossRef]
74. Robles, D.J.; Jeevarajan, J. Fire and gas characterization studies for lithium-ion cells and batteries. In Proceedings of the 2020 NASA Aerospace Battery Workshop, Huntsville, AL, USA, 17–19 November 2020.
75. Sturk, D.; Rosell, L.; Blomqvist, P.; Tidblad, A.A. Analysis of Li-Ion Battery Gases Vented in an Inert Atmosphere Thermal Test Chamber. *Batteries* **2019**, *5*, 61. [CrossRef]
76. Gully, B. *Technical Reference for Li-ion Battery Explosion Risk and Fire Suppression*; 2019-1025, Rev. 4; DNV GL AS Maritime: Høvik, Norway, 2019.
77. Maloney, T. *Lithium Battery Thermal Runaway Vent Gas Analysis*; U.S. Department of Transportation: Washington, DC, USA, 2016.
78. Bergström, U.; Gustafsson, Å.; Hägglund, L.; Lejon, C.; Sturk, D.; Tengel, T. *Vented Gases and Aerosol of Automotive Li-ion LFP and NMC Batteries in Humidified Nitrogen under Thermal Load*; FOI-R—4166-SE; Försvarets Forskningsinstitut FOI: Stockholm, Sweden, 2015.
79. Zhang, J.; Guo, Q.; Liu, S.; Zhou, C.; Huang, Z.; Han, D. Investigation on gas generation and corresponding explosion characteristics of lithium-ion batteries during thermal runaway at different charge states. *J. Energy Storage* **2024**, *80*, 110201. [CrossRef]
80. Amano, K.O.A.; Hahn, S.-K.; Butt, N.; Vorwerk, P.; Gimadieva, E.; Tschirschwitz, R.; Rappsilber, T.; Krause, U. Composition and Explosibility of Gas Emissions from Lithium-Ion Batteries Undergoing Thermal Runaway. *Batteries* **2023**, *9*, 300. [CrossRef]
81. Amano, K.O.A.; Hahn, S.K.; Tschirschwitz, R.; Rappsilber, T.; Krause, U. An Experimental Investigation of Thermal Runaway and Gas Release of NMC Lithium-Ion Pouch Batteries Depending on the State of Charge Level. *Batteries* **2022**, *8*, 41. [CrossRef]
82. Chen, S.C.; Wang, Z.R.; Wang, J.H.; Tong, X.; Yan, W. Lower explosion limit of the vented gases from Li-ion batteries thermal runaway in high temperature condition. *J. Loss Prev. Process Ind.* **2020**, *63*, 103992. [CrossRef]
83. Crafts, C.C.; Doughty, D.H.; McBreen, J.; Roth, E.P. *Advanced Technology Development Program for Lithium-Ion Batteries: Thermal Abuse Performance of 18650 Li-ion Cells*; Sandia National Laboratories (SNL): Albuquerque, NM, USA; Livermore, CA, USA, 2004.
84. Abbott, K.C.; Buston, J.E.H.; Gill, J.; Goddard, S.L.; Howard, D.; Howard, G.E.; Read, E.; Williams, R.C.E. Experimental study of three commercially available 18650 lithium ion batteries using multiple abuse methods. *J. Energy Storage* **2023**, *65*, 107293. [CrossRef]
85. Kennedy, R.W.; Marr, K.C.; Ezekoye, O.A. Gas release rates and properties from Lithium Cobalt Oxide lithium ion battery arrays. *J. Power Sources* **2021**, *487*, 229388. [CrossRef]
86. Sun, J.; Li, J.; Zhou, T.; Yang, K.; Wei, S.; Tang, N.; Dang, N.; Li, H.; Qiu, X.; Chen, L. Toxicity, a serious concern of thermal runaway from commercial Li-ion battery. *Nano Energy* **2016**, *27*, 313–319. [CrossRef]
87. Larsson, F.; Andersson, P.; Blomqvist, P.; Mellander, B.E. Toxic fluoride gas emissions from lithium-ion battery fires. *Sci. Rep.* **2017**, *7*, 10018. [CrossRef] [PubMed]
88. Willstrand, O.; Pushp, M.; Ingason, H.; Brandell, D. Uncertainties in the use of oxygen consumption calorimetry for heat release measurements in lithium-ion battery fires. *Fire Saf. J.* **2024**, *143*, 104078. [CrossRef]
89. Li, W.F.; Wang, H.W.; Zhang, Y.J.; Ouyang, M.G. Flammability characteristics of the battery vent gas: A case of NCA and LFP lithium-ion batteries during external heating abuse. *J. Energy Storage* **2019**, *24*, 11. [CrossRef]
90. Wang, Y.; Song, Z.; Wang, H.; Li, H.; Zhang, Y.; Li, C.; Zhang, X.; Feng, X.; Lu, L.; Ouyang, M. Experimental research on flammability characteristics and ignition conditions of hybrid mixture emissions venting from a large format thermal failure lithium-ion battery. *J. Energy Storage* **2023**, *59*, 106466. [CrossRef]
91. Alexandrino, K.; Alzueta, M.U.; Curran, H.J. An experimental and modeling study of the ignition of dimethyl carbonate in shock tubes and rapid compression machine. *Combust. Flame* **2018**, *188*, 212–226. [CrossRef]
92. Henriksen, M.; Vaagsaether, K.; Lundberg, J.; Forseth, S.; Bjerketvedt, D. Laminar burning velocity of gases vented from failed Li-ion batteries. *J. Power Sources* **2021**, *506*, 230141. [CrossRef]
93. Henriksen, M.; Vaagseather, K.; Gaathaug, A.V.; Lundberg, J.; Forseth, S.; Bjerketvedt, D. Laminar Burning Velocity of the Dimethyl Carbonate-Air Mixture Formed by the Li-Ion Electrolyte Solvent. *Combust. Explos.* **2020**, *56*, 383–393. [CrossRef]
94. Wang, Q.; Jiang, L.; Yu, Y.; Sun, J. Progress of enhancing the safety of lithium ion battery from the electrolyte aspect. *Nano Energy* **2019**, *55*, 93–114. [CrossRef]
95. García, A.; Micó, C.; Marco-Gimeno, J.; Bernal, A. A tabulation method of Li-ion Thermal Runaway mechanisms for the acceleration of high dimensional simulations. *J. Energy Storage* **2024**, *102*, 113982. [CrossRef]
96. García, A.; Monsalve-Serrano, J.; Gimeno, J.M.; Egea, J.M.H. Experimental measurement and modeling of the internal pressure in cylindrical lithium-ion battery cells under abuse conditions. *J. Energy Storage* **2024**, *103*, 114288. [CrossRef]

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