Interactive effects of changing stratospheric ozone and climate on tropospheric composition and air quality, and the consequences for human and ecosystem health

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Abstract

The composition of the air we breathe is determined by emissions, weather, and photochemical transformations induced by solar UV radiation. Photochemical reactions of many emitted chemical compounds can generate important (secondary) pollutants including ground-level ozone (O_3) and some particulate matter, known to be detrimental to human health and ecosystems. Poor air quality is the major environmental cause of premature deaths globally, and even a small decrease in air quality can translate into a large increase in the number of deaths. In many regions of the globe, changes in emissions of pollutants have caused significant changes in air quality. Short-term variability in the weather as well as long-term climatic trends can affect ground-level pollution through several mechanisms. These include large-scale changes in the transport of O_3 from the stratosphere to the troposphere, winds, clouds, and patterns of precipitation. Long-term trends in UV radiation, particularly related to the depletion and recovery of stratospheric ozone, are also expected to result in changes in air quality as well as the self-cleaning capacity of the global atmosphere. The increased use of substitutes for ozone-depleting substances, in response to the Montreal Protocol, does not currently pose a significant risk to the environment. This includes both the direct emissions of substitutes during use and their atmospheric degradation products (e.g. trifluoroacetic acid, TFA).

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Summary

The composition of the air we breathe is determined by emissions, weather, and photochemical transformations induced by solar UV radiation. Photochemical reactions of many emitted chemical compounds can generate important (secondary) pollutants including ground-level ozone (O_3) and some particulate matter, known to be detrimental to human health and ecosystems. Poor air quality is the major environmental cause of premature deaths globally, and even a small decrease in air quality can translate into a large increase in the number of deaths. In many regions of the globe, changes in emissions of pollutants have caused significant changes in air quality. Short-term variability in the weather as well as long-term climatic trends can affect ground-level pollution through several mechanisms. These include large-scale changes in the transport of O_3 from the stratosphere to the troposphere, winds, clouds, and patterns of precipitation. Long-term trends in UV radiation, particularly related to the depletion and recovery of stratospheric ozone, are also expected to result in changes in air quality as well as the self-cleaning capacity of the global atmosphere. The increased use of substitutes for ozone-depleting substances, in response to the Montreal Protocol, does not currently pose a significant risk to the environment. This includes both the direct emissions of substitutes during use and their atmospheric degradation products (e.g. trifluoroacetic acid, TFA).

1   Introduction

The composition (and the quality) of the air we breathe is critical for life on Earth and is affected by natural and anthropogenic processes throughout the biosphere as well as by solar radiation (**Fig. 1**). In addition to gases, the atmosphere contains many types of particulate matter (PM), that have been both naturally generated (e.g., sea-salt, dust) and anthropogenic (e.g., photochemical smog). These gases and particles circulate in the troposphere (lower atmosphere) and are changed by chemical and physical processes. A key driver of these processes is the amount of UV radiation transmitted through stratospheric O_3 to the troposphere. Changes in intensity of UV radiation in the troposphere, particularly near the ground, will affect the composition of the atmosphere and change the quality of the air that organisms depend on. These changes can be both harmful and beneficial. For example, UV radiation accelerates the removal of many chemical compounds that are emitted into the

atmosphere by human activities. These UV-driven chemical reactions can generate transient products (such as ground level ozone and particulate matter) that degrade air quality before they are removed. Climate change will also affect atmospheric circulation (wind patterns) and will interact with solar radiation to modify the composition of the atmosphere.

Insert Fig 1 after this point

Poor air quality has large direct and indirect effects on human health, as well as on terrestrial and aquatic ecosystems. Exposure to polluted air has been associated with a variety of adverse effects in humans; however, identifying associations and causation between effects and a chemical within the mix of gases and particles is difficult.

Adverse effects of poor air quality on the environment include for example, damage to trees other plants, and agriculture, with potentially significant impacts on crop production and hence food security. The effect can also be indirect, whereby air-borne pollutants can change the composition of the soil and water, which can then affect living organisms. Vegetation can, in turn, improve or degrade air quality through multiple complex interactions with the atmosphere.

The World Health Organization estimates that poor air quality is now the largest environmental cause of mortality worldwide, exceeding that of poor water quality.[2] An important distinction should be made between the quality of air indoors, and that outdoors for which UV radiation plays a significant role. Poor indoor air quality, coming from sources such as cooking and heating, has long been recognized as a significant concern for human health, and recent estimates suggest that the health impacts of indoor air have declined (**Fig. 2**). Based on the number of deaths globally, the impact of outdoor air pollution on human health has increased. The relationship between concentrations of atmospheric components and human health is widely recognized but quantification is still a matter of significant uncertainty and study.

Insert Fig 2 after this point

Fig. 1 Global annual deaths from environmental risk factors. Risk factors shown are primarily from the Global Burden of Disease (GBD) Study (ref.[3]). These are ambient particulate matter (PM) pollution, household air pollution from solid fuels where the major impact is indoors (Indoor), Poor water quality (Water) and ambient ozone pollution (O_3). Also shown is an alternate estimate of deaths from ambient O_3 pollution (O_3 Malley),[4] highlighting the differences in current estimates of human sensitivity to O_3.

Outdoor air quality is most often defined by key atmospheric components that have a substantial impact (criteria pollutants). For this assessment, the two measures of air quality that will be mainly considered are the ground-level concentrations of ozone (O_3) and particulates (more specifically particulate matter smaller than 2.5 micrometers ($PM_{2.5}$)), since both are sensitive to changes in stratospheric O_3 and climate. There are several other criterial pollutants that are important to air quality that will not be considered in detail (such as sulfur and nitrogen oxides).

This assessment is predicated on the understanding that the changes in air quality at a particular location are associated with changes in stratospheric ozone depletion and climate that are driven by three key influences (**Fig. 1**):

1. Chemical changes primarily driven by solar UV radiation, particularly UV-B radiation.

2. Pollutants emitted locally (e.g., by motor vehicles, trees), regionally (e.g., emissions from other cities) or globally (e.g., stratospheric O_3, which is not a pollutant until it is transported to the ground).

3. Atmospheric characteristics, i.e., weather, in terms of temperature, wind (transporting pollutants), rain, and cloudiness, which over longer time frames constitute climate.

The production of surface O_3 by UV-driven photochemistry is illustrated in **Fig. 3** using a simplified reaction scheme. Ozone is produced by the UV-driven photooxidation of hydrocarbons in the presence of nitrogen oxides ($NO_x = NO + NO_2$). These chemical transformations cause more O_3 to be formed, generate PM, and aid in the final removal of pollutants from the atmosphere. The actual rate of production of O_3 and PM depends on the available amounts of UV radiation, hydrocarbons, and NOx, and therefore the exposure of humans and vegetation is expected to vary significantly with time and location.

Insert Fig 3 after this point

Here, we assess the changes in air quality due to changes in stratospheric O_3 combined with climate change, emphasizing the new understanding since the last United Nations Environment Program Environmental Effects Assessment Panel quadrennial report.[5] It should be noted that the chemistry in the urban atmosphere can be quite different to that in rural areas.[6] What will not be directly assessed is the effectiveness of regulation, i.e., air quality controls and the relative merits of various air quality standards. However, it is worth noting that substantial progress has been made in controlling avoidable emissions of hydrocarbons from sources like motor vehicles in developed countries (e.g., ref.[7]).

2   Atmospheric circulation and air quality

Physical transport of air masses around the globe is critical for determining air quality for all environments. For air quality, this can be separated into two large-scale processes. The first is the transport of air from one continent to another. The second is the downward transport of O_3 from the stratosphere to the Earth's surface, which is a key process that determines the baseline concentration of O_3, that is, the concentration of O_3 for a location without local or regional anthropogenic inputs. Climate-driven changes in environmental conditions (e.g., temperature, humidity, precipitation) affect both the production and removal of pollutants, including O_3 and PM, and must therefore also be considered.

2.1   Stratospheric-tropospheric exchange

The exchange of air between the stratosphere and troposphere can change tropospheric concentrations of O_3. In a relatively clean atmosphere, this effect may be quantifiable. Ozone-sonde profiles of the concentration of O_3 have been collected since 1987 in New Zealand. These have been used to evaluate atmospheric models. Using these models, it has then been estimated that, between 1960 and 2010, there has been a decrease in concentration of 4 µg O_3 m^{-3} in the lowest 1.5 km of the atmosphere (near the ground) that was attributed to stratospheric O_3. That is, this decrease in concentration of O_3 at ground level is the combined effect of decreases in stratospheric O_3 due to ozone depleting substances (ODS) and changes in transport from the stratosphere.[8] Such an attribution is very difficult from ground-based measurements in the Northern Hemisphere (e.g., ref.[9]) because of the large variations in sources. However, more accurate estimates are possible using ozonesondes measuring at 5 km above the surface and at high altitude sites well away from local pollution sources and hence also much closer to the stratospheric source as well.[10] From an analysis of

measurements made at the Jungfraujoch in Switzerland (3.6 km altitude, 1988–2008) and a chemical transport computer model, it is estimated that 30% of the O_3 is attributable to stratospheric sources, and 95% of the variation at a monthly timescale is attributed to variations in stratospheric input. Trends in the measurements for the 20-year period are variable and not well captured by the computer simulation.[10] A study of measurements over the USA found evidence that enhanced ground-level O_3 followed specific meteorological conditions indicating enhanced transport of stratospheric air to the surface.[11] Model calculations suggest that significant changes (+ 50%) in stratospheric-tropospheric O_3 exchange could occur by 2100, as a result of changes in both climate and ODS.[12]

2.2   Effects of climate change on air quality

Air quality is directly affected by several environmental conditions including temperature, humidity, clouds and precipitation, local wind speed and long-range transport by winds, as well as land and vegetation type (e.g., urban *vs* forested settings). Many of these conditions are sensitive to both climate change and stratospheric O_3 depletion. For example, stratospheric O_3 depletion has already caused large changes in the weather of the Southern Hemisphere,[13] with probable (but as yet unquantified) effects on urban and regional air quality.

Increasing global temperatures are expected to lead to poorer air quality. Current observations point to a co-occurrence of elevated temperatures, high O_3, and high $PM_{2.5}$ at ground level in the eastern USA, suggesting a direct link between elevated temperatures and poorer air quality.[14] Similarly, in China, it has been estimated that the severe haze events with significant health impacts are correlated with lower wind speeds, driven by climate change.[15]

Modelling the impact of future climate change on air quality is difficult. Firstly, there is a need to consider the various drivers of change, such as changes in greenhouse gas concentrations and, for the southern hemisphere, recovery from the spring-time depletion of stratospheric O_3.[16] Then, for reliable estimates of impacts, it is necessary for the model to capture changes in wind speed and direction, temperature, rainfall, and cloudiness. Estimates of the impacts are known to be affected by things like the assumed building heights within the urban environment.[17] Ideally, estimates of likelihood of wildfires also need to be included.[18]

Recent estimates have been made for India for the period up to 2050, suggesting increases in surface O_3 in the north due to climate change.[13, 19, 20] These changes are attributed to changes in sources and sinks of O_3. For India, concentrations of $PM_{2.5}$ are predicted to increase at all latitudes due to climate change,[19] although the predicted changes depend on changes in human activity.[20] For Europe, an ensemble of models predict, for 2050, decreases in concentrations of surface O_3 in summer and increases in winter, with the magnitude (and significance of the trends) dependent upon the magnitude of the changes in emissions.[21] Similar general trends were predicted for deaths from O_3 in the USA.[22]

3   Effects of UV radiation on air quality

UV radiation drives photochemical reactions that transform emitted chemicals, such as hydrocarbons and nitrogen oxides, into more toxic secondary pollutants such as O_3 and particulate matter (PM). While emissions are obviously of primary importance, the rate of formation of secondary pollutants is limited by the availability of UV photons (see **Fig. 3**). UV radiation also regulates the self-cleaning ability of the troposphere by forming highly reactive hydroxyl radicals (OH) that react with many pollutants and accelerate their removal from the atmosphere.

3.1 UV radiation and ozone at the Earth's surface

Ozone has long been recognized as a significant concern in both urban and rural environments. A major review of the observations and trends has been undertaken.[23-29] We highlight some of the results of these assessments, as well as the relevance of stratospheric O_3 changes to ground level ozone.

The assessment of the importance of any changes in O_3 at ground level depends on who (or what) is exposed. A range of metrics have been developed and adopted in various regions. For short-term human exposure, one metric used is the 4th highest daily maximum 8-hour running mean (4MDA8). For plants, a cumulative measure of O_3 exposure is generally accepted as being more appropriate, with an "accumulated exposure over a 40 ppbv (80 µg m^{-3}) threshold" (= AOT40) is often used, typically calculated from hourly averages for the growing season.[30]

Concentrations of O_3 at ground level have changed significantly in recent years, with the direction and extent of the change being dependent on location, broad regions showing decreases and increases, as shown in **Fig. 4**.[24] These changes are primarily a response to changes in emissions of precursors to O_3 (primarily changes in NO_x and volatile organic compounds). There have been increases in emissions where there has been rapid economic growth and decreases with the implementation of emission controls.

Insert Fig 4 after this point

Fig. 4 Trends in concentration of O_3 at ground level for the 15-year period (2000 - 2014), measuring O_3 with the human health related metric 4MDA8. The top panels on both sides are for North America, the lower left is for Europe and the lower right side is East Asia. Blue arrows indicate a decrease in concentration of O_3 and red an increase (from ref.[24]). The dark colours (red/blue) indicate a trend that is significant at the 95% confidence level. Lighter shades indicate lower levels of significance.

The distribution of concentrations of O_3 by site is also changing as emissions (and emission controls) change. As an example, **Fig. 5** shows the changes in concentrations of surface O_3 observed in Berlin, a pattern of change seen in many locations where the yearly average concentration of surface O_3 is decreasing. There is a clear decrease in events with high concentrations of O_3 and an increase in the number of periods with more moderate concentrations. There is also a decrease in the number of periods of low concentrations of O_3 (< 20 µg m^{-3}). These periods of low concentrations of O_3 were caused by the reaction of O_3 with NO_x, and there has been a reduction in the concentration of NOx in these urban atmospheres. In regions where increases in precursor emissions dominate, the opposite pattern (increased frequency of periods with low and high concentrations of O_3) is observed.

Insert Fig 5 after this point

Fig. 5 Changes in observed hourly average ozone concentration at ground level for Berlin for the period 1990 to 2013. Observations have been sorted into 5 ppb wide "bins". Plotted is the trend in the number of observations within each bin. The colour indicates the probability (p) that the change is not significant (from ref.[31]).

