

Changes in tropospheric composition and air quality due to stratospheric ozone depletion †

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Increased UV-B through stratospheric ozone depletion leads to an increased chemical activity in the lower atmosphere (the troposphere). The effect of stratospheric ozone depletion on tropospheric ozone is small (though significant) compared to the ozone generated anthropogenically in areas already experiencing air pollution. Modeling and experimental studies suggest that the impacts of stratospheric ozone depletion on tropospheric ozone are different at different altitudes and for different chemical regimes. As a result the increase in ozone due to stratospheric ozone depletion may be greater in polluted regions. Attributable effects on concentrations are expected only in regions where local emissions make minor contributions. The vertical distribution of NO_x (NO + NO₂), the emission of volatile organic compounds and the abundance of water vapor, are important influencing factors. The long-term nature of stratospheric ozone depletion means that even a small increase in tropospheric ozone concentration can have a significant impact on human health and the environment.

Trifluoroacetic acid (TFA) and chlorodifluoroacetic acid (CDFA) are produced by the atmospheric degradation of hydrochlorofluorocarbons (HCFCs) and hydrofluorocarbons (HFCs). TFA has been measured in rain, rivers, lakes, and oceans, the ultimate sink for these and related compounds. Significant anthropogenic sources of TFA other than degradation HCFCs and HFCs have been identified. Toxicity tests under field conditions indicate that the concentrations of TFA and CDFA currently produced by the atmospheric degradation of HFCs and HCFCs do not present a risk to human health and the environment.

The impact of the interaction between ozone depletion and future climate change is complex and a significant area of current research. For air quality and tropospheric composition, a range of physical parameters such as temperature, cloudiness and atmospheric transport will modify the impact of UV-B. Changes in the chemical composition of the atmosphere including aerosols will also have an impact. For example, tropospheric OH is the 'cleaning' agent of the troposphere. While increased UV-B increases the OH concentration, increases in concentration of gases like methane, carbon monoxide and volatile organic compounds will act as sinks for OH in troposphere and hence change air quality and chemical composition in the troposphere. Also, changes in the aerosol content of the atmosphere resulting from global climate change may affect ozone photolysis rate coefficients and hence reduce or increase tropospheric ozone concentrations.

Introduction

Changes in air pollutants such as ozone (O₃), nitrogen oxides (NO_x), hydrogen peroxide (H₂O₂), formaldehyde (HCHO) and nitric acid (HNO₃) in the lower atmosphere (troposphere) have important implications for human and environmental health. Understanding the factors that influence the concentrations of these pollutants is important for the protection of humans and their environment. The relationship between ultraviolet-B radiation (UV-B, 280–315 nm) and the photolysis of atmospheric trace gases such as O₃, NO₂, H₂O₂, HCHO, and HNO₃ in the troposphere was reviewed in previous reports.^{1,2}

The previous report^{1,3} concluded that stratospheric ozone depletion causes changes in the chemical composition of the atmosphere through increased penetration of UV-B to the troposphere. Tropospheric ozone levels are sensitive to local concentrations of NO_x and hydrocarbons. Increased UV-B is expected to increase the concentration of hydroxyl and other peroxy radicals and result in faster removal of pollutants. The

effects of UV-B increases on tropospheric constituents like ozone, while not negligible, will be difficult to detect because the concentrations of these species are also influenced by many other variable factors, such as anthropogenic emissions.

The replacement of the ozone-depleting chlorofluorocarbon (CFC) refrigerants with less persistent products such as the hydrochlorofluorocarbons (HCFCs) and hydrofluorocarbons (HFCs) has also raised the question of the environmental significance of their breakdown products. In the previous report the environmental and human health significance of these breakdown products were not completely understood.¹

Here we present a brief review of relevant new information. In addition, a discussion of interactions between tropospheric composition, air quality, changes due to ozone depletion, and climate change has been added. The conclusions are based on the current state of the science.

