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Abstract

In the North Sea region, poor air quality has serious implications for human health and the related societal costs are considerable. The state of air pollution is often used as a proxy for air quality. This chapter focuses on the two atmospheric pollutants of most significance to human health in Europe—particulate matter and ground-level ozone. These are also important ‘climate forcers’. In the North Sea area, the effects on air quality of emission changes since preindustrial times are stronger than the effects of climate change. According to model simulations, this is also the case for future air quality in the North Sea region, but substantial variation in model results implies considerable uncertainty. Short-term events such as heat waves can have substantial impacts on air quality and some regional climate models suggest that heat waves may become more frequent in the coming decades. If the reductions in air pollutant emissions expected through increasingly stringent policy measures are not achieved, any increase in the severity or frequency of heat waves may have severe consequences for air quality. Climate and air quality interact in several ways and mitigation optimised for a climate or air quality target in isolation could have synergistic or antagonistic effects.

16.1 Introduction

The state of air pollution is often expressed as air quality. The concentrations of gaseous pollutants and particulate matter are then used as a measure of air quality. However, it is often not meaningful to discuss air quality without addressing the multiple impacts of air pollution. Major air pollutants may be clustered according to their properties and impacts, and this is shown in Fig. 16.1. After being emitted into the atmosphere pollutants undergo chemical oxidation and form new compounds with different properties and impacts. Pollutants then remain in the atmosphere until they are removed through cloud and precipitation processes or by

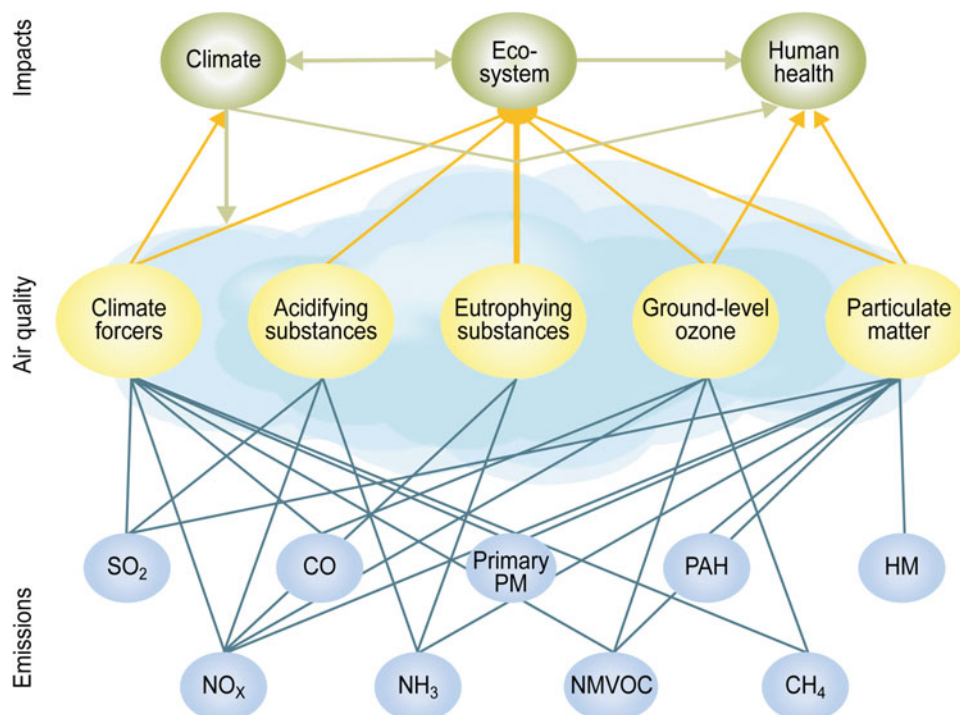
direct deposition to the earth’s surface. Differences in chemical reactivity and removal rates result in atmospheric lifetimes ranging from seconds to months. Air quality and related impacts are therefore influenced by local meteorological features, regional (transboundary) processes, and intercontinental transport. The pathway from emissions to impacts is complex. The focus in this chapter is limited to impacts on human health, climate, and climate-air quality interactions and mainly excludes impacts on ecosystems (acidification, eutrophication, carbon sequestration, crops, and vegetation) and materials. Impacts of climate change on ecosystems are covered in Chaps. 8, 9, 10, and 11.