The ability of computer models to replicate observed concentrations of surface O_3 has been limited, with models predicting lower concentrations of O_3 in winter than observed and higher concentrations in summer (by up to 20 µg m^{-3}), particularly in N America.[32] The model biases in winter are believed to be due to not including the impact of long-range transport of air.[33, 34] A recent modelling study suggests that 66% of the higher concentrations in summer could be due to models not adequately describing the impact of the forest canopy

on reducing solar radiation and modifying movement of air.[34] This suggests that forecasts of concentrations of O_3 at ground level could soon become more robust.

3.1.1 Future changes in surface ozone

Future concentrations of O_3 at ground level will depend on a range of environmental factors and the relative importance of these factors will depend on location. Estimates indicate that, due to climate change, there will be an increase in O_3 in the lower atmosphere at most latitudes, with the largest changes being due to an increase in the concentrations of NO_x driven by an increased frequency of lightning in the tropics.[35, 36] The absolute magnitude of these changes in surface O_3 will depend on the increases in the concentrations of greenhouse gases. In addition to this, an increase in tropospheric O_3 of similar magnitude is expected due to downward transport of stratospheric air (see section 2.1) that will be altered by the recovery of stratospheric O_3 through the phase-out of ODS[36] and the response of stratospheric O_3 to climate change.[37] In northern mid-latitudes, these changes will be largely offset by a reduction in surface O_3 due to a reduction in precursors of O_3 from the control of emissions implemented in many countries. However, for the tropics, the background O_3 concentration is expected to increase by around 10% by 2050. These global scale changes help define the underlying concentration of O_3 that is experienced in urban and regional areas. More local sources (and sinks) of O_3 then determine the concentrations that humans and plants are exposed to.

3.1.2 Modulation of tropospheric ozone by changes in UV radiation

If, as expected with the ongoing success of the Montreal Protocol, amounts of O_3 increase in the stratosphere, reactions in the troposphere dependent upon UV radiation will slow down, thereby decreasing both the production and destruction of surface O_3 (see also **Fig. 3**).[38] Coarse resolution spatial modeling (4° x 5°, roughly 400 x 400 km at mid-latitudes) predicted this will cause an increase in surface O_3 over the USA.[39] However, it is expected that there will be areas within cities where the slower production should reduce concentrations of O_3.[40] Running a model of atmospheric chemistry at higher spatial resolution (12 km x 12 km) over the contiguous USA,[38] highlights that the impact of increases in stratospheric O_3 on ground level O_3 is not uniform, with decreases in the concentration of O_3 at locations in or near major cities, and increases elsewhere (**Fig. 6**), with a population-weighted increase in concentrations of O_3 overall. The increase outside cities is small (about 1 µg m^{-3}) but represents an additional negative effect on air quality. Effects on human health will depend on the balance between the direct impact of decreased UV radiation on health[41] and reduced air quality, both impacts which involve significant uncertainties.[42]

Insert Fig 6 after this point

Fig. 6 Changes in ground-level O_3 because of a change from current (2000 – 2010) stratospheric ozone to that predicted for 2085, calculated for a 12 x 12 km grid (from ref.[38]).

3.2 Particulates and UV radiation

Particulate matter (PM) suspended in the atmosphere is a major component of air pollution. Also known by other names such as aerosols, haze, and smog (**Fig. 7A**), elevated concentrations of PM have been associated with adverse effects on human health (see below), degradation of visibility (as they scatter light), modification of the hydrological cycle

(through interactions with clouds and precipitation (e.g., ref.[43])), and reduction in efficiency of photovoltaic solar collectors.[44] Particles are generally complex mixtures of different components (see **Fig. 7B**) that include both primary (e.g. soot) and secondary (e.g. sulfates and organics) compounds. A substantial fraction of PM is influenced by the chemical and physical state of the atmosphere and is therefore sensitive to changes in environmental variables including UV radiation, temperature, winds, and humidity.

Fig. 7 (A) Photochemical haze over Mexico City seen from approaching aircraft. (**B**) Electron microscopy image of a typical particle collected from the photochemical haze (from ref.[1]).

Insert Fig 7 after this point

It is useful to distinguish between PM emitted directly from sources at the Earth's surface (primary aerosols), and those formed in the atmosphere by condensation of various gases (secondary aerosols). These secondary aerosols include sulfate, nitrate, and a multitude of organic compounds that condense following atmospheric reactions of emitted precursor gases such as SO_2, NO, and hydrocarbons.[45] Primary aerosols include dust/soils, sea salt, biological particles (e.g., pollen) and soot, and other carbonaceous particles produced during combustion of fossil fuels and wildfires. Natural and anthropogenic sources contribute to both primary and secondary aerosols, e.g., organic aerosols may be formed from emissions of isoprene and terpenoid compounds from plants, as well as from fugitive emissions from industrial and transportation sectors.

UV radiation has a direct role in the formation of secondary aerosols by producing hydroxyl (OH) radicals that oxidize (harmful) precursors to chemicals that will more readily condense into particles, e.g., SO_2 to H_2SO_4 or NO to HNO_3. Increases in UV radiation lead to more rapid formation of secondary PM, resulting in more intense local smog episodes. Changes in weather patterns (and hence climate) can affect both primary and secondary aerosols, e.g., by changes in wind-driven transport patterns, by temperature effects on condensation and/or evaporation, uptake of water (deliquescence/efflorescence) on particles, and eventual removal by incorporation into precipitation.

3.2.1 Distribution and trends of aerosols

Measurements of aerosols are made routinely in many cities and regions as part of observational networks monitoring air quality, using a variety of analysis techniques (e.g. refs[46-49]). In recent years, estimates of the distribution of aerosols over large geographical scales have become available from satellite-based instruments.[50-52] In addition to providing a global climatology of aerosols,[50] this has allowed more detailed assessment of aerosols in specific regions that had previously only sparse measurements, e.g., over China,[53] the Mediterranean Sea,[54] India,[55] and the Saharan desert.[56] Combined with ground-based measurements and numerical models, these observations clearly show the contribution to PM from sulfate emitted by urban and industrial activities, black and organic carbon mostly from biomass burning, and wind-driven dust and sea-salt (see **Fig. 8**).

Insert Fig 8 after this point

Long-term trends in aerosols depend strongly on geographic location and reflect several regional factors, such as economic development and emission controls for aerosols and their precursors. This geographic dependence is illustrated in **Fig. 9** for the period 1990-2015. Strong positive trends are seen over China and the Indian subcontinent (see also ref.[57]), with rates of increase reaching 3% per year (change in optical depth of 0.3 in 25 years), while negative trends of similar magnitude are seen over central Europe and the Eastern USA.[58, 59] The changes in PM over Europe have been calculated to have increased surface O_3 through enhanced photolysis and increased biogenic volatile organic compound (BVOC) emissions.[60] The trends observed are also reflected in changes in aerosol properties, as the dominant source of the particles changes.[61] These trends in aerosol loading and composition have important implications for health effects as discussed below.

Insert Fig 9 after this point

Fig. 9 Changes (1990 to 2015) in the aerosol column optical depth at 550 nm, computed as the mean of six global models (from ref.[59]).

In calculating conditions within extreme aerosol pollution events, Li et al.[62] found significant changes in UV intensity at ground level that altered the chemical composition and hence air quality. Validation of the models with observations in cases like this should lead to higher confidence in estimates of changes in air quality due to changes in UV radiation, and hence the impact of ozone recovery on air quality.

3.3 UV radiation and the atmospheric global oxidation capacity

As noted before, the hydroxyl radical is a highly reactive molecule that initiates the chemical transformation of many gases in the atmosphere (e.g., volatile organic compounds). This transformation typically accelerates the removal of these gases from the atmosphere, although some of the short-lived intermediate products formed can also be harmful (e.g. secondary aerosol). Hydroxyl radicals may be formed in several ways in the atmosphere, but the main route is via the photolysis of O_3 by UV radiation with a wavelength less than 330 nm (**Fig. 3**).

The concentrations of OH are highly variable in both space and time because of these production- and destruction-reactions. At the same time, however, a change in the global average concentration of OH can have a significant impact upon the removal of many

important air pollutants, as well as some greenhouse gases (e.g., methane) and ozone-depleting substances (e.g., HCFCs).

Because of the importance of OH for the chemistry of the atmosphere and in particular for air quality, there have been several approaches used to assess changes in the global concentrations of tropospheric OH. The most widely used has involved the assessment of trends in the concentration of methyl chloroform (1,1,1-trichloroethane), which is a synthetically produced compound that has been regulated under the Montreal Protocol. Methyl chloroform is primarily removed from the atmosphere through reaction with OH. Given a reasonable estimate of the amount of methyl chloroform released and a well-known atmospheric concentration, the removal rate (and hence the concentration of OH) can be inferred (e.g., ref.[63]). One limitation of this method is that the concentration of methyl chloroform is rapidly decreasing due to the implementation of the Montreal Protocol and soon concentrations will be too low to be useful for estimating concentrations of OH.[64]

Estimates have also been derived from the observed concentrations of methane, which is also primarily removed from the atmosphere by OH. However, this method is hindered by a lack of knowledge of the magnitude and variability of sources of methane.[65] A recent summary of the difficulties in quantifying the methane budget and therefore global concentrations of OH recognizes the critical need for more work in this area.[66] An alternative method has been proposed that uses four halocarbon compounds to estimate the global concentrations of OH (HFC-32 (CH_2F_2), HFC-134a (CH_2FCF_3), HFC-152a (CH_3CHF_2), and HCFC-22 ($CHClF_2$)).[64] In their initial assessment, the authors found that the combination currently did as well as methyl chloroform alone and may provide a useful method in the years to come as the concentration of methyl chloroform continues to decrease.

Model estimates of trends in concentration of OH are problematic due to uncertainty in the chemical reactions that produce OH and the sources and emissions of precursors. There also remain significant uncertainties in the rates of key chemical and photochemical reactions.[67, 68] This is evidenced by the difficulty models have in predicting the correct ratio of OH between the hemispheres.[69] A recent study of the atmosphere over the tropical West Pacific found that estimates of concentrations of OH in the troposphere could be changed by 20–30 % (in both directions) by correcting the concentrations calculated for key precursors to those observed in the field.[70] A recent model of global OH suggests that the generation of OH from the oxidation of biogenic volatile organic compounds (BVOCs) has been underestimated.[71] This would make the concentration of OH in the atmosphere much less dependent upon anthropogenic emissions, and significantly alter the predictions of future atmospheric composition. These BVOC emissions also may be stimulated by exposure to UV-B radiation.[72, 73] Future BVOC emissions will be altered by changes in land-use and climate change.[73, 74]

4   Impacts of air pollution on human and environmental health

Exposure to air pollution is known to be detrimental to the health of humans as well as natural and cultivated ecosystems. The direct effects of exposure to UV radiation on human health and plants (including terrestrial organisms and crops) are discussed by Lucas et al.[41] and by Bornman et al.,[73] respectively. Here, we focus on how two principal pollutants, surface O_3 and PM, whose UV-dependent formation was discussed above, are considered to affect human health and agricultural yields. For human impacts, we note the existence of both large-scale assessments, carried out either by government agencies or others (NGOs), as well as a large growing body of scientific literature based on analyses ranging from molecular to clinical and epidemiological scales.

4.1 Adverse effects of poor air quality on human health

4.1.1 Global and regional assessments

The impact of PM on human health is now widely acknowledged, although quantitative estimates vary considerably among studies. **Table 1** summarizes premature mortality derived by numerous large-scale studies. Global estimates range by a factor of three, from 1.4 to 4.2 million deaths per year.

Insert Table 1 after this point

Table 1. Estimates of global and regional mortality for particulate matter and O_3, million per year.

Source	Year	Mortality per year (millions)		Region
		Particulate matter	O_3	
OECD [75]	2010	1.4	0.35	Global
	2050	3.6	0.75	Global
Fang et al. 2015[76]	2000	1.5	0.38	Global
Lelieveld et al. 2015[77]	2010	3.2	0.14	Global
GBD 2016[3]	2005	3.7	0.19	Global
	2016	4.1	0.23	Global
Malley et al. 2017[4]	2010		1.04-1.23	Global
Fann et al. 2012[78]	2005	0.05-0.2	0.005	USA
GBD 2016[3]	2016	1.1	0.07	China
	2016	1.0	0.09	India
	2016	0.15	0.011	Western Europe
	2016	0.09	0.012	USA
EEA 2017[79]	2014	0.43	0.09	Europe
Malley et al. 2017[4]	2010		0.27-0.33	China
			0.40-0.45	India
			0.03-0.05	N. America

The geographic distribution of these deaths is presumed to follow closely the amount of $PM_{2.5}$, which is likely to penetrate more deeply into the respiratory system than larger particles. Changes in concentrations of $PM_{2.5}$ in the atmosphere since 1960 have been estimated and used to compute corresponding changes in aerosol-associated mortality (**Fig. 10**).[58]

Insert Fig 10 after this point

Fig. 10 Changes in deaths globally due to particulates for the period 1960 to 2009 (from ref.[58]).

4.1.2 Overview of recent human health studies

Reports of associations between poor air quality and various diseases in humans continue to accumulate in the literature. A large proportion of the global population is potentially exposed to excessive concentrations of atmospheric particulates.[80] For $PM_{2.5}$, it has been reported that, in 2016, 95% of the world's population was living in areas where

ambient concentrations of $PM_{2.5}$ were greater than the World Health Organization annual average guideline (10 µg m^{-3}).[80] Global population-weighted concentrations of $PM_{2.5}$ were greater in 2016 (51.1 µg m^{-3}) than 2010 (43.2 µg m^{-3}), a change driven by increases in South-Asian countries. As was pointed out earlier,[65] outdoor air pollution is recognized as a Group-1 human carcinogen by the International Agency for Research on Cancer.[81] Many of the recent studies on the impact of air pollutants report an increased risk of many adverse (non-cancer) health effects with increased levels of air pollution.

Almost all epidemiological studies only report associations or linkages between a disease and exposures to one or more chemicals. It is important to understand that association or linkage is not proof that the exposure caused the disease (causality). As shown in the Hill guidelines of causality,[82] other information which supports the association must be considered.

Below, we provide an overview of recent studies on the adverse effects in humans that have been associated with air pollution. This is not a meta-analysis or systematic review, but is provided to highlight the importance of air pollution to health, especially in countries and locations where air pollutants are found in large concentrations.[83] Because this section is based in the epidemiological literature, terms specific to this branch of science are explained in footnotes at first mention (as well as in the abbreviations). Many of these studies assessed exposures to air pollutants that included significant concentrations of gases and particulates of varying chemical composition. Therefore, it may not be possible to defensibly assign causality to a pollutant or sub-component of a pollutant, i.e., PM.