Changes in photodissociation rate coefficients due to stratospheric ozone depletion

Reductions in stratospheric ozone cause increased penetration of UV-B radiation to the troposphere and increases the photodissociation rates of tropospheric ozone and other trace gases

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such as NO_x . The trends in photodissociation rates of tropospheric ozone and other chemical species were estimated by De Winter-Sorkina^{4,5} for the period 11/1978 to 4/1993, based on the TOMS/Nimbus-7 (version 7) total ozone data and Stratospheric Aerosol and Gas Experiment (SAGE) ozone trends together with experimental ozonesonde data. A three-dimensional chemical transport model was used to investigate the impact of changing photodissociation rates on tropospheric composition. The results showed that the trends in the daytime average photodissociation rate coefficient of ozone were $+18.0 \pm 4.7\%$ (95% confidence interval) per decade in February at the surface for zonal averages between 40 and 50°N. More importantly, there were also significant tropospheric ozone photodissociation trends of between +3 and 15% per decade at the surface at northern mid-latitudes in spring, summer and the first half of autumn. Significant long-term trends in tropospheric ozone photodissociation rate coefficients below 50°S were predicted throughout the year, peaking at $+42 \pm 15\%$ per decade at the surface in October between 60 and 70°S. In general, the trends in ozone photodissociation rate coefficients due to stratospheric ozone depletion increased with altitude, reaching their maximum in winter-spring at about 8–12 km, the exact altitude depending on latitude. Overall, the trend in daytime average ozone photolysis rate coefficients is larger than the change in stratospheric ozone that caused them.

De Winter-Sorkina⁵ found that the magnitude of the trends for the photodissociation rate coefficients for other atmospheric trace gases varied, although the regions and times of maximum impact were broadly the same as that for ozone photolysis. For example, at 40°N, the trends at the surface in January–April were calculated to be around +8% per decade for the photolysis of CH_3CHO to produce CH_3 , +6% per decade for HNO_3 and +2% per decade for H_2O_2 , CH_3OOH and N_2O_5 . Once again, much larger changes are observed at 60 to 70°S with a trend of near 25% per decade for the photolysis reaction for CH_3CHO listed above. The relative uncertainty in these estimates is similar to that of the ozone photolysis estimates.

Long-term global changes in tropospheric concentrations of ozone due to stratospheric ozone depletion have been calculated using the MOGUNTIA model⁵ by assuming that trace gas emissions remained constant. This gave a trend of $+2.4 \pm 1.3\%$ per decade for OH, $-1.8 \pm 1.3\%$ per decade for CO and $-0.8 \pm 0.7\%$ per decade for O_3 and CH_4 . Thus, a small decrease in global tropospheric ozone was estimated due to stratospheric ozone reduction, with the tropospheric trends small or comparable to other factors, especially the changes due to anthropogenic emissions. The response of global OH concentrations to stratospheric ozone loss was found to be equivalent to the effect of a possible 10% NO_x emission increase and 85% of this response was found to be equal to an estimated 6.5% CO decrease or a possible 10% tropical H_2O increase.⁶

For all these chemical compounds, the observable impact of stratospheric ozone depletion on tropospheric trends can be overshadowed by local or regional emissions. Only in regions where local emissions make minor contributions will long-term effects due to stratospheric ozone reduction be clearly discernable from the changes due to other sources. However, the cumulative effect of long-term increases in ground level ozone concentrations may cause significant adverse environmental effects.

Changes in chemical composition of the troposphere due to stratospheric ozone depletion

The influence of increased UV-B photolysis on tropospheric chemical composition has been recognized more widely and has been examined in more detail recently through modeling and experimental studies. They include perturbations due to stratospheric ozone depletion as well as the influence of various tropospheric factors. The physical and chemical reactions

involved in these processes have been reviewed previously.¹ The net ozone production or loss depends mainly on the NO_x concentration. Modeling studies suggest that an increase in UV-B (due to stratospheric ozone depletion) causes a decrease in tropospheric ozone in the clean environment of the southern hemisphere (very low NO_x), and an increase in tropospheric ozone in the polluted areas of the northern hemisphere.^{7,8} The near UV photolysis of other chemical species such as HCHO ($\lambda < 330$ nm) or CH_3CHO ($\lambda < 330$ nm), which are secondary sources of radicals, can also affect the surface ozone concentrations in an indirect way.⁹

However, in certain cases, the photochemical link between total and surface ozone can be perturbed by weather-related atmospheric transport. For example, low total-column ozone could be related to upper troposphere/lower stratosphere anticyclones associated on the one hand with large scale subsidence of ozone rich air masses, and on the other hand with fair weather conditions with large ozone formation.¹⁰ Also, high total ozone column could be related to cut-off lows or upper troughs, synoptic systems associated with tropopause folds which may enhance the tropospheric ozone levels significantly within a few hours.¹¹