Air quality and climate interact in several ways. Air pollutants can affect climate both directly and indirectly through their influence on the radiative balance of the atmosphere. Primary particulate matter (primary PM, Fig. 16.1) affects climate directly, while pollutants such as carbon monoxide (CO), non-methane volatile organic compounds (nmVOC), polycyclic aromatic hydrocarbons (PAH), nitrogen oxides (NO_x), sulphur dioxide (SO₂) and ammonia (NH₃) although

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Fig. 16.1 Major air pollutants, clustered according to impacts on climate, ecosystems and human health (EEA 2012). From *left to right* the pollutants shown are as follows: sulphur dioxide (SO₂), nitrogen oxides (NO_x), carbon monoxide (CO), ammonia (NH₃), particulate matter (PM), non-methane volatile organic compounds (NMVOC), polycyclic aromatic hydrocarbons (PAH), methane (CH₄), heavy metals (HM)



having a negligible direct radiative ('greenhouse') effect, have an important indirect climate effect by acting as precursors for components that are both harmful pollutants and act as 'climate forcers' (e.g. ozone and particulate matter). On the other hand, air quality is also sensitive to climate change itself, since climate change drives changes in the physical and chemical properties of the earth and atmosphere. Climate policies also imply energy efficiency and technical measures that change emissions of air pollutants. Equally, air quality mitigation measures will impact on greenhouse gas emissions.

The main chemical components addressed in this chapter are PM and ground level ozone (O₃). These are generally recognised as the two pollutants that most significantly affect human health in Europe. The impacts of long-term and peak exposure to these pollutants range in severity from impairing the respiratory system to premature death. Particulate matter in the atmosphere originates from direct emissions (e.g. black and organic carbon, sea salt, dust, pollen) or is derived from chemical reactions involving precursor gases such as SO₂, NO_x, NH₃, PAH and nmVOC. The particles of greatest human health concern are 10 µm in diameter or less (PM₁₀). Of particular concern are those 2.5 µm or less (PM_{2.5}) since these could pass from the lungs into the bloodstream. The size and sign of the particulate climate effect (i.e. the particulate-driven temperature change) varies according to particle size, composition, shape, and altitude, and the albedo of the underlying surface. Particles can also change precipitation patterns and surface albedo.

Ozone is a secondary pollutant and greenhouse gas formed by complex chemical reactions involving NO_x, nmVOC, CO and methane (CH₄). In addition to its impact on human health, high O₃ levels may damage plants, affect agriculture and forest growth, and impact on CO₂ uptake. Compared to O₃ and PM, CH₄ is a relatively long-lived and thus well-mixed greenhouse gas and one link with regional air quality is that short-lived air pollutants such as NO_x, nmVOC, and CO may influence its chemical removal in the atmosphere. Changes in CH₄ concentration in turn affect the atmospheric oxidation capacity and thereby the speed of chemical cycles and removal of pollutants. The rise in global CH₄ levels over recent decades has contributed to rising background O₃ concentrations in the northern hemisphere.

16.2 Current Status

Poor air quality in Europe is a serious human health issue and the related external costs (costs imposed by a producer or a consumer on another producer or consumer, outside of any market transaction between them) are considerable. In 2010, annual premature mortalities were over 400,000 in the EU area and the total external costs of the health impacts were estimated at EUR 330–940 billion (EU 2013). Similar estimates are reported in other studies (Watkiss et al. 2005; Anenberg et al. 2010; Amann et al. 2011; Brandt et al. 2013a; Fang et al. 2013; Silva et al. 2013). Particulate matter is the principal pollutant in terms of human health impacts.