Effects of air pollutants on mortality. Several studies have assessed increased mortality arising from exposure to individual air pollutants in humans. A global estimate of deaths in persons $\geq$ 30 years of age attributable to O_3 was based on the results of exposure and respiratory mortality from the American Cancer Society Cancer Prevention Study II.[4] Mathematical models were used to estimate 6-month average exposures and risk thresholds corresponding to the minimum or 5th centile of O_3 exposures for 2010. The authors estimated that, globally, 1.04–1.23 million respiratory deaths in adults in 2010 were attributable to exposure to O_3. The study also indicated that increases in estimated attributable mortality were larger in northern India, southeast China, and Pakistan than in Europe, eastern USA, and northeast China. These results were consistent with an earlier regional study in the USA that reported a relative risk of death from respiratory causes of 1.040 (95% CI[a] 1.010, 1.067) is associated with an increment in concentration of O_3 of 20 µg m^{-3}.[84] A prospective study[b] of cancers other than lung cancers was carried out as part of the above Cancer Prevention Study II,[85] reported increased exposure to $PM_{2.5}$ (per 4.4 µg m^{-3}) was significantly positively associated with deaths from cancers of the kidney (HR = 1.14; 95% CI of 1.03, 1.27) and bladder (HR = 1.13; 95% CI of 1.03, 1.23). Increased exposures to NO_2 (increment of 12 µg m^{-3}) was positively associated with mortality from colorectal cancer (HR of 1.06; 95% CI of 1.02, 1.10). No association between any type of cancer was associated with O_3.

A study on the effects of increased concentrations of O_3 and total daily mortality was conducted in seven cities in Jiangsu Province, China (2013–2014).[86] Daily total mortality was 0.55% higher (Bayesian 95% credibility interval = 0.34-0.76) for each 10 µg m^{-3} increment in

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[a] A confidence interval is an interval that will contain a population parameter a specified proportion of the time, 95% in this case.

[b] A prospective study is an epidemiological study which follows disease incidence in a cohort over time.

the 8-h average concentration of O_3, with a 2-day lag. Similar associations with exposure to ambient concentrations of O_3 were reported in Italy[87] and Iran.[88] A nationwide study in 272 Chinese cities between 2013 and 2015 showed that a 0.24% increase in daily mortality from all nonaccidental causes (95% Prediction Interval (PI^c) = 0.13%–0.35%) was associated with an increment in 8-h maximum concentration of O_3 of 10 μg m^{-3}, approximately half that reported above.[89] However, no association between increased concentrations of O_3 and daily mortality from respiratory diseases was reported.

Exposure to particulates have also been associated with premature mortality. Exposures were modelled from measured values using geocoding, a national-level hybrid land-use regression and Bayesian maximum entropy interpolation. Based on 623,048 participants followed from 1982–2004 and 43,320 non-lung cancer deaths, higher exposures to $PM_{2.5}$ (per 4.4 μg m^{-3}) were significantly associated with increased risk of death from cancers of the kidney (Hazard Ratio^d [HR] = 1.14; 95% CI of 1.03, 1.27) and bladder (HR = 1.13; 95% CI of 1.03, 1.23). Higher exposures to NO_2 (increment of 12 μg m^{-3}) were positively associated with mortality from colorectal cancer (HR of 1.06; 95% CI of 1.02, 1.10). No association between any type of cancer was associated with greater exposure to O_3. In a prospective cohort study of 189,793 older men (age > 40 in 1990 and 1991) in 45 locations in China, non-accidental mortality from several causes was associated with concentrations of $PM_{2.5}$.[90] Annual average levels of $PM_{2.5}$ were estimated from measurements and models and, for the period of investigation (2000–2005), ranged from 4.2 to 83.8 μg m^{-3}. HRs and the (95% CIs) for a 10-μg m^{-3} increment in $PM_{2.5}$ were 1.09 (1.08–1.09) for non-accidental causes; 1.09 (1.08–1.10) for cardiovascular disease; 1.12 (1.10–1.13) for chronic obstructive pulmonary disease (COPD); and 1.12 (1.07–1.14) for lung cancer. The analytical model included adjustments for smoking and other lifestyle variables. These data provide consistent evidence of increases in mortality associated with exposures to air pollutants.

Effects of air pollution on the respiratory system. The respiratory system is expected to be the primary target of air pollutants. This is the primary site of uptake and where concentrations and effects would be greatest. This has been shown in several studies[65] with the suggested mechanisms involving inflammation, vasoconstriction, and coagulation of blood.[91] Some studies reported on mortality only (included above), while others also reported on ill health (morbidity), which is discussed below.

A meta-analysis of the relationship between ambient outdoor air pollution and exacerbation of chronic obstructive pulmonary disease (COPD) was based on 37 papers.[92] In this analysis of a total of 1,115,000 COPD-related acute events and 130,000 deaths, the authors indicated that there was a significant positive association between concentrations of pollutants ($PM_{2.5}$, NO_2, and SO_2) and COPD-related morbidity and mortality. Unfortunately, exposure to O_3 was not included in the study. A study of lung function in response to daily measures of air pollutants was carried out in Belgium.[93] Several repeated measures of lung function were obtained from 2449 adults over a 4-year period between 2011 and 2015. No significant association was observed for O_3, but several measures of lung function decreased in response to elevated concentrations of PM_{10} and NO_2 in the days prior to testing.

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^{c} Prediction Interval [PI] is a range of values in which future observations will fall, with a certain probability, given what has already been observed.

^{d} HR, the hazard ratio is the ratio between the responses observed in the exposed and control groups.

Effects of air pollution on human health are a global issue and linkages like the above have been reported from several countries.[94-98]. Mortality from acute respiratory distress syndrome (ARDS) was associated with exposure to O_3 and particulate matter.[99] After controlling for other (unstated) variables in the model, treatment in a hospital located in an area with high O_3 was associated with an increased odds ratio (OR[e]) of mortality of 1.11 (95% CI; 1.08–1.15, p < 0.01) for in-hospital mortality. Adjusting for all factors, for each 20-µg m^{-3} increment in the concentration of O_3, the OR for mortality was 1.07 (95% CI; 1.06–1.08, p < 0.01). Exposure to O_3 is associated with changes in lung function in asthmatics as well as in the general population.[100, 101] A 20-µg m^{-3} increment in the concentration of O_3 was associated with increased incidence of cough and stuffy nose (OR; 1.23, 95% CI: 1.00, 1.51), as well as absenteeism in school children in Greece.[101] In another study on childhood lower respiratory diseases in China,[102] an interquartile range increment in concentrations of PM_{10} (72 µg m^{-3}), NO_2 (26 µg m^{-3}), and SO_2 (16 µg m^{-3}) significantly increased the daily childhood lower respiratory diseases with 6-days cumulative effect (difference of estimates was 2.8%, 95% CI: 0.6–5.0%; 4.1%, 1.2–7.0%; 5.6%, 2.6–8.6%, respectively). Childhood lower respiratory diseases were not significantly associated with interquartile increments in concentration of $PM_{2.5}$ (42 µg m^{-3}), O_3 (76 µg m^{-3}), and CO (410 µg m^{-3}).

Effects on the cardiovascular system. The cardiovascular system is probably the second most sensitive system to air pollutants in humans. Many studies have reported associations between estimated exposure to air pollutants and cardiovascular mortality and morbidity in adults and children.

As part of the Black Women's Health Study in the USA, incidence of hypertension was investigated in participants representing 348,154 person-years from 1995 to 2011 with 9,570 cases identified.[103] Exposures were estimated using a spatiotemporal model from the US EPA and were based on measured values from ground-level monitoring sites and satellite observations. Several covariates, such as self-reported weight, smoking and alcohol history, and hours/week of vigorous exercise were considered. Daily 8-hour maximum concentrations of O_3 were averaged for the years 2007–2008 to approximate the long-term average exposure. Long-term exposure to greater concentrations of O_3 (the highest quintile[f] of exposure concentrations, 82–112 µg m^{-3}) was associated with increased incidence of hypertension (HR = 1.09, 95% CI; 1.00–1.18) but the opposite was reported for NO_2 (48–73 µg m^{-3}) where higher NO_2 concentrations were related to lower incidence (HR = 0.92, 95% CI; 0.86–0.98). In a 272-city study in China for data from 2013–2015,[89] a positive association between 10 µg m^{-3} increments in concentration of O_3 and higher daily mortality was reported from cardiovascular diseases (0.27%, 95%, PI = 0.10–0.44%), including hypertension (0.60%. 95% PI = 0.08–1.11%), coronary diseases (0.24%, 95%, PI = 0.02–0.46%), and stroke (0.29%, 95%PI = 0.07–0.50%). While the proportional increases were small, the effects in large populations were significant.

Particulates have also been associated with effects on the cardiovascular system. A large study on Medicare patients was conducted in the USA.[104] The study included data on 60,925,443 individuals with 460,310,521 person-years of follow-up from 2000 through 2012. Exposures were based on postal code, and 10-µg m^{-3} increments in annually averaged

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[e] Odds ratio [OR] Odds that an outcome will occur given a particular exposure, compared to the odds of the outcome occurring in the absence of that exposure.

[f] Quintile, any of the four values that divide the items of a frequency distribution into five classes with each containing one fifth of the total population. The upper quintile of a data-set includes the values falling between 80 and 100% in cumulative frequency distribution.

concentration of $PM_{2.5}$ were associated with an increase in all-cause mortality of 7.3% (95% CI, 7.1–7.5). A statistically significant increase in mortality was also observed for a 20-µg m^{-3} increment in O_3 concentration. Deaths among men, blacks, and people with Medicaid eligibility were more than that in the rest of the population. The authors point out that there was evidence of adverse effects related to exposures at O_3 concentrations below the current air quality criterion in the USA (140 µg m^{-3}).

Several large studies have shown linkages between heart attack and stroke, and ambient concentrations of air pollutants. A study on relative risk of hospitalization for acute myocardial infarction in Wallonia (Belgium) between 2008 and 2011 showed a stronger association between heart attack and incremental concentrations of NO_2 (interquartile range[g] [IQR] = 14–26 µg m^{-3}, relative risk[h] [RR] = 1.029; 95%CI, 1.009–1.049, P = 0.005) than PM_{10} (IQR = 14–30 µg m^{-3}, RR = 1.012; 95%CI, 1.001–1.023, P = 0.027).[105] There was no significant association between risk of heart attack and incremental concentration of O_3 (IQR = 33–57 µg m^{-3}). A study in China with 13,535 patients with acute ischemic stroke hospitalized to 12 participating centers showed an association between cardioembolic stroke and PM_{10} and concentrations of SO_2.[106] Odds ratios were greatest for SO_2 (1.56, 95% CI 1.13–2.16). A multicity (14 large cities) case-crossover study of the association between ambient air pollution and hospitalization for ischemic and haemorrhagic stroke between 2014 and 2015 was carried in China.[107] Based on 200,958 cases of ischemic stroke, an interquartile increment in the six-day average concentration of PM_{10} (86 µg m^{-3}), SO_2 (38 µg m^{-3}), NO_2 (26 µg m^{-3}), CO (810 µg m^{-3}), and O_3 (67 µg m^{-3}) was associated with a significant increase in admissions only for SO_2 (1.6%, 95% CI 1.0– 2.3%) and NO_2 (2.6%, 95% CI 1.8–3.5%). For hemorrhagic stroke, the only significant association was with concentration of NO_2 on day of admission. In a second paper[108] based on the same exposure data but testing the effects of air pollutants on hospitalization for acute myocardial infarction, the only significant increases found were associated with SO_2 (2.0%, 95% CI, 1.2–2.9%) and NO_2 (2.2%, 95% CI 1.4–3.1%). Significant increases in hospitalization were not observed for O_3. In another study in China on 147,624 stroke admissions in Beijing in 2013–2014 showed a slight increase (0.82% and 0.73% resp.) in admissions with every 10 µg m^{-3} same-day increment in concentration of NO_2 and SO_2.[109] Similar results were reported in a study in Taiwan on hospital admissions for myocardial infarction.[110] This increase was reported for warm (> 23°C) as well as cool days (< 23°C) and was statistically significant. In contrast, a study of 1758 incident reports of stroke in South London (UK) between 2005 and 2012 found no evidence of a significant association between all stroke or ischaemic stroke and same day increased exposure to $PM_{2.5}$ (IQR = 10.1–18.0 µg m^{-3}), PM_{10} (IQR = 17.2–28.9 µg m^{-3}), O_3 (IQR = 23.2–49.3 µg m^{-3}), NO_2 (IQR = 33.6–53.6 µg m^{-3}), or NO_X (IQR = 50.5–92.4 µg m^{-3}).[111] The authors also reported negative (but not significant) associations between incidence of haemorrhagic stroke and an increment in concentration of 10 µg m^{-3} of PM_{10} (-14.6%, 95% CI 0.7–26.5%) and $PM_{2.5}$ (-17.0%, 95% CI, 3.3–33.3).

Kawasaki disease is an acute and multi-systemic vasculitis that occurs predominantly in infants and young children from East Asia and the Pacific. Kawasaki disease can cause myocardial infarction and sudden death in children and young adults. The seasonality of the disease has suggested an environmental cause, although there is also an increased genetic risk.[112] In a study of 695 hospital admissions for Kawasaki disease in Taiwan (2000–2010),

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[g] The interquartile range [IRQ] is the difference between 75th and 25th centiles in a cumulative frequency distribution of values.

[h] The relative risk [RR] is the risk of an outcome relative to that for an unexposed population.

an increased risk was associated with 58 µg m^{-3} increments in concentrations of O_3 on the day of hospitalization in the summer months only (OR adjusted for temperature, humidity, and wind = 1.21; 95% CI, 1.01, 1.44).[113] There were no associations with concentrations of CO, NO_2, PM_{10}, or SO_2. The authors noted that a biological mechanism for the putative effects of O_3 has not been elucidated. As has been pointed out,[114] additional studies from other regions are needed to confirm the association of Kawasaki disease with ground-level concentrations of O_3.

A meta-analysis of the association between air pollution and cardiac arrest outside a hospital showed significant associations with 10-µg m^{-3} increments of PM_{10}, $PM_{2.5}$, NO_2, and O_3.[115] The largest association was with $PM_{2.5}$ (3.9%; 95% CI, 1.2–6.6%).