Although the effects of change in UV intensity resulting from stratospheric ozone depletion on tropospheric chemistry are theoretically well understood from model calculations, there is a paucity of observational evidence. Stratospheric ozone is controlled by dynamical and chemical factors acting on different time scales and so the influence on surface UV-B and related tropospheric ozone chemistry should also act on different time scales. Long-term decreases in surface ozone in polar regions have been reviewed.¹ The first observational evidence of changes in tropical tropospheric ozone associated with stratospheric ozone changes on a time scale of an 11 year solar cycle has been identified using the TCO (tropospheric column ozone) data derived from the TOMS satellite data.¹²

On a day-to-day basis, *i.e.*, when circulation in the upper troposphere/lower stratosphere causes variations in total ozone column,¹³ there is even less observational evidence of the photochemical link between total and surface ozone due to the long lifetime of ozone and the fact that the photochemical link between total-column and surface ozone can be perturbed by dynamic coupling as discussed above. Thus, the photochemical influence of changing UV-B on *in-situ* ozone production and loss rate of 10% can easily be masked by transport effects, especially in late winter and at high latitudes.¹⁴ In addition, the sensitivity of a regional scale photochemistry model to changes in UV radiation, caused by moderate variations in either the total ozone column (25 DU) or in the aerosol optical depth, was found to be small (< 0.5 nmol mol⁻¹ for monthly average, June 1996).¹⁵ Only under sufficiently large changes of UV-B (that is, a large change in total ozone column) and suitable meteorological conditions can such an influence be detected, as presented in a case study on Swiss mountains in late winter¹⁴ and in a case study on Crete, Greece, during the PAUR II (Photochemical Activity and solar Ultraviolet Radiation) campaign.¹⁶

A modeling study by Ma and van Weele¹⁷ emphasized that influences on tropospheric pollutants are not isolated from interactions with other factors. It was found that the response of net ozone production to stratospheric ozone depletion was different at different altitudes and for different chemical coherent regimes. In addition to the surface concentration of NO_x , the vertical distribution of NO_x as well as the emissions of non-methane hydrocarbons (NMHCs), the abundance of water vapor, and ozone itself, are important factors. The threshold NO_x concentration, at which the response to stratospheric ozone depletion of the net ozone production changes from negative to positive, depends on the chemical regime and varies during a day, but it is typically about 1 nmol mol⁻¹ in a 24 h period. That is, it is around 100 times larger than

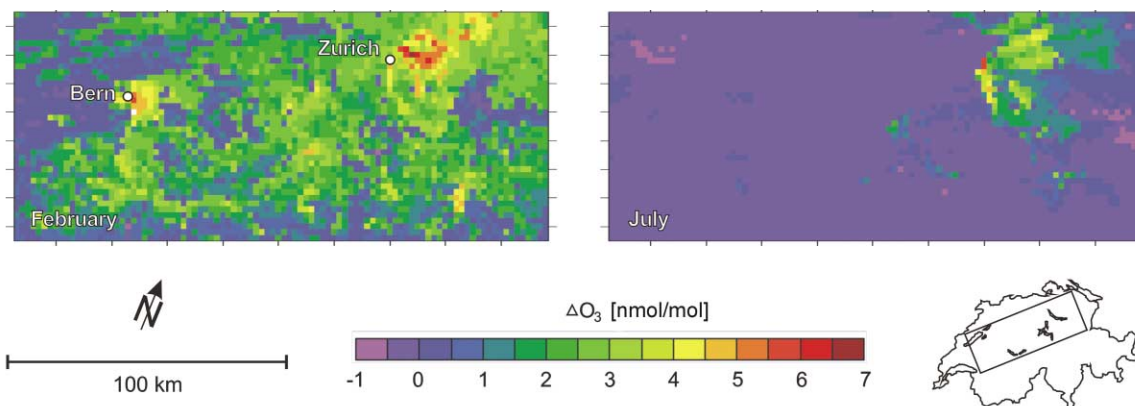


Fig. 1 Ground level ozone production due to stratospheric ozone depletion. The calculations are for the region of Switzerland shown in the map. The left hand panel shows the calculated impact of ozone depletion from 400 DU to 240 DU for a simulation of an event in February 1998. The right hand panel shows a calculation for 30 July 1993, with an ozone decrease from 360 DU to 280 DU. (Figure based on the work of Brönnimann *et al.*²⁰)

the threshold NO_x concentration for ozone production. Water vapour and NMHCs can further modulate tropospheric ozone production.

