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 μg/m³ 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 $PM_{2.5}$ 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 $PM_{2.5}$ mass concentrations; thus, compliance with $PM_{2.5}$ 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 $PM_{2.5}$ 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 $PM_{2.5}$ mass. Including BC/EC as an indicator of local emission reductions might compensate for the limited ability to influence total $PM_{2.5}$ 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 0.2–0.4 μg/m³ in rural sites, 1.0–2.0 μg/m³ in urban background stations and 1.4–5.1 μg/m³ 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 $PM_{2.5}$. 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 μg/m³ to 3.9 μg/m³, while for elemental carbon the means generally ranged from 0.47 μg/m³ to 3.5 μg/m³ and reached 7.5–8.8 μg/m³ 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 1.08–1.15 μg/m³ for black carbon and 0.5–0.8 μg/m³ 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 µm 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/cm^3) to 10% for typical urban background concentrations (about 10 000 particles/cm^3) (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 (PM_{10}, $PM_{2.5}$), 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/cm^3 (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/cm^3) or 1-hour mean concentrations exceeding levels found usually in all urban microenvironments (20 000 particles/cm^3) 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 PM_{10}. 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 PM_{10} 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 PM_{10} 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 PM_{10} 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 PM_{10} 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 PM_{10} 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 $RBPM_{10}$ and $RBPM_{2.5}$ levels without the dust contribution.
- Determine the net dust PM (NDPM) levels in PM_{10} and $PM_{2.5}$ ($NDPM_{10}$ and $NDPM_{2.5}$) for the regional background by subtracting $RBPM_{10}$ and $RBPM_{2.5}$ from the bulk PM_{10} and $PM_{2.5}$ levels at the reference regional background-monitoring site.
- At the nearby urban area, $NDPM_{10}$ 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 $NDPM_{10}$ and $NDPM_{2.5}$ values from the urban PM_{10} and $PM_{2.5}$ levels, are the APM loads during the dust days (APM_{10} 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 PM_{10} reductions observed in most reviewed studies. PM_{10} reductions varied within 7–30% of the daily mean PM_{10} 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. The verification code for this document is 201671