In terms of morbidity, a study in Taiwanese adults between 2001 and 2014 reported a small but statistically significant association between $PM_{2.5}$ and blood pressure and risk of hypertension.[116] A total of 125,913 non-hypertensive participants was selected from a group of 361,560 adults ≥ 18 years-old for follow-up. Exposures to $PM_{2.5}$ were estimated from the participants address using a satellite-based spatio-temporal model. A 10-µg m^{-3} increment in the 2-year average concentration of $PM_{2.5}$ was associated with an increase in systolic blood pressure of 0.45 mm Hg (95% CI, 0.40–0.50). The same increment in exposure was associated with a 3% increase in risk of developing hypertension (95% CI, 1–5%). Similar results were reported in a study of 9,354 school children (5-17 years old) in China.[117] Increments in the 5-day mean interquartile concentrations of PM_{10} (50 µg m^{-3}) and O_3 (53 µg m^{-3}) were associated with elevations of 2.1 mm Hg (95% CI, 1.7–2.4) and 3.3 mm Hg (95% CI, 2.9–3.7) in systolic blood pressure, respectively. The ORs for an increment of 1 IQR ranged from 1.6 to 2.7 for PM_{10} and 1.12 to 3.33 for O_3 and were significant. In another study, an analysis of daily visits to emergency rooms in Beijing, China between Jan 1, 2014 and Dec 31, 2015 (7,088,309 visits) showed significant associations with same-day concentrations of O_3.[118] A 10-µg m^{-3} increment of 8-hour averaged O_3 was significantly associated with a 0.24% (95% CI, 0.21%–0.26%, $P < 0.01$) increase in daily visits to emergency rooms across the entire study period. When concentrations were ≤ 100 µg O_3 m^{-3}, an increment of 10 µg O_3 m^{-3} was associated with an increase in visits of 0.31% (95% CI, 0.27%–0.35%) and, when concentrations were between 100 and 160 µg O_3 m^{-3} the increase in visits was 0.43% (95% CI, 0.36%–0.50%). The authors reported that $PM_{2.5}$, PM_{10}, NO_2, SO_2, and CO were not significant confounders. Admissions were significantly greater on warm (daily mean temperature > 18.5°C) than cool days (< 18.5°C), most likely because of correlations between temperature and concentration of O_3.

Effects on the reproductive system. Other studies have shown linkages between exposure to air pollutants and reproductive function. A study on *in-vitro* fertilization in France showed a reduction in the success of stimulated production of eggs in association with prior acute exposures to NO_2 or PM_{10}.[119] Interestingly, exposure to greater concentrations of O_3 for between 1 day to 2 months before folliculogenesis resulted in enhancement of egg production and successful fertilization. A study on 13,775 pregnancies in North-East Scotland showed that greater annual mean exposure increments of PM_{10} (10 µg m^{-3}), $PM_{2.5}$ (5 µg m^{-3}), and NO_2 (10 µg m^{-3}) were associated with smaller newborns.[120] Exposure to O_3 was not included in this study. A study on birth-weight of 8,948 newborns (2012-2013) in Sao Paulo (Brazil) showed that exposure to elevated concentrations of O_3 in the 30- and 90-day prenatal window was associated with low birth-weight in both sexes (OR = 1.39, 95% CI, 1.05–1.85 and 1.49, CI 1.10–2.00, respectively).[121] Exposure to increased concentrations of PM_{10} was associated with a protective effect but only in female newborns. For the 30-day window, concentrations of PM_{10} ranged from 21.7–85.5 µg m^{-3} and for O_3 from 37.9-71 µg m^{-3}. An unrelated study of vascularization of the placenta in the first trimester in low-risk pregnant mothers (229

participants) in Sao Paulo showed that there was a significant decrease in this measure with increasing concentration of NO_2 (40.5 μg m^{-3}, ± Standard Deviation [SD]^i 7.7 μg m^{-3}).[122]. There was no association with concentrations of O_3 (8.2 μg m^{-3}, ± SD 1.15 μg m^{-3}). In a study of hypertensive disorders of pregnancy in Florida (USA), the odds of this condition were reported to be increased by each 10 μg m^{-3} increment in weekly averaged concentration of O_3.[123]. The study was based on 655,529 pregnancies with conception dates between 2005 and 2007. The ORs were all < 2 and non-significant, except for gestation time > 24 weeks. The authors noted that exposure early in pregnancy was associated with higher ORs for hypertensive disorders of pregnancy. A recent study of preterm births in cities across Ontario, Canada, and exposures to $PM_{2.5}$ showed an increased risk associated with an increased exposure to $PM_{2.5}$ [IQR = 2.6 μg m^{-3}; OR = 1.08 (95% CI = 1.01, 1.15)] in the first trimester.[124] This study also investigated the oxidative potential of the $PM_{2.5}$ collected in the air samplers used to characterize exposures. Oxidative potential was determined from *in vitro* incubation of particles with glutathione or ascorbic acid. Consumption of these reactants was used to estimate oxidative potential, which, across 32 cities ranged from 0.01 to 0.36 % depletion of glutathione per μg of $PM_{2.5}$ and 0.03 to 0.4 % depletion of ascorbic acid per μg of $PM_{2.5}$. When oxidative potential was included in the exposure estimate for $PM_{2.5}$, the above OR increased to 1.31 (CI = 1.07, 1.61) and 1.12 (CI = 0.95, 1.32) glutathione and ascorbic acid, respectively. These results point out the need to characterize the chemical properties of atmospheric particulates when conducting epidemiological studies, as mentioned above when considering PM composition.

A study in the USA examined risk of stillbirth in a retrospective cohort of 223,375 single-baby deliveries from 12 clinical sites.[125] Based on an interquartile increment in exposures, O_3 was significantly associated with stillbirth (RR = 1.13–1.22) on days 2, 3, and 5–7 prior to delivery (range of IQRs = 35.4–35.6 μg m^{-3}). Relative risk for the whole pregnancy was 1.39 (95% CI, 1.05–1.84). No significant increases or decreases in risk were observed for NO_2, PM_{10}, and $PM_{2.5}$ but a significant decrease in RR (0.80, 95% CI, 0.66–0.96) was observed in the first trimester for SO_2 (IQR = 7.7 μg m^{-3}). Based on these data, the authors suggested that 8000 stillbirths per year in the US might be attributable to increased exposure to O_3.

A study in the USA assessed the effect of air pollution on erectile dysfunction (ED) in men aged 57 to 85.[126] The 412 participants were from the National Social Life, Health, and Aging Project and daily concentrations of air pollutants were obtained from models ($PM_{2.5}$) and measurements (NO_2, all year [IQR for the 7 years ranged from 13.6–14.9 μg m^{-3}] and O_3, in the summer only [IQR for the 7 years ranged from 13.6–16.4 μg m^{-3}]). Based on an increase of concentrations averaged over 1–7 years, ORs for ED were >1 but < 2 for all pollutants. None of the ORs were statistically significant. No ED-specific mechanism was investigated or proposed.

Other health effects. The incidence of Parkinson's disease has been associated with air pollutants in a critical review.[127] The strongest linkages were with air pollutants related to traffic and urbanization (NO_2, NO_x, and CO). However, inconsistent (positive and negative) associations between Parkinson's and O_3 were reported in this meta-analysis.

An increased incidence of dementia associated with air pollutants has been reported. A study population of 2.1 million individuals from Ontario, Canada was identified in 2001 and followed to 2013.[128] Historical long-term exposures to $PM_{2.5}$, NO_2, and O_3 were estimated with the aid of satellite observations, a land-use regression model, and an optimal

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^{i} Standard Deviation [SD], a measure that quantifies the amount of variation or dispersion of a set of data values.

interpolation method. The authors reported a positive association between dementia and $PM_{2.5}$ (HR of 1.04; 95% CI = 1.03–1.05) for every 4.8 µg m^{-3} (IQR) increase in concentration. For NO_2, the HR was 1.10 (95% CI = 1.08–1.12; IQR = 27 µg m^{-3}) but, for O_3 (IQR = 12.6 µg m^{-3}), there was no significant association. The authors estimated that 6% of the cases of dementia were attributable to poor air quality ($PM_{2.5}$ and NO_2). A similar observation for the association of an increased incidence of dementia with exposure to NO_2 was reported in a smaller study from Sweden (1721 participants).[129] Changes in the human brain associated with PM exposure have been reported.[130] Magnetic resonance imaging (MRI) of the brain in 1753 individuals in Maryland, Minnesota, North Carolina and Mississippi were compared as a function of exposure to PM_{10} and $PM_{2.5}$ in three time periods (1990–1998, 1999–2007, and 1990–2007). Higher past exposures to PM were associated with smaller deep-gray brain volumes[j].

Impaired physical function has been associated with air pollution. In a study of 1,762 Dutch older (75 ± SD 9 years) adults,[131] data were collected between 2005–2006, 2008–2009, and 2011–2012 on performance-based activities (walking speed, ability to rise from a chair, putting on and taking off a cardigan, and a balance-test). An association in decreased performance and NO_2, NO_x, $PM_{2.5}$, and PM_{10} was reported (IQRs of 8.9, 13.5, 1.4, and 1.5 µg m^{-3}, respectively). Significantly decreased performance was observed only for the upper exposure quartile for NO_2 (> 43 µg m^{-3}), NO_x (> 67 µg m^{-3}), and PM_{10} (> 29 µg m^{-3}). The authors noted that the decrease in score resulting from exposure to air pollutants was equivalent to an increase in age of nine months. A large-scale study (75,000 people) in China,[132] reported that exposure to air pollutants was associated with decline in cognitive ability in terms of verbal and mathematical tests in adults ranging in age from 25 to 65+ years. Air quality was measured using the air pollution index, a composite of daily concentrations of SO_2, NO_2, and PM_{10}, obtained from the city-level air quality reports for 86 major cities in 2000 and most of the cities in China in 2014. Because of the use of the composite, it was not possible to attribute a response to a specific chemical pollutant. Collectively, these studies show that exposures to air pollutants can result in non-lethal effects that significantly affect quality of life, especially in the elderly.

Increased concentrations of ambient O_3 were reported to be associated with increased risk of Type-2 diabetes.[133] This study used data from a cohort of 45,231 African American women from 56 cities across the USA. Concentrations of O_3 were estimated using modeled predictions adjusted with ground measurements from 2007–2008. For an interquartile increment in concentration of O_3 of 13 µg m^{-3}, a significant increase in incidence of 1.18 (95% CI, 1.04–1.34) of Type-2 diabetes was reported. This appeared to be specific to O_3 and was not associated with $PM_{2.5}$.

Overall, air pollutants in general have very significant and diverse effects in humans. In general, fine particulates ($PM_{2.5}$) are most important, but O_3 and NO_2 are also relevant for some outcomes. Ozone appears to not be associated with an increased frequency of dementia, an important condition in the elderly, but other air pollutants are, particularly PM, where direct effects also have been observed on the brain. Future changes in health impacts will depend on changes in UV radiation and the UV-induced changes in pollutant concentrations, as well as changes in emissions and climate.

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[j] Smaller deep-gray brain volumes are characteristic of Alzheimer's disease.

4.2 Future changes in human health due to climate

Future climate change has been estimated to produce significant changes in air quality as measured by exposure of humans to O_3,[134, 135] with actual exposure dependent on a range of factors. While this study was based on modeling changes in demographics of humans, it assumed that human behaviour will not respond to changes in climate. Further, it did not attempt to capture changes in local emissions due to changing air quality. As has been pointed out, there is considerable background variability in concentrations of ground-level O_3.[136] Thus, detecting trends in air quality data due to climate changes will probably require longer-term datasets (> 15-years) with averaging times currently used.

Climate change on its own is predicted to directly change the incidence of heat-waves with consequences for humans.[137, 138] A study based on historical data on deaths in New York City related to increased temperature predicted that a range of responses could occur.[139] Based on the response of the population to greater temperatures in the last 80 years and predictions from 33 climate-change models, the authors suggest that, for the IPCC Representative Concentration Pathway (RCP) 4.5, annual mortality would decrease from 638 heat-related deaths between 2000 and 2006 to 167 in the 2080s (June – September). This is driven by high adaptation and assumed demographic change. However, for RCP 8.5, the estimated annual number of deaths was estimated to increase to 3,331 with minimal adaptation.[139] In addition to increased mortality, decreased and increased temperatures resulting from climate change are predicted to boost demand for medical services. This has been reported for developed-[140, 141] and developing-countries.[142] In the latter case, a study in China reported that, in 2011-2014, north-south variations in effect of temperature in admissions to emergency departments were observed but admissions were greater at temperatures below the average than at temperatures above average.

Interactions between effects of air pollutants on humans and climate continue to be observed. The association between concentrations of O_3 and general mortality, cardiovascular, and respiratory mortality in Zhengzhou, China were observed to be more pronounced in the cold than in the summer months.[143] Daily mean temperature and relative humidity were 23.9°C and 57% in the warm season and 8.2°C and 52.8% in cold season, respectively. The association between atopic dermatitis in children in Korea and increases in tropospheric O_3 and other atmospheric pollutants was weaker with greater outdoor and relative humidity[144] A study on the interactions between climate change and future concentrations of tropospheric O_3 suggested differences between two RCPs.[145] For RCP 4.5, predicted increases in the concentration of O_3 for the 2050s from a combination of climate change and emission control policies was estimated to result in a small increase in annual premature deaths (50) in the USA. For RCP 8.5, predicted increases in concentrations of O_3 were estimated to result in over 2,200 additional premature deaths annually by the 2050s. A similar projection to 2025-2035 of concentrations of O_3 and temperatures in cities in the USA suggested that mortality attributable to $O_3 > 80$ µg m^{-3} would increase by 7.7% (95% CI, 1.6–14.2%).[146] Mortality from concentrations of $O_3 > 150$ µg m^{-3} was predicted to increase by 14.2% (95% CI, 1.6–28.9%). Increases in concentrations of O_3 and interactions with climate change are predicted to result in increased mortality and morbidity.

4.3 Adverse effects of poor air quality on vegetation, crop plants, and food security

The effects of air pollutants on vegetation, particularly increased concentrations of O_3, have been reviewed in previous reports[65, 138] and these effects continue to be important. While concentrations of PM_{10} and $PM_{2.5}$ appear to have few effects on plants, the effects of

increased concentrations of O_3 continue to be documented, impacting the production of crops and the yields of fibre and food for humans and domestic animals. Forests store the largest terrestrial pools of carbon and contribute to the stabilization of the global climate system. However, forests are threatened by climate change and associated air pollution, such as O_3 and NO_x.[147] There are possible negative feedbacks in this system as O_3 tends to counteract stimulation of plant-growth by elevated concentrations of CO_2.

Sensitivity to ambient and above-ambient concentrations of O_3 continues to be reported for distinct species of vegetation (**Table 2**), although some photosynthetic organisms have been shown to be insensitive to O_3. Chlorolichens, for example, possess antioxidant systems that are highly expressed because of their need to tolerate very variable water-stresses (poikilohydric).[148] These antioxidant systems were shown to protect two species of lichen (*Parmotrema perlatum* and *Xanthoria parietina*) from exposures to 500 µg O_3 m^{-3} for 5 h per day for 14 days. Lichens are important in food chains for Arctic caribou but Arctic regions are estimated to rarely experience ground-level concentrations of O_3 > 80 µg m^{-3},[149] so food chains in the Arctic are unlikely to be adversely affected by transport of O_3 to the lower troposphere.