The importance of the vertical distribution of NO_x on the net ozone production due to stratospheric ozone depletion in the polluted boundary region is illustrated in the observations of Brönnimann and co-authors^{14,18,19} at two elevated sites in Europe (Chaumont; 1,140 m above sea level (ASL), and Rigi; 1,030 m ASL) where enhanced near-surface ozone concentrations were recorded. Mean diurnal cycles of ozone concentration showed a strong increase from late morning to late afternoon and, at the same time a decrease at the high alpine site Jungfraujoch (3,580 m ASL). The different diurnal ozone cycles can both be explained photochemically by taking into account the large difference in NO_x concentrations (about two orders of magnitude). Brönnimann *et al.*¹⁸ also attempted to quantify the effect of changing UV-B radiation on surface ozone peaks in a day-to-day scale using a time series of measurement at the mountain sites. Seven years of ozone, NO and NO_x data, meteorological measurements from Chaumont, total ozone and UV-B measurements from Arosa (1,847 m ASL), and surface albedo from satellite observations were investigated. The study was restricted to fair weather days with moderately high NO_x concentrations. The estimated net effect on ozone peaks is normally within a range of 4 nmol mol^{-1} , a range of about 6 nmol mol^{-1} is predicted for large UV-B changes. This is greater than 5% of the Swiss Air Quality 1 hour standard. Assuming that all other factors are constant, the trend due to the stratospheric ozone depletion observed in the last decades was found to be less than $0.12 \text{ nmol mol}^{-1} \text{ yr}^{-1}$. A 3-D mesoscale photochemical model (Metphomod) gave similar results to these measurements.²⁰

The Metphomod model calculations were performed for the entire Swiss Plateau,²⁰ and for the large stratospheric ozone depletion used in their calculation the changes in near ground level ozone was $0\text{--}3 \text{ nmol mol}^{-1}$ (see Fig. 1). However, for the city of Zurich, they estimated changes of around $+5 \text{ nmol mol}^{-1}$ for both the winter and summer 'ozone event' they studied. In the urban airshed, such an increase could not simply be assigned, based on measurements, to stratospheric ozone depletion. However, it does represent a significant contribution to the overall ozone loading.

Pirjola²¹ used a Lagrangian model to assess the interaction of elevated UV-B, volatile organic compound (VOC) emissions from plants (isoprene and some terpenes) and SO_2 . In the base case for emission rates of these organics, Pirjola reported a linear relationship between sulfate particle numbers and UV-B, with a 2.5-fold increase in particles for a 50% increase in UV-B. Under these conditions, the OH radical concentration increased by 9.8%. The concentration of biogenic organics is expected to increase with increasing CO_2 , and Pirjola has investigated this

also. The model shows a decrease in OH and particle formation and an increase in ozone with an increase in organics. The increase in particle formation through a UV-B ($X_{\text{UVB}}(\%)$) enhancement can be offset by an increase in biogenic organics ($Y_{\text{Org}}(\%)$) as described by

$$Y_{\text{Org}} = 1.23 X_{\text{UVB}} + 3.34 \quad (1)$$

Humidity, temperature, SO_2 emissions, and pre-existing particle concentrations were found to have little impact on the relationship.

The expected impact of stratospheric ozone depletion on formaldehyde (HCHO) is unclear, as both its production and destruction will be enhanced by ozone depletion. However, recent measurements of formaldehyde from Neumayer station in Antarctica²² found roughly twice the HCHO concentration in the spring during the ozone hole period than observed at a corresponding time in autumn. This suggests that, at least under Antarctic conditions, the concentration is enhanced by stratospheric ozone depletion.