Insert Table 2 after this point

Table 2. Reports of adverse effects observed in plants after exposure to O_3 in semi-field experiments, where plants are exposed to O_3 in partially open experimental chambers.

Species of plant and adverse effect observed	Exposure		Refer-ence
	Conc. (µg m$^{-3)}$)	Time of exposure	
Larix kaempferi and *L. gmelinii* var. *japonica* cross *L. kaempferi,* (discolouration of leaves, reduced concentration of chlorophyll, and height of plants)	132	2 years	[150]
Oryza sativa (rice, var. Nipponbare, L81, and BRRI dhan28), (reduced height and tillering, stomatal conductance, lipid peroxidation, biomass, and yields)	154	5 weeks after transplant to harvest	[151]
Triticum aestivum (wheat, var. Yannong 19, Yangmai 16, Yangmai 15, Yangfumai 2, and Jiaxing 002), (decreased photosynthesis)	104	7 h per day from tillering to harvest	[152]
Cryptomeria japonica (no effect on growth and photosynthesis, increased height increments in year-one)	66	10 h per day for 194 and 208 d over two years	[153]
Sixty species of woody plants from temperate and subtropical regions of China, (reduced biomass of woody plants, reduced photosynthesis and transpiration, reduced chlorophyll content, growth)	232	Variable	[154]
Ficus insipida tropical tree from Panama (downregulation of secondary metabolites, accelerated senescence of leaves).	60 (ambient)	Max. ambient value, duration not reported	[155]
Lactuca sativa (lettuce var. Romana and Canasta), (reductions in yield but differences between cultivars)	130 (filtered)–212 (ambient)	24 h for 22–34 days	[156]
Populus deltoids and *P. euramericana* (clones), (changes in concentrations of carbon, nitrogen, phosphorus in different "organs" of the plants)	120	10 h per day for 96 days	[157]
Plantago major and *Sonchus oleraceus* (reductions in the light-saturated net photosynthesis, stomatal conductance, and transpiration rate)	170	9 h per day for 30 days	[158]
Zea mays (maize, 18 diverse maize inbred and hybrid lines), (accelerated senescence of leaves with variation between lines)	200	8 h per day for 62–75 days in	[159]

Species of plant and adverse effect observed	Exposure		Refer-ence
	Conc. (µg m^{-3})	Time of exposure	
		three seasons, 2013–2016	
Machilus ichangensis and *Taxus chinensis* (subtropical trees from China), (reduced rate of photosynthesis, changes in biomarkers of oxidative stress)	300	8 h per day for 245 days	160

Further estimates of decreased yields in important crops due to exposure to O_3 have been reported. In a study of production of rice in China,[161] it was shown that, for each day with concentrations of $O_3 > 240$ µg m^{-3}, there was a loss of yield of $1.12\% \pm 0.83\%$. The authors point out that increased concentrations of O_3 at ground-level in China may lead to reductions in rice yields large enough to have implications for the global rice market. A study of concentrations of O_3 in the delta of the Yangtze River during the growing season for rice and wheat reports significant losses.[162] In 2015, the estimated loss of yield was 10 - 36% for wheat and 7–24% for rice, estimated by the authors to be equivalent to losses of 2.1 and 2.4 billion US$ respectively. Similar conclusions were drawn in an extensive survey of concentrations of NO_x (a precursor of O_3) in India and their potential effects on yields of wheat and rice.[163] Overall, there is compelling evidence that many species of plants can be adversely affected by O_3 and that this results in economically significant losses in yields. These points need to be considered in decision-making in response to future changes that are expected to result from changes in stratospheric O_3, tropospheric air pollutants and climate change.

Some advances have been made in terms of monitoring the effects of O_3 on plants. It had been suggested that yields of O_3-sensitive and O_3-tolerant genotypes of snap bean (*Phaseolus vulgaris* L.; S156 and R123) could be used for biomonitoring of ambient O_3. However, a study conducted in outdoor plant environmental chambers in Greece showed that yields were also sensitive to temperature in both strains but with a difference in response.[164] Similar effects of local climate were observed in studies on clover (*Trifolium subterraneum* and *T. striatum*) exposed to O_3 in Spain;[165] however, here the authors suggested that Phytotoxic Ozone Dose (POD), based on the accumulated O_3 flux into the leaves, was a more accurate indicator of damage. A review of the use of reflectance spectroscopy to monitor responses of plants to air pollutants suggested that this was a reliable technique for assessing damage in leaves from exposure to O_3 and several other air pollutants.[166] The influence of local temperature and water stress (see below) on the effects of O_3 is potentially important in relation to the interaction of climate change with exposure to O_3 in plants.

Several interactions between tropospheric O_3, biological processes, other stressors, and nutrients have been reported in the literature. Exposure of birch (*Betula platyphylla* var. *japonica*) to above-ambient concentrations of O_3 resulted in decreased feeding by leaf-beetles (*Agelastica coerulea*),[167] but this response was not observed in another study on the same tree for the same beetle.[168] The reason for this inconsistency is not clear. Plants have been reported to respond to exposure to O_3 by increasing the production and release of antioxidants.[169] These defense compounds include several volatile chemicals, such as geranyl acetate, α-cadiene, trans-farnesol, cis-β-farnesene, which might indirectly contribute to air pollution. The concentration of O_3 was high (200 µg m^{-3}) so the relevance of this to lesser exposures is unclear. When stressed, many plants emit BVOCs, which can be used by other nearby plants to "gain information" that is used to adjust their own defenses. Using cabbage (*Brassica oleracea* var. *capitata*) as a test species, Giron-Calva et al.[170] showed that exposures to elevated concentrations of O_3 (100–120 µg m^{-3} compared to a background of 40

µg m^{-3}) impaired the plant-to-plant signaling, thus potentially exacerbating damage by herbivores.

Drought can interact with plant responses to exposure to O$_3$. In a study on the Mediterranean oak, *Quercus ilex*, damage from acute (5-h) exposure of 400 µg m^{-3} O$_3$ differed between well-watered and water-deprived plants.[171] The normal response to O$_3$, such as release of phytohormones and signaling molecules, ethylene, abscisic acid, salicylic acid, and jasmonic acid, were observed in watered plants but not in water-deprived plants. Thus, interactions between tropospheric O$_3$ and drought are likely to decrease protective responses in plants and make them more susceptible to damage from O$_3$ alone. Another study on the same species of oak showed that exposure to salt (NaCl) via soil decreased protective responses to a 5-h exposure to O$_3$ at 160 µg m^{-3}.[172] Others have shown opposite effects where other stressors mitigated the effects of O$_3$ on plants. Water stress in an O$_3$-sensitive hybrid poplar clone '546' [*Populus deltoides* cv. '55/56' crossed with *P. deltoides* cv. 'Imperial'] was protective against exposures to 80 µg m^{-3} O$_3$.[173] Exposures in this study were for 9 hours per day and lasted for 96 days during the summer months. In a study on the Chinese ornamental tree, *Lonicera maackii,* exposures to 160 µg O$_3$ m^{-3} and water restriction increased the production of bioindicators of stress but decreased visible injury on leaves.[174] In soybeans (*Glycine max*), exposures to 80 µg O$_3$ m^{-3} for 50 days caused a reduction in conductance of water in whole plants,[175] suggesting adverse interactions between exposure to O$_3$ and water-stress.

Reduced availability of zinc, an essential nutrient for plants was protective for damage by O$_3$ in seedlings of durum wheat (*Triticum turgidum* subsp. *durum*).[176] Exposures were acute (4-h) at a large concentration of 300 µg m^{-3}, conditions that are not common. In another study, exposure of bamboo (*Phyllostachys edulis* and *Oligostachyum lubricum*) to 100 µg O$_3$ m^{-3} for 10 h per day for 112 days reduced uptake and distribution of mineral nutrients (nutrient (Ca, Mg, and Fe) in the plants.[177] No interactions, positive or negative, were observed in saplings of silver birch (*Betula pendula*) to 24-h mean concentrations of O$_3$ ranging from 72 to 136 µg m^{-3} (for 165 days in the first year and 122 days in the 2nd year) and four rates of nitrogen fertilizer (10, 30, 50, or 70 kg N ha^{-1} y^1).[178] Similar lack of interactions were reported in another study on inhibition of isoprenoid-biomarkers of exposure to 80 µg O$_3$ m^{-3} in Cathay poplar (*Populus cathayana*) treated with nitrogen fertilizer at 50 or 100 kg N ha^{-1} y^1.[179] In soybean (*Glycine max*) interactive effects were found from exposure to enhanced UV radiation (5% increase in UV-B and UV-A radiation over ambient) and O$_3$.[180] Plants were exposed to O$_3$ at a mean concentration of 90 µg m^{-3} for 8 h d^{-1} for 53 days. Both the number of seeds and mass of seeds per plant were significantly reduced for the combination treatment compared to the UV radiation or O$_3$ alone; however, the design of the study did not allow differentiation of additivity and synergism. Studies of the interactive effects of O$_3$ and other stressors are relatively few and reported results are not always consistent. A better understanding of these mechanisms would be useful in developing regulatory policies for better protection of crops from the effects of O$_3$ and changes in climate.

Tolerance to tropospheric O$_3$ in plants has been identified as a pathway to reducing the effects of air pollutants on the economic and yield benefits of crops important to humans.[181] In a number of cases, quantitative trait loci for tolerance to O$_3$ have been identified in model and crop plants; however, as has been pointed out *"there is considerable research to be done before O$_3$-tolerant germplasm is available to growers for most crops"*.[181] Development of these traits has been *"hampered by the lack of translation of laboratory experiments to the field, and the lack of correlation between visual leaf-level O$_3$ damage and yield loss to O$_3$ stress"*.[181]

In a meta-analysis of the interactions between arbuscular mycorrhiza (beneficial fungi that colonize roots of plants and facilitate uptake of nutrients, such as phosphorus) and O_3, it was reported that colonization by arbuscular mycorrhiza offered protection against O_3.[182] Regardless of species of fungi, colonization by arbuscular mycorrhiza had little effect at exposures up to 160 µg m^{-3}, but did confer protection at greater concentrations, even though rates of colonization were reduced by longer time of exposure to O_3. This observation suggests that arbuscular mycorrhiza could be used as a tool to protect crops against adverse effects of O_3.

With the increasing dependence on genetically engineered (GE) crop plants, care should be taken to avoid compromising resistance to pollutants such as O_3 when other traits are inserted into the genome. A study on the effects of elevated O_3 on conventional and GE-rice showed a greater loss in yield from the variety engineered to express the insecticidal gene for *Bacillis thuringiensis endotoxin*, Bt Shanyou63 (Bt-SY63) compared to its non-GE counterpart Shanyou63 (SY63₃).[183] Plants were exposed for 91 days, but exposure was variable and time-weighted mean concentrations or cumulative load above AOT40 were not provided. A graphical display showed concentrations in the treatment enclosures ranging from 0 to 250 µg m^{-3} and AOT40 values as large as 1500 µg m^{-3} h.

5 Emissions to the atmosphere relevant to the Montreal Protocol

Most emissions of relevance to air quality are either not manageable (e.g., emissions from vegetation) or are managed by local authorities (e.g., tailpipe emission standards for internal combustion engines) rather than by international agreements like the Montreal Protocol.

The substances that replaced those initially regulated by the Montreal Protocol may contribute to poorer air quality. One of the advantages of chlorofluorocarbons (CFCs) was that they were inert in the lower atmosphere and had no direct impact on air quality. Their replacements have been specifically chosen to be less stable, and hence may be important for air quality. As these compounds are directly relevant to the implementation of the Montreal Protocol, their impacts on air and environmental quality need to be considered. Focusing on refrigeration, these replacements include ammonia, hydrocarbons, hydrofluorocarbons (HFCs) and hydrofluoroolefins (HFOs). Carbon dioxide is not discussed as a refrigerant, as it is unlikely that any release from uses of relevance to the Montreal Protocol will have direct adverse effects on human health, the environment, or food-security. Because of the way they are used, refrigerants regulated under the Montreal Protocol will eventually be recovered or released to the atmosphere where they will either degrade or cause adverse effects on stratospheric O_3. While it has been suggested that HFOs might become relevant pollutants in groundwater,[184] this is inconsistent with their properties. Given the physical and chemical properties of the fluorinated refrigerants (low solubility in water, large air-water partition coefficients, and short half-lives in the troposphere), they will not partition into surface- or ground-waters in biologically relevant amounts.[185] If the alternative refrigerant, ammonia, is released to the atmosphere it will dissolve in water droplets and could enter surface water via precipitation, where it might cause toxicity to aquatic organisms, especially fish.[186]

5.1 Trifluoroacetic acid from replacements of ODS and refrigerants with large GWPs

Trifluoroacetic acid (TFA) is a persistent substance that is formed in the atmosphere from several HCFCs, HFCs, and HFOs, the use of which falls under the purview of the

Montreal and Kyoto Protocols, and their Amendments. When released into the atmosphere, several of these gases react with the hydroxyl radical (OH) to form TFA as a terminal residue.[187] The fate of TFA in the environment (illustrated in **Fig. 11**) and its potential effects on humans and the environment have been discussed in most of the reports of the EEAP, including the last Quadrennial Assessment Report[5] and the Update Reports since then[65, 138, 188] and the reader is referred to these. In the critical review and risk assessment on the fate and effects of TFA in the environment,[187] conservative estimates of production and release of precursors were used so that the estimated additional inputs of TFA to the global environment are an upper-bound. The following is a summary of recent findings and a perspective on the relevance of the formation of TFA in the environment.

Insert Fig 11 after this point

Fig. 11 Trifluoracetic acid (TFA) formed from HFCs and HFOs in the atmosphere will rapidly partition from air to water in the atmosphere. It will combine with cations in soil and surface water and accumulate in endorheic water bodies (salt lakes) and the oceans. TFA produced from HFCs and HFOs can be reasonably well quantified but this is not true for natural sources and other chemicals used by humans (modified from ref[187], with permission).

Previous reports[65, 138, 187] have consistently noted that TFA is very recalcitrant to breakdown in the environment. This finding still holds true. A new report confirmed the catalytic defluorination of TFA in the presence of electrolysed sulfuric acid ($S_2O_8^{2-}$) and UV radiation[189] but this is unlikely to occur in nature and the energy requirements in laboratory conditions are too large to be practical for decontamination of water. Reductive defluorination of long-chain perfluorinated compounds can be catalysed by natural and anthropogenic compounds containing cobalt.[190] This study did not report defluorination of TFA but the conditions for these reactions are unlikely to be found in the environment.

Experiments with microbiota isolated from samples collected from a site with a long history of industrial contamination and activated sludge obtained from a municipal wastewater treatment plant showed that TFA was recalcitrant to biologically-mediated degradation.[191]

Other than the HCFCs, HFCs, and HFOs, there are many other sources of TFA in the environment. Because of its resistance to decomposition, it is likely to be the end-product remaining after the metabolism and environmental degradation of some 1.4 million compounds which contain the C-CF_3 moiety. While many of these compounds are likely produced in small quantities, there are virtually no data on global production and release to the environment. The use and release of refrigerants and blowing agents is documented under the Montreal and Kyoto Protocols, thus making it possible to estimate the historical, current, and future contribution of these sources to environmental loadings of TFA. The future use of precursors of TFA that are regulated under the Montreal Protocol will change in response to the Kigali Amendment but how these changes will affect future use and release of these compounds is still unknown (Velders, G, pers. comm., Jan 15, 2018).

Inputs of TFA into the environment from future conversion of all mobile (mainly automotive) air conditioners (MACs) to HFO-1234yf in China, the USA, and Europe have been estimated using a global 3-D chemical transport model,[192] extending and confirming previous modelling for North America.[193] This modelling highlights the variability in transport globally from these sources. However, the limited emission inventory, uncertainties in the chemical degradation pathway,[194] and very limited hazard assessment restricts the usefulness of these results in a global assessment. Because the HFCs that can break down in the atmosphere to produce TFA have long half-lives (1-100 years),[195] they can be distributed globally before significant degradation occurs. Thus, the formation of TFA from these HFCs is distributed globally rather than locally. The half-life of the newly introduced refrigerant,

HFO-1234yf, is much shorter, of the order of 11 days,[196] thus, breakdown will occur closer to the regions of release and larger concentrations of TFA would be expected in surface waters.[192, 193, 197] In assessing risks from local deposition of TFA, it is important to note that TFA will form neutral salts once in contact with soil and/or surface waters. For example, from the estimates of production of TFA in China, the USA, and Europe[192] and assuming no dilution, this would be equivalent to 761, 573, and 740 ng Na salt L^{-1}, respectively, all of which are still several orders of magnitude less than the chronic "no observable effect concentration" (NOEC) of 10,000,000 ng L^{-1} for TFA-Na salt from a microcosm study.[198] Although the authors point out that concentrations in precipitation in arid regions would be greater, TFA salts would run-off into the ocean where they would be diluted or accumulate in endorheic bodies of water where environmental effects of other mineral salts, which are present in much greater concentrations, outweigh those of TFA salts.[187] Overall, there is no evidence to suggest that these local depositions of TFA will result in risks to the environment, especially when eventual dilution in the oceans or endorheic bodies of water occurs. A recent review of this topic[199] reached similar conclusions with respect to TFA formed from HCFCs, HFCs, and HFOs.

Occurrence of TFA in dry-deposition from air near two landfills (Shuangkou and Baodi) in Tianjin, China[200] suggests TFA was produced from sources other than HCFCs, HFCs, and HFOs. However, it is most likely that TFA found in air, cloud- and fog-water, rainfall, and snow is the result of breakdown of precursors regulated under the Montreal and Kyoto Protocols (**Fig. 11**). Once precipitation reaches the surface and mixes with surface waters, there are many other unidentified and unregulated substances that could be precursors.[191] To regulate all these potential precursors would present a very large global challenge.

There is still no indication that exposure to current and projected concentrations of salts of TFA in surface waters present a risk to the health of humans and the environment. Since the risk assessment conducted in 2016, no novel studies indicating adverse long-lasting effects of TFA and its salts have been published. In fact, the reverse is true. Previous reports of contact toxicity from exposure to concentrated TFA (as a strong acid used in industry) have not indicated systemic toxicity.[201, 202] TFA in rain-water is millions of times more dilute than that in industrial solutions and would not cause adverse effects.

5.2   Hydrocarbons

Among the replacements for ozone depleting refrigerants are hydrocarbons. The release of hydrocarbons (such as propane and n-butane) from refrigeration systems will add to the burden of hydrocarbons in the atmosphere, and potentially increase the concentration of O_3 following atmospheric decomposition (see Fig. 3). Given that the atmospheric lifetimes of these hydrocarbons are of the order of 1–2 weeks,[203] formation of O_3 following their release (Fig. 3) will occur over national to continental distances. The assessment of the impact of emissions of hydrocarbon refrigerants on air quality in the refereed literature is limited.[204] A more comprehensive estimate has been made for three cities in the USA in a consultancy report.[205] These cities were chosen because of the problems they face with pollution by O_3. The estimates of a "worst case" increase in O_3 (as measured by the maximum increase in the 8-hour average) is around 13 μg m^{-3}, but a realistic estimate of 0.3 μg m^{-3} (for reference, the ground level 8-hour regulatory standard for O_3 is 140 μg m^{-3} in the USA). This highlights significant uncertainty and the need for some caution in the implementation of these hydrocarbon refrigerants to avoid the "worst case" situation.

5.3   Ammonia

Ammonia (NH_3) has been used for over a century as a refrigerant, particularly in larger (industrial) installations. In the USA, which has a Hazardous Substances Emergency Events Surveillance (HSEES) system for the nine states of Colorado, Iowa, Minnesota, New York, North Carolina, Oregon, Texas, Washington, and Wisconsin, NH_3 was the most commonly released chemical involved in single chemical incidents reported for the years 1999-2008, involving some 3366 incidents.[206]

Ammonia presents a significant problem for air quality. Besides refrigeration there are several large biogenic and combustion sources of NH_3 that appear not to be well quantified. Ammonia in the atmosphere reacts with several compounds to produce aerosols and hence increase concentrations of $PM_{2.5}$. There are large improvements in air quality that could be made through the limitation of emissions of NH_3, and it seems likely that this will lead to tighter emission controls from all managed sources.[207] However, use of NH_3 as a replacement refrigerant would represent only a small fraction of current emissions. Full replacement by NH_3 of current emissions of CFCs, HCFCs, and HFCs, estimated to total 170,000 tonnes yr^{-1} (for 2017; G. Velders, pers. comm., Feb. 2018), would be small compared to estimated emissions of NH_3 of 34,500,000 tonnes yr^{-1} from agriculture,[208] or 8,500,000 tonnes yr^{-1} from industrial and residential activities.[209]

6   Knowledge Gaps

UV radiation, and specifically UV-B radiation, provides much of the energy for both the generation and removal of many air pollutants. However, the multitude of pollutant sources and the importance of external factors such as temperature and wind, tend to mask the sensitivity of air quality to UV radiation. While the basic chemical mechanisms behind UV-driven generation of pollutants are well understood, quantifying the impact of stratospheric ozone changes on air quality remains a significant challenge. Changes in climate will also have a significant impact on air quality, not only through changes in temperature, wind, and rain in the lower atmosphere but also through potential changes in transport of stratospheric ozone to the troposphere. This transport remains poorly characterized by both models and observations.

A critical component in predicting changes in the global atmosphere is to constrain the likely changes in the key chemical oxidants, especially the OH radical. However, while techniques to measure OH locally are now well established, there is still no consensus on the trend in global OH concentration in the recent past, and this is a prerequisite to having confidence in future predictions.

There have been significant advances in relating air pollutants to human health recently through the study of very large populations. These techniques are most useful for acute impacts, and more chronic effects remain a challenge to quantify, given the multitude of changes people are experiencing over a lifetime. Further, some of the measures of air quality are, in themselves, limited. The use of $PM_{2.5}$ as a measure of the aerosol loading that is relevant to human health remains debatable. Beyond the question of the size of particles, the impact of the chemical composition of aerosols on human health remains unclear. While there is good reason to expect a relationship between aerosol composition and health, the framework for studying this has yet to be developed.

7   Conclusions

Photochemically-produced O_3 and secondary particulate matter near the ground are well established as major environmental threats to human health, with of the order of 3 million deaths per year ascribed to poor air quality (Table 1). Emissions of precursors for O_3 and particulate matter (particularly VOCs and NOx) will continue to be the major driver of poor air quality. UV radiation (particularly UV-B), provides most of the energy to drive the formation of O_3 and particulates in the atmosphere. Because of the Montreal Protocol, past and future changes in UV radiation at ground level are relatively small (outside Antarctica), so that the relative effects of actions under the Montreal Protocol on air quality are expected to be small. However, even a small relative change in UV radiation (e.g. 1%) could be of major importance given the already large number of people affected by poor air quality.

Ozone can be transported from the stratosphere down to the biosphere under certain meteorological conditions. While this has been long recognized, quantifying the magnitude of this transport remains difficult. While the amount of O_3 reaching the earth's surface is small, it represents a significant source for O_3. Future recovery in stratospheric O_3 concentrations as well as an increase in vertical mixing due to climate change will change the amount of O_3 reaching the Earth. These changes have the potential to increase the amount of O_3 reaching the surface from the stratosphere, thereby increasing the probability of air pollution events that adversely affect humans and the environment.

Trends in the concentrations of O_3 at the surface depend on geographic location and are affected by changes in UV radiation. The long term (20+ year) trend in surface O_3 depends heavily on the emissions of key precursors (volatile organic compounds and nitrogen oxides). Locations where emission controls have made a significant impact have shown decreases in surface O_3. However, in regions where economic growth drives emissions, there has been an increase in concentrations of O_3. The recovery of stratospheric O_3 will lead to a decrease in the UV radiation in the troposphere. This will lead to a small decrease in O_3 in large cities but an increase in O_3 outside these major source regions, potentially increasing the O_3 concentration to which the population is exposed.

UV-driven photochemical reactions convert volatile compounds into non-volatile products, leading to the formation and growth of many particles. While the overall process is generally understood, the details and the impact of changing UV radiation on the amount of PM in the atmosphere is still not well characterized.

Exposure to fine particulates ($PM_{2.5}$) and O_3 has been associated with diverse and very significant adverse effects in humans. PM have been associated with more causes of morbidity and mortality than is the case for O_3 and, in general, are an order of magnitude more important for adverse effects on human health. Ozone has only been associated with mortality from acute respiratory distress, cardiovascular disease, still-birth, and morbidity for conditions such as cough, hypertension, type-two diabetes, and Kawasaki disease.

One of the major difficulties in interpretation of studies on the effects of air pollutants on humans is the characterization of actual exposures. Models are often used to translate PM amounts at locations where measurements are made to locations where people live. These measurements are often based on average values and individual exposures will typically vary more than the estimated values. Despite this variability, many associations are still significant, suggesting that better estimates of exposure for individuals might provide more robust evidence of the effect of PM on health. A second source of uncertainty is the almost complete lack of knowledge of the composition of PM to which humans are exposed. Classification of particles based on size (PM_{10} or $PM_{2.5}$) does not consider the chemical

composition of the particles or their biological potency. Particles have been shown to contain metals, inorganic compounds, and a wide range of organic substances. Characterization of the composition of PM in atmospheric monitoring, as has been done in very few instances, and their variation over time and location, would be useful in understanding causality and could then potentially lead to better protection of human health.

There is compelling evidence that many species of plants are adversely affected by elevated concentrations of O_3, which can result in economically significant losses for nutritionally significant crops such as wheat and rice in India and China, where concentrations are generally high. The concentration of O_3 in these regions could increase due to ozone recovery and climate change (as mentioned above). On the other hand, exposure to particulates ($PM_{2.5}$ and PM_{10}) appears to have little direct effects on vegetation, including crop plants.

Interactions between plant metabolism and tropospheric O_3, other stressors, and nutrients have been reported in the literature. Increases in temperature and water-stress have been shown to enhance the adverse effects of O_3 in plants. Studies of the interactive effects of O_3 and other stressors are relatively few and reported results are not always consistent. A better understanding of the impact of exposure to O_3 and mechanisms of interaction with other stressors would be useful in formulating regulatory policies for the better protection of crops from the effects of changes in surface O_3 and climate.

The phase-out of ODSs has led to a range of other chemicals being considered, especially for refrigeration, with the potential for these to be released to the atmosphere. These substances include ammonia, hydrocarbons, hydrofluorocarbons, and carbon dioxide. Ammonia can have a significant detrimental effect on the environment in general, but except at the site of a significant release, the amounts used for refrigeration are small relative to sources such as agriculture. The release of refrigerant hydrocarbons (volatile organics) could have negative impacts on air quality, although current estimates suggest that it is small. Carbon dioxide for this use is unlikely to have an impact on air quality.

Trifluoracetic acid is a highly persistent, terminal breakdown product resulting from the atmospheric degradation of several hydrochlorofluorocarbons (HCFCs), hydrofluorocarbons (HFCs) and hydrofluoroolefins (HFOs) that are regulated under the Montreal Protocol (with the Kigali Amendment). Trifluoroacetic acid will dissolve in water in the atmosphere and be deposited on the Earth's surface via precipitation, where it will react with minerals in soil, sediment, and surface water to form salts. These salts of TFA will accumulate in the oceans and the terrestrial water bodies with no outflow (salt lakes). Based on upper-bound estimates of use and release of the HFCs and HFOs in the next 50 years, the addition of TFA salts to the existing natural background concentrations in oceans and salt lakes presents a *de minimis* risk to human and environmental health. It should be noted that TFA is a potential terminal residue of many industrial chemicals, pesticides, and pharmaceuticals, most of which are eventually released into the environment. While past and projected use of HFCs and HFOs is reasonably well understood, the relative contribution of these other industrial sources of TFA to the global load is almost completely unknown and represents a challenge to regulators.