Over the past few years, significant advances have been made in understanding how aerosols affect actinic fluxes and therefore tropospheric chemistry. This advance benefited greatly from the UV monitoring networks that have been put in place to monitor stratospheric ozone depletion. Dickerson *et al.*²³ have shown that scattering by sulfate aerosols actually increases O_3 production in the planetary boundary layer (PBL) of the eastern USA. Absorbing aerosols will reduce photolysis rates and reduce PBL- O_3 production in polluted areas, but may actually reduce PBL- O_3 loss under low NO_x conditions, *i.e.* in more pristine regions.

Atmospheric production, fate, and effects of trifluoroacetic acid and related substances

The hydrochlorofluorocarbons (HCFCs) and hydrofluorocarbons (HFCs) HCFC-123, HCFC-124 and HFC-134a degrade to give trifluoroacetic acid (TFA).¹ Use of HFCs may increase as the products are used to replace other products such as SF_6 , resulting in increased production of TFA. Environmental TFA can also be produced during the breakdown of other organofluorine compounds released to the atmosphere by human activities, *e.g.*, halothane and isoflurane anaesthetics. TFA is also produced through the breakdown of TFM (a lampricide), trifluoromethyl-containing agrochemicals such as trifluralin, and Teflon.^{24,25} TFA is widely used in the chemical industry in processes where it is either consumed or becomes part of a chemical waste stream. A natural source of TFA has also been proposed from the weathering of fluorites that have been found to contain CF_4 .²⁶ However, TFA has not yet been detected from these materials.

Residues of TFA have been observed in water and air samples from many geographical areas (USA, Canada, Australia, South Africa, Germany, Israel, Ireland, France, Switzerland, Finland, and China) and show that TFA is a ubiquitous contaminant of the hydrosphere. Concentrations have been reported up to 0.04 mg L^{-1} .²⁷ Another halogenated acetic acid with similar properties to TFA is chlorodifluoroacetic acid (CDFA), which is also the breakdown product of HCFCs, and is formed *via* degradative reactions occurring in the atmosphere.²⁸ Measured concentrations of CDFA are smaller than TFA. A maximum of 0.0002 ng L^{-1} was reported in water samples taken in Canada.²⁸ Both of these halogenated acetic acids (HAAs) are persistent in the environment.

Fate and effects of TFA and CDFA in the ecosystem

Fate. TFA and related products are water-soluble, have negligible vapor pressure, and accumulate in surface-waters (Fig. 2).¹ Because of their great water solubility and low

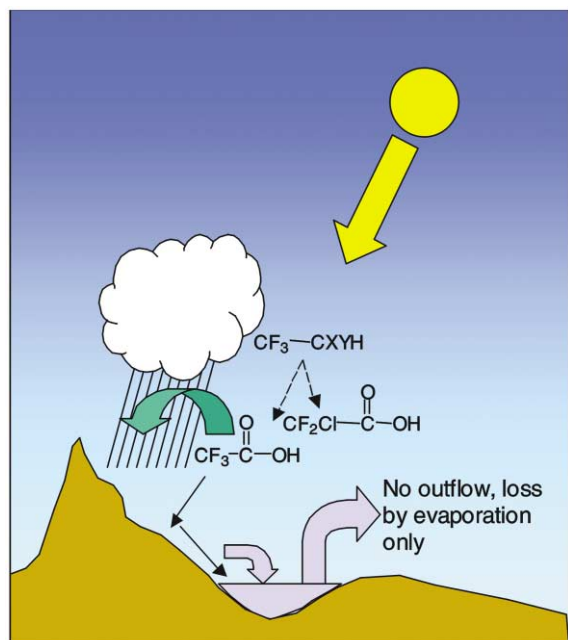


Fig. 2 Illustration of the formation of persistent, water-soluble breakdown products of the HFCs and HCFCs ($\text{CF}_3\text{-CXYH}$) and their movement and concentration by evaporation, along with other water-soluble salts to surface waters.

octanol–water partition coefficient (K_{ow}), TFA and related halogenated acetic acids (HAAs) are not likely to bioconcentrate, bioaccumulate, or biomagnify through the aquatic food chain, although accumulation in terrestrial plants has been observed following hydroponic exposures.¹ TFA is a stable ion in the aqueous phase and no significant loss process such as hydrolysis, photolysis, or formation of insoluble salts has been identified.¹ Degradation of TFA may occur under certain conditions in the laboratory.^{1,29} However, these processes appear to be less relevant in the field. Studies in aquatic microcosms containing sediments and biota have not demonstrated any measurable dissipation of TFA over periods of up to two years.³⁰ Similarly, CDFA exhibited no degradation in outdoor aquatic microcosms over the period of almost a year.³¹