Acknowledgements

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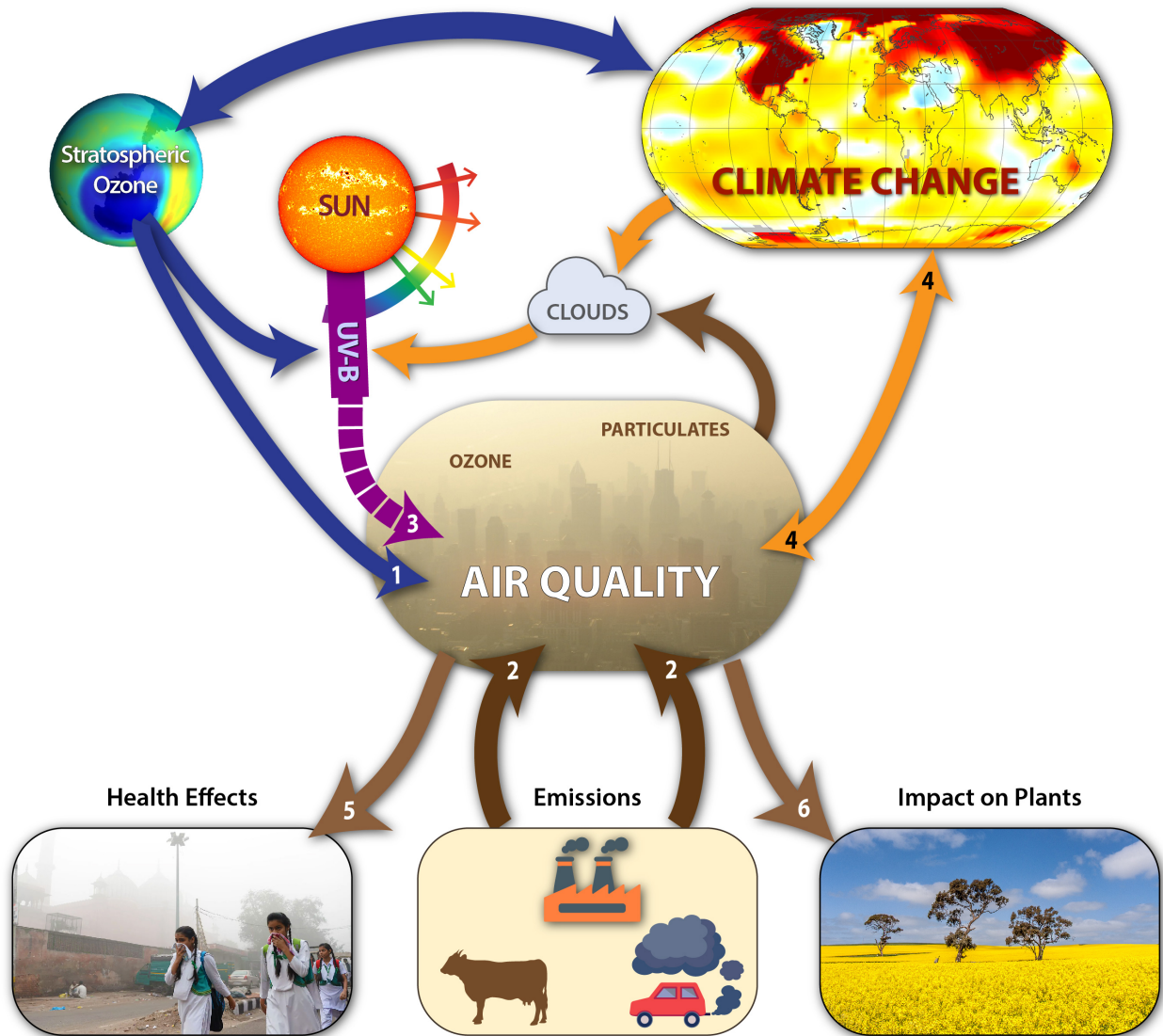
Wilson et al, list of figure captions

Fig. 1 Atmospheric composition is determined by the mixture of emissions to the atmosphere, transport within the atmosphere, and UV-B radiation. The key interactions determining the composition includes (1) transport of ozone from the stratosphere, (2) emission of a wide range of substances from the ground, (3) transformation of material through the action of UV radiation (and particularly UV-B), and (4) mixing of the pollutants in the atmosphere. The resultant O_3 and aerosols, in turn, have impacts on human health (5) and plants (6).
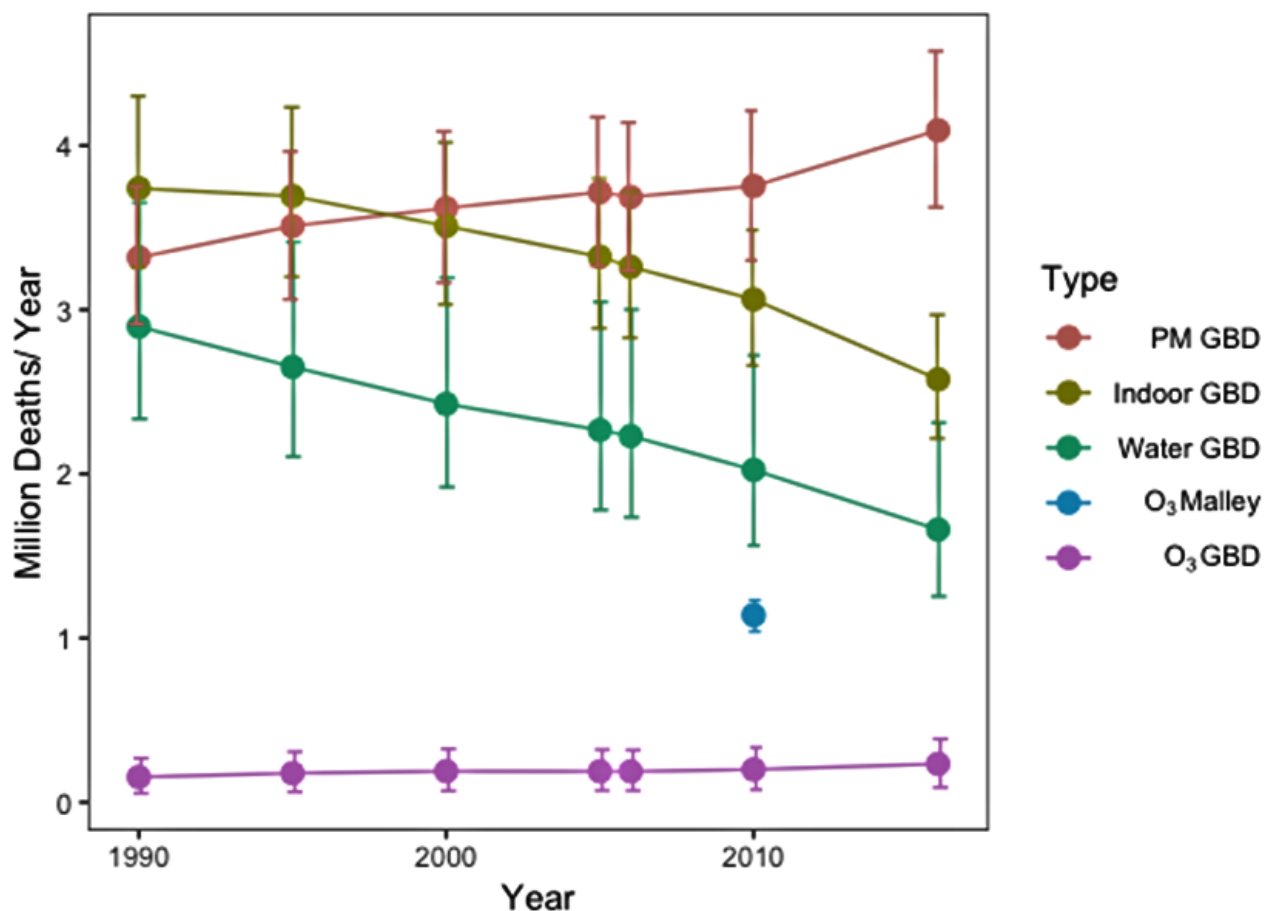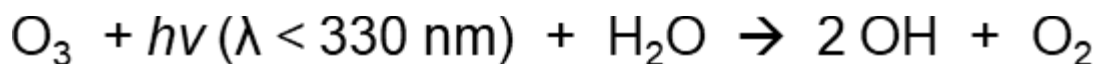
Fig. 1 Global annual deaths from environmental risk factors. Risk factors shown are primarily from the Global Burden of Disease (GBD) Study (ref.[3]). These are ambient particulate matter (PM) pollution, household air pollution from solid fuels where the major impact is indoors (Indoor), Poor water quality (Water) and ambient ozone pollution (O_3). Also shown is an alternate estimate of deaths from ambient O_3 pollution (O_3 Malley),[4] highlighting the differences in current estimates of human sensitivity to O_3.

$$O_3 + hv\,(\lambda < 330\ nm) + H_2O \rightarrow 2\,OH + O_2$$

$$RH + OH + O_2 \rightarrow H_2O + RO_2$$

$$RO_2 + NO \rightarrow RO + NO_2$$

$$NO_2 + hv\,(\lambda < 420\ nm) + O_2 \rightarrow NO + O_3$$ **Fig. 3**
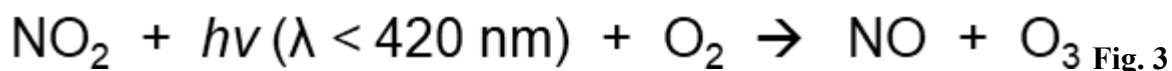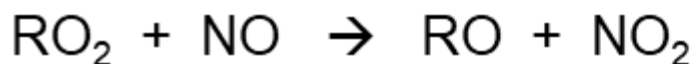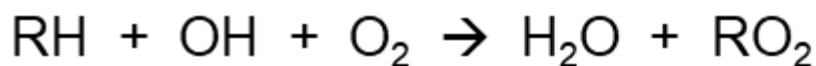
Generalized reaction scheme for the formation of ground-level ozone (O_3) by UV-driven oxidation of hydrocarbons (RH) in the presence of nitrogen oxides (NO and NO_2). The symbol

hν represents a photon of wavelength (λ) shorter than the value indicated. Further reactions of RO can lead to organic particulate matter (PM) and additional ozone.
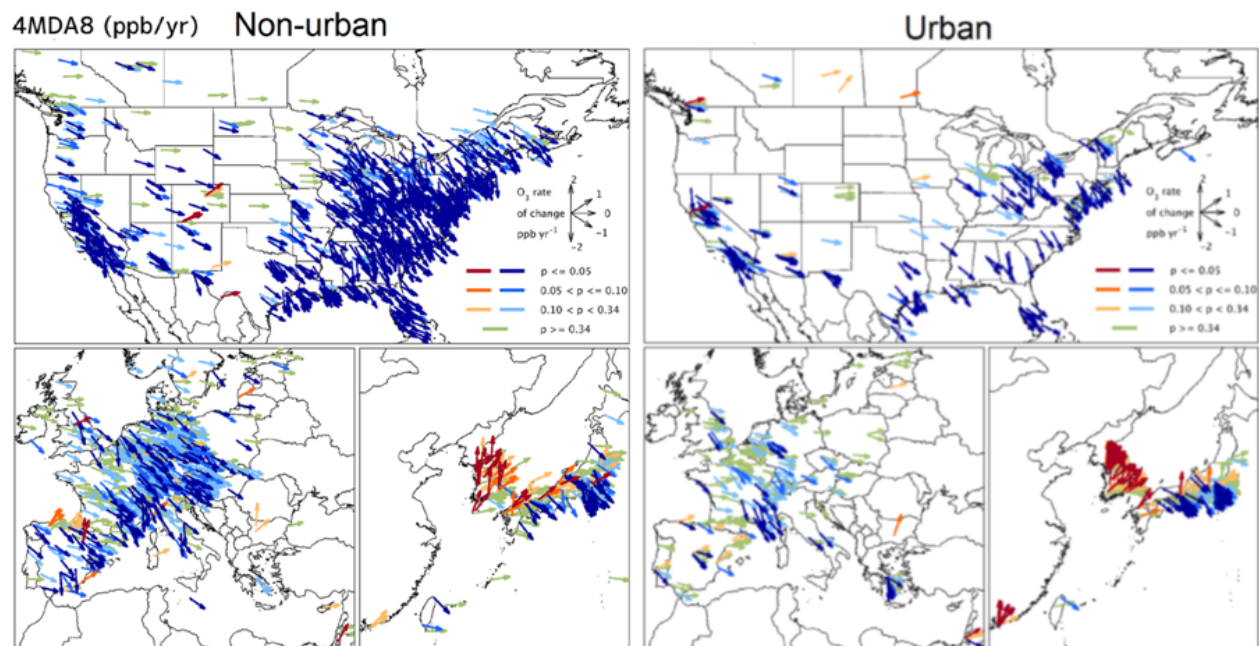

Fig. 4 Trends in concentration of O_3 at ground level for the 15-year period (2000 - 2014), measuring O_3 with the human health related metric 4MDA8. The top panels on both sides are for North America, the lower left is for Europe and the lower right side is East Asia. Blue arrows indicate a decrease in concentration of O_3 and red an increase (from ref.[24]). The dark colours (red/blue) indicate a trend that is significant at the 95% confidence level. Lighter shades indicate lower levels of significance.
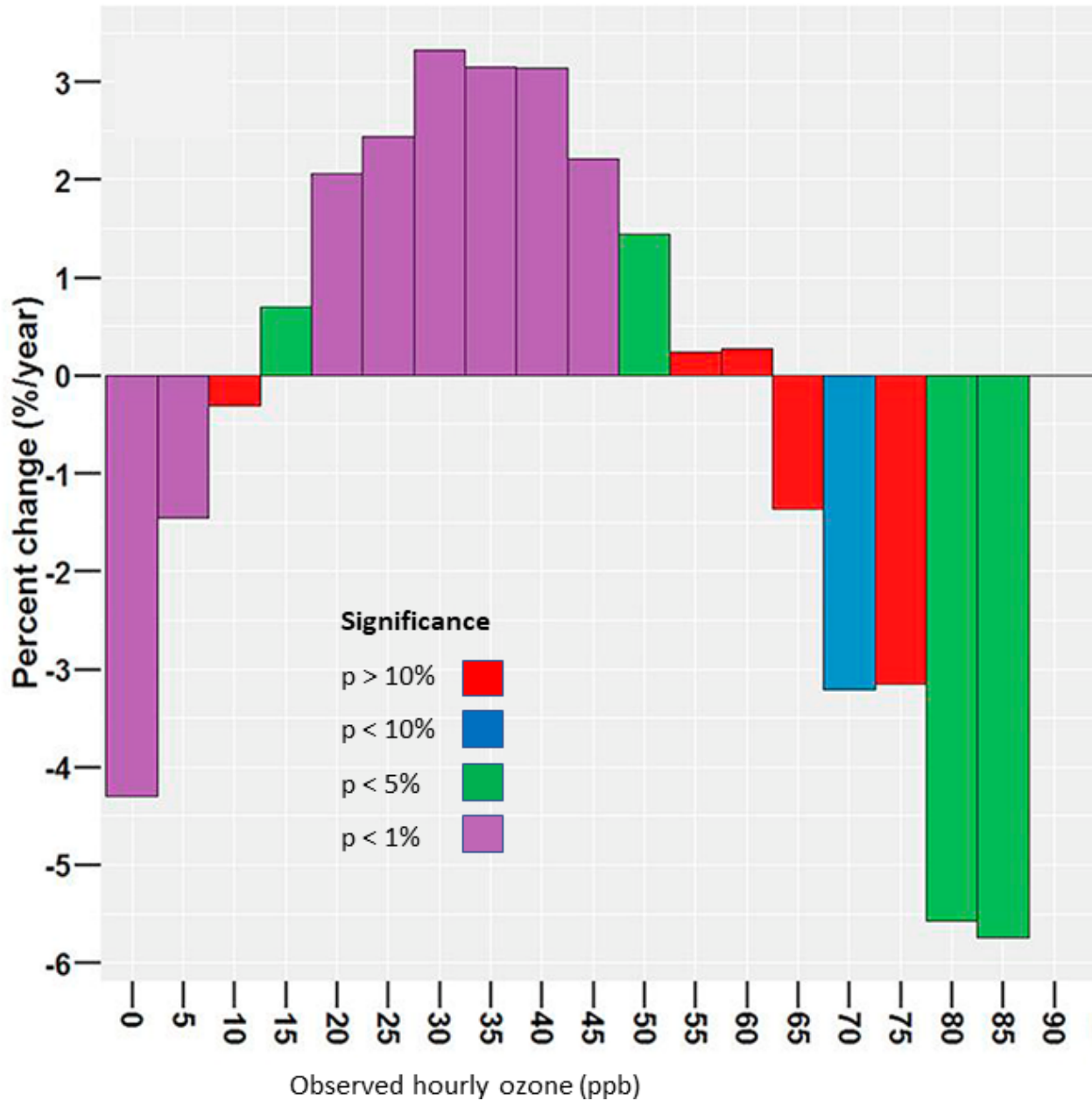
Fig. 5 Changes in observed hourly average ozone concentration at ground level for Berlin for the period 1990 to 2013. Observations have been sorted into 5 ppb wide "bins". Plotted is the trend in the number of observations within each bin. The colour indicates the probability (p) that the change is not significant (from ref.[31]).
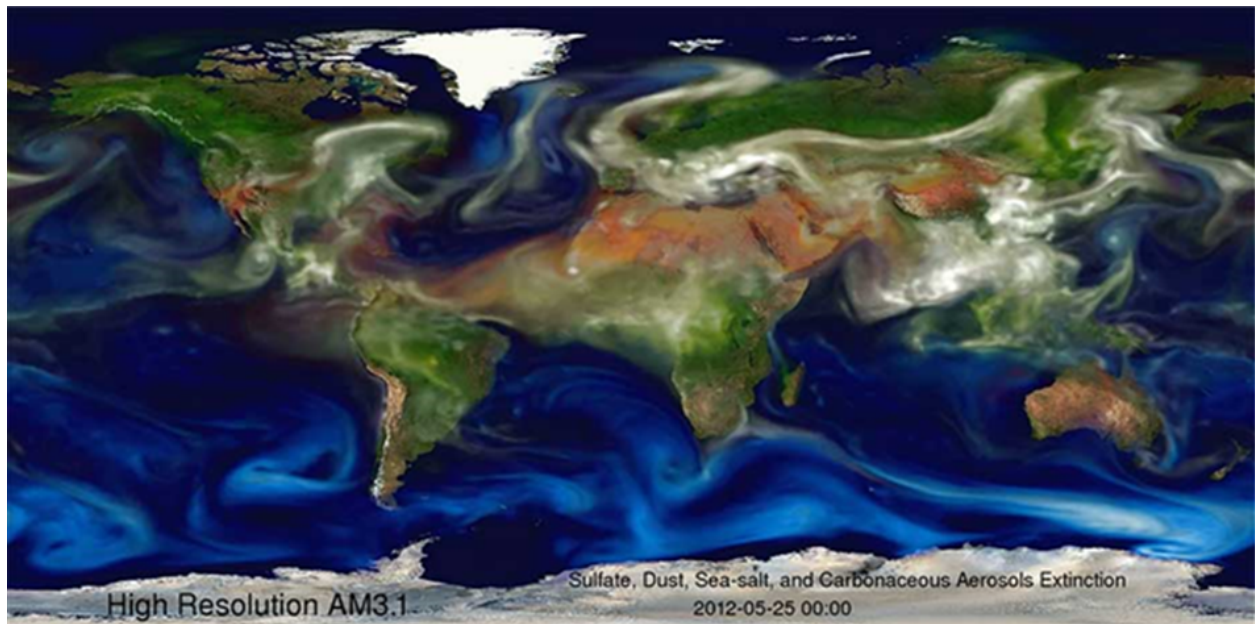
Fig. 6 Changes in ground-level O_3 because of a change from current (2000 – 2010) stratospheric ozone to that predicted for 2085, calculated for a 12 x 12 km grid (from ref.[38]).

Fig. 2 (A) Photochemical haze over Mexico City seen from approaching aircraft. **(B)** Electron microscopy image of a typical particle collected from the photochemical haze (from ref.[1])

Fig. 8 Aerosol plumes colored according to predominant particle type: dust (orange-red), sulfate (white), black carbon and organics (green), and sea salt (blue), for one day in 2012 (from Paul Ginoux, https://www.gfdl.noaa.gov/visualizations-aerosols-and-clouds/)
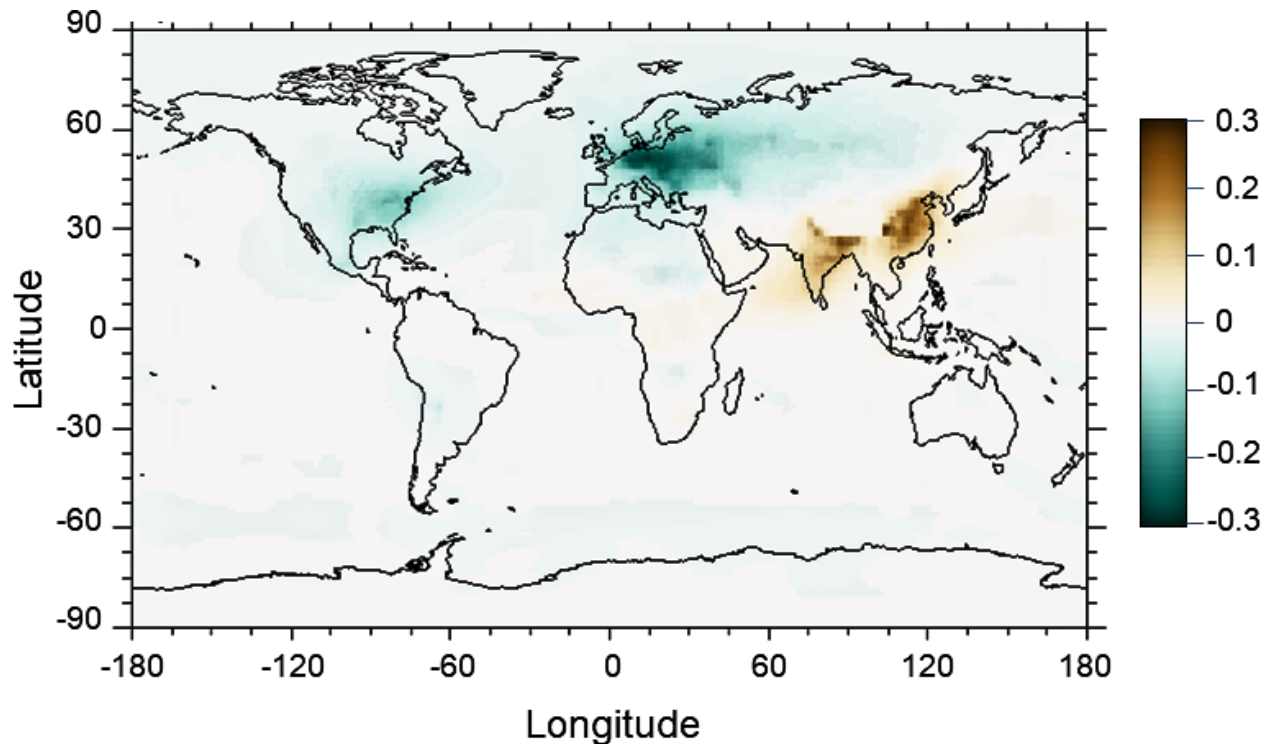

Fig. 9 Changes (1990 to 2015) in the aerosol column optical depth at 550 nm, computed as the mean of six global models (from ref.[59]).
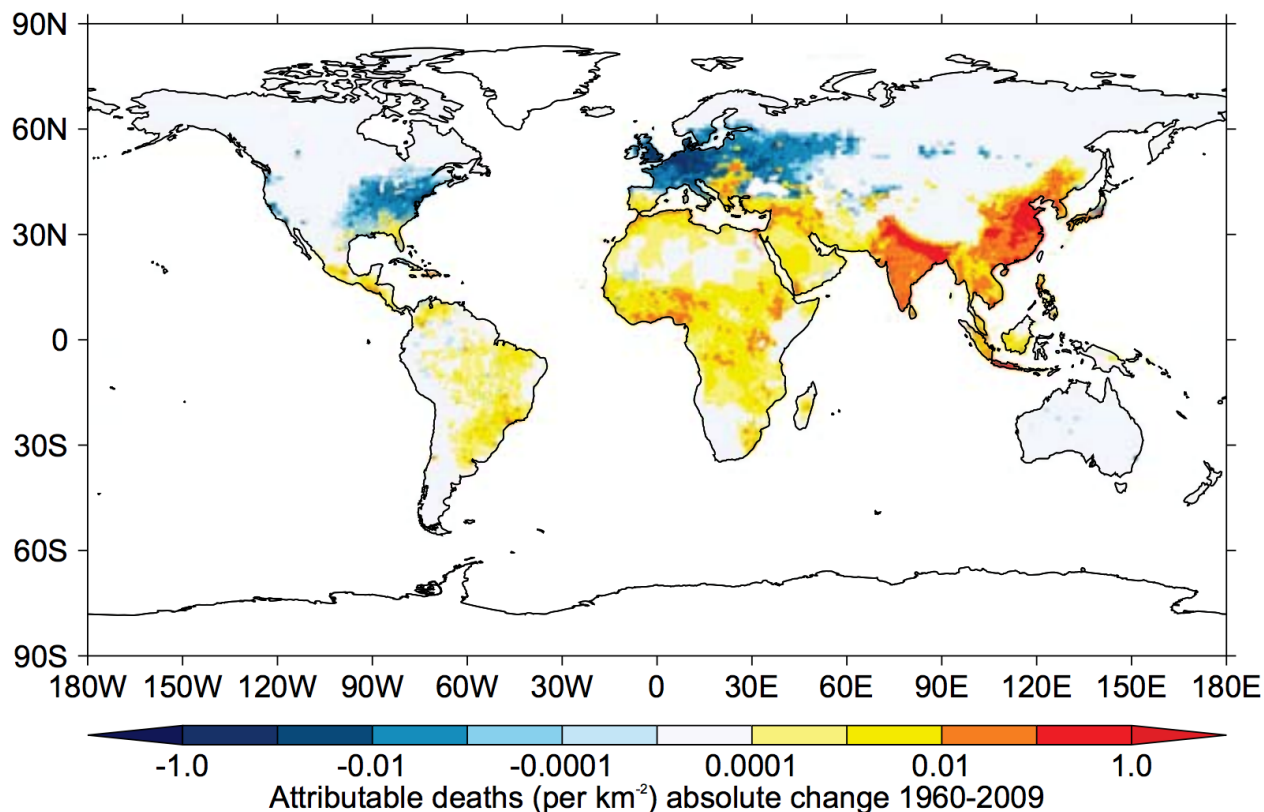
Fig. 10 Changes in deaths globally due to particulates for the period 1960 to 2009 (from ref.[58])
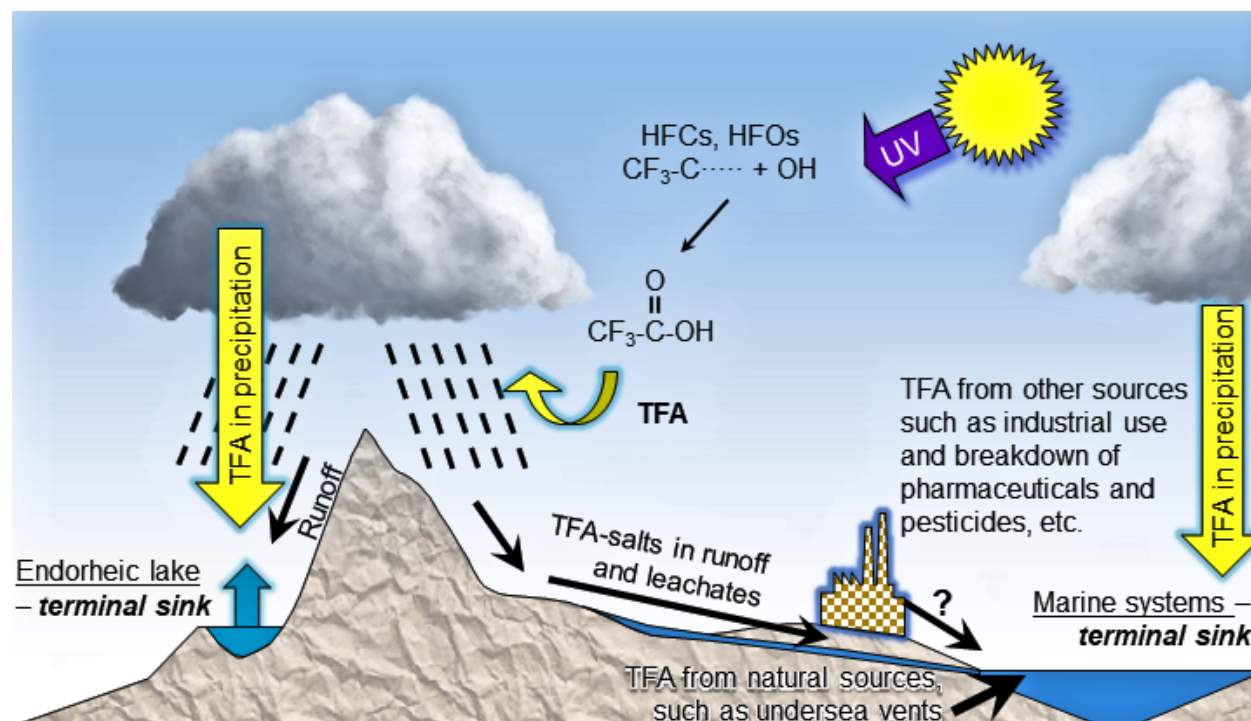

Fig. 11 Trifluoracetic acid (TFA) formed from HFCs and HFOs in the atmosphere will rapidly partition from air to water in the atmosphere. It will combine with cations in soil and surface water and accumulate in endorheic water bodies (salt lakes) and the oceans. TFA produced from

HFCs and HFOs can be reasonably well quantified but this is not true for natural sources and other chemicals used by humans (modified from ref[187], with permission).