Toxicity to aquatic organisms. TFA is not considered highly toxic to aquatic plants and animals in acute laboratory studies.¹ Previous field studies on TFA showed that it did not alter the algal species composition in the stream microcosm.¹ Studies on the common aquatic plants *Myriophyllum spicatum* and *M. sibiricum* in outdoor microcosms exposed to TFA for 49 days showed no signs of toxicity at concentrations up to

10 mg L^{-1} TFA.³² Citrate concentrations in the plants were monitored and no increases were observed in exposed *versus* controls, implying that the citric acid cycle was not impacted at the concentrations used.³³ Similarly, CDFA demonstrated low toxicity to aquatic plants in laboratory studies EC_{25} s of $14\text{--}18 \text{ mg L}^{-1}$ in *Myriophyllum* spp.²⁸ and in microcosm studies, either on its own or in combination with TFA.³¹

Toxicity to terrestrial organisms. TFA is not considered highly toxic to terrestrial plants. No effects were observed in a variety of crop plants at concentrations ranging from 1 to $1,000 \text{ mg L}^{-1}$.¹ Effect concentrations for CDFA in terrestrial plants have not been reported in the literature. TFA is not metabolized in mammalian systems but is rapidly excreted. As the neutralized salt, TFA is not considered highly toxic to mammals *via* the inhalation or the oral route.¹ It is also not mutagenic and, although it has not been tested, is not considered a likely carcinogen.

Risk assessment. The previous risk assessment of TFA was based on an estimated average concentration of 0.0001 mg L^{-1} in rainwater, by the year 2020.¹ With additional sources of TFA now identified, concentrations may be somewhat higher than this estimate. These concentrations are not expected to present risks to terrestrial plants through foliar soil absorption routes of exposure.

Based on a no observed effect concentration (NOEC) of about 0.10 mg L^{-1} (as TFA) for the most sensitive standard algal species, *Selenastrum capricornutum*, a predicted no effect concentration in aquatic systems ($\text{PNEC}_{\text{aqua}}$) of 0.1 mg L^{-1} was proposed.¹ Lack of effects of TFA on plants in aquatic microcosms under field conditions at concentrations up to 10 mg L^{-1} confirms low risk and the conservatism of the suggested $\text{PNEC}_{\text{aqua}}$.

On the basis of discussions in the previous review¹ and more recent information, risks from TFA in the ecosystem are judged to be low. Although less data exist for CDFA, risks from this substance are also judged to be low.

Uncertainties as to the sources of TFA in the environment have been reduced through the identification of other pathways of formation. However, CDFA has been identified as a persistent degradation product of the HCFCs. CDFA is not considered toxic to aquatic plants but sensitivity of other species has not been reported. Although additional data on environmental concentrations of TFA (and CDFA) have been reported, continuing monitoring is recommended to confirm trends in surface and rainwater.

Interactions with global climate change

As discussed previously in this volume³⁴ and more fully elsewhere,^{35,36} interactions between climate change and stratospheric ozone depletion may occur through many chemical, radiative and dynamic mechanisms. For example, changes in temperature will change reaction rates, variations in cloudiness will have a direct impact upon photolysis rates, and long-range transport can change the chemical composition. Several authors have reviewed potential interactions between ozone depletion and climate change.^{37–40} The review by Staehelin *et al.*³⁸ concluded that the interactions between stratospheric ozone depletion and climate change would be a productive research domain in the future. Only a few examples of the possible interactions will be given here.

Variations in ozone concentration, in particular in the upper troposphere and lower stratosphere, would be a crucial issue in the next decades because of links to climate change. Stratospheric ozone depletion leads to an increase in UV-B fluxes in the troposphere, which affect the photochemistry in the troposphere, especially the OH radicals. Increases in concentration of some greenhouse gases will act as sinks for OH in the troposphere, thus changing air quality and composition in the

troposphere as suggested in the previous UNEP assessment¹ and by Ma and van Weele.¹⁷ Studies by Prinn *et al.*⁴¹ using methylchloroform, suggest a recent decrease in global OH that may be due to anthropogenic emissions. An increase in atmospheric CO₂ concentration would accelerate photosynthesis (CO₂ fertilization) which might enhance the emissions of biogenic VOC's in forests and other natural ecological systems, and their oxidation reactions with hydroxyl radical which will act as a sink for OH.²¹

Other sources of tropospheric air pollutants may be affected by global warming. It is known that local and large-scale biomass fires such as are used for land-clearing are rich sources of NO_x, CO, CH₄, and other NMHCs, leading to enhanced tropospheric ozone production.⁴² During the 1994 South-East Asian haze episode (September–October 1994), measurements in Asia showed increased tropospheric concentrations of NO_x (up to double) and ozone (~20%).⁴³ Besides extensive forest fires, local burning of biomass and fossil fuels also would have contributed to this. Thus, climate changes resulting from global warming may increase the risk of large-scale forest and brush-fires which affect concentrations of tropospheric air pollutants. Also, forest-fires induce haze aerosols that can increase multiple scattering, thus improving the efficiency of UV-B absorption of the boundary layer ozone. However, ozone production in the boundary layer may also be hindered by UV-B absorption by aerosols.⁴⁴

Changes in the aerosol content of the atmosphere associated with global change may affect ozone photolysis rates⁴⁵ and hence the tropospheric ozone levels. It was calculated that soot particles and mineral dust particles can reduce the ozone photolysis rate by 6% to 11%,⁴⁵ and reductions of up to 50% have been estimated elsewhere.^{44,46} During the Sahara dust events in the PAUR II campaign, Balis *et al.*⁴⁷ reported greatest effects on production of excited oxygen atoms (J(O¹D)) during days with high aerosol content. Box model calculations for a case study during the same experiment at Crete were carried out. The model indicated that the presence of absorbing aerosols offset part of the increase in local net ozone production rate caused by the decrease in total column ozone concentration when enough NO_x was present.¹⁶ A sensitivity analysis of photolysis rates and ozone production in the troposphere in response to aerosol properties used a coupled transport-chemistry-radiative transfer model,⁴⁸ and reported reductions in ground level ozone concentrations of up to 70% in the presence of absorbing aerosols in the boundary layer (Fig. 3).

Although it is recognized that tropospheric ozone and other air pollutants are important with respect to human health and

the environment, these pollutants are largely anthropogenic and concentrations vary in response to local human activity. The impact of the coupling of stratospheric ozone depletion and climate change on concentrations of pollutant will be difficult to quantify experimentally, even though it could be significant, especially in the regions where air quality has a large impact on human health.

Since global warming is expected to increase demand for refrigerants, the breakdown products of these substances are likely to increase in concentration in the troposphere at a greater than expected rate. While this could increase concentrations of persistent breakdown products such as TFA and CDFA, inputs to most environments would still be small enough (by a factor of about 1,000) that biologically significant concentrations would not be expected to occur in flowing surface waters. However, in areas where there is little or no outflow and increased evaporation due to increased surface temperatures, these products are expected to increase in concentration. While this may present a risk to aquatic organisms, these areas would also experience increases in concentrations of other water-soluble materials such as has already occurred in salt lakes and playas. The effects of these naturally occurring salts and other materials would likely be greater and more biologically significant than those of TFA or CDFA and would occur in the absence of these substances anyway. The results of this interaction with global climate change are judged to be of low significance.

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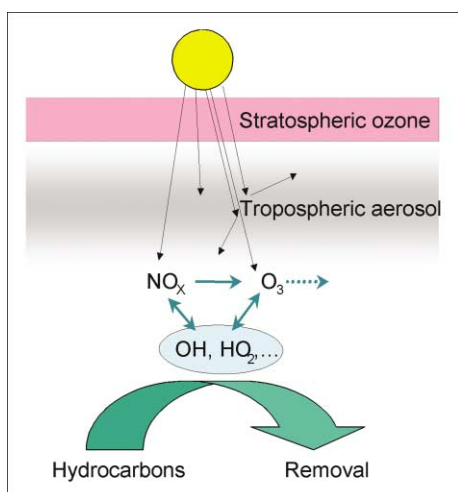


Fig. 3 Illustration of the impact of tropospheric aerosols upon atmospheric chemistry. The aerosol can reduce the intensity of radiation, leading to a reduction in ozone production. Such a reduction offsets the impact of stratospheric ozone depletion.

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