Fig. 16.2 Estimated number of premature deaths due to air pollution per 2500 km² grid cell in 2000 (Brandt et al. 2013a). The number of premature deaths is dependent both on pollution level and population density

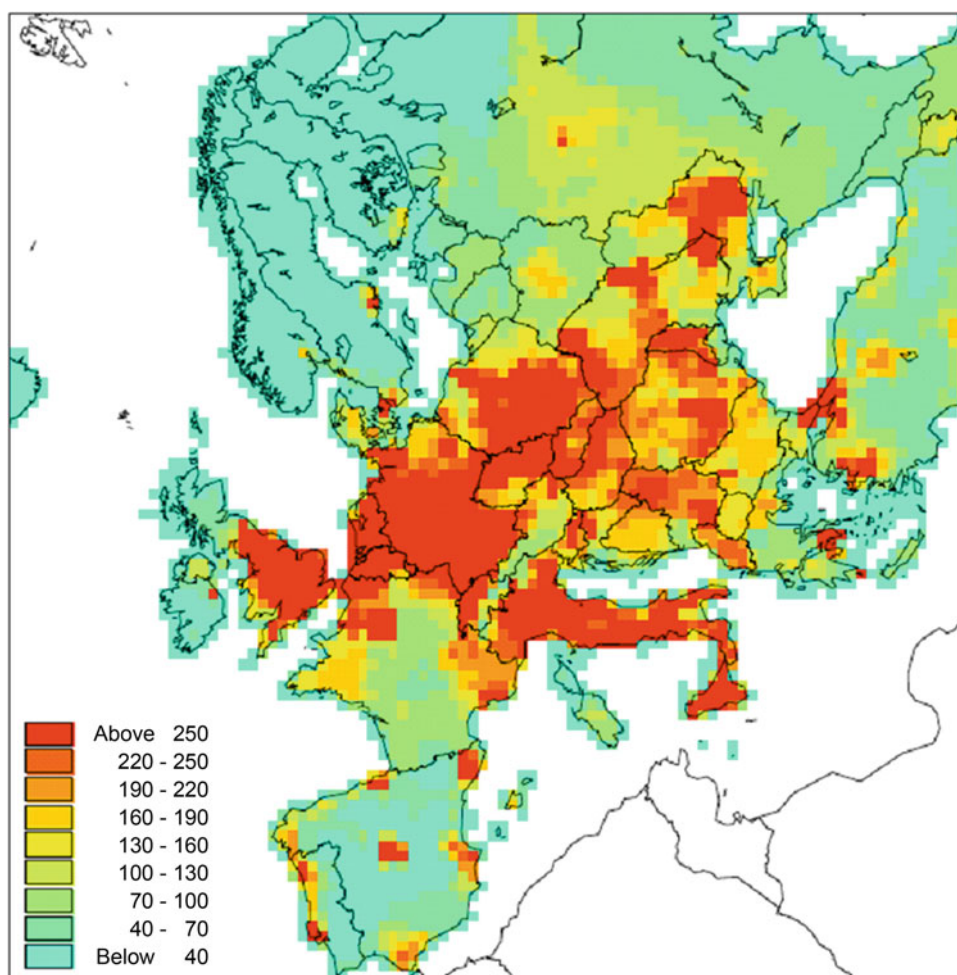


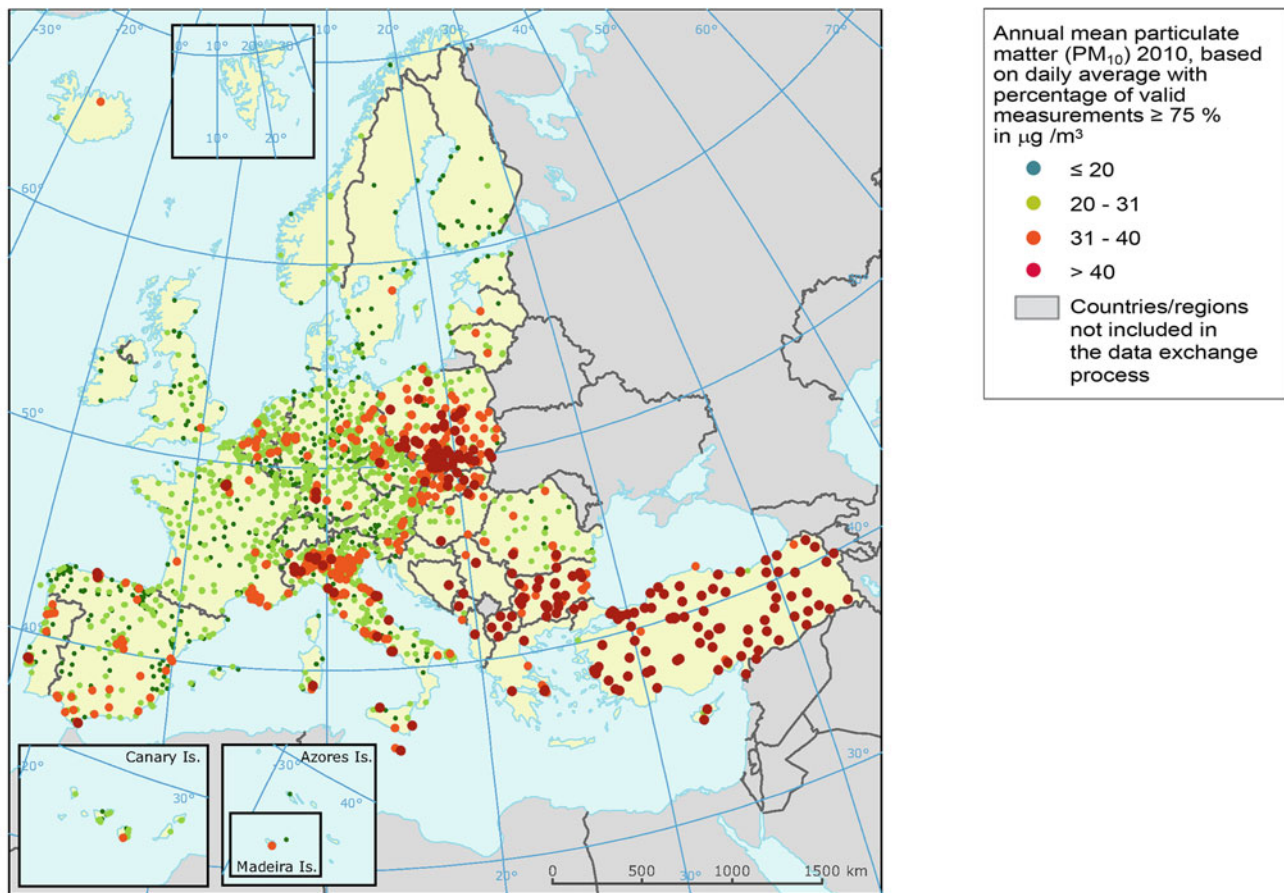
Figure 16.2 shows an example of a model estimate of the geographical distribution of premature deaths in Europe in 2000 due to PM and O₃ (Brandt et al. 2013a). The number of premature deaths in highly populated regions of the North Sea countries is relatively high. Brandt et al. (2013a) estimated the number of premature deaths in Europe to have declined from 680,000 in 2000 to 570,000 in 2011. This decline reflects measures resulting in lower pollution levels in some regions of Europe such as the North Sea area (see Sect. 16.3).

16.2.1 Current Air Quality

In the European Union, PM_{2.5} resulting from human activity is estimated to have reduced average life expectancy in 2000 by 8.6 months (EEA 2012). Figure 16.3 shows the annual mean measured concentration of surface PM₁₀ across Europe in 2010. It should be noted that winter 2010 was colder than normal in the North Sea region (Blunden et al. 2011), resulting in higher PM concentrations than expected from a simple extrapolation of the long-term trend (Tsyro et al.

2012). It is clear that some stations in countries adjacent to the North Sea exceeded the EU daily limit value (orange and red dots). Exceedance at one or more stations occurred in 23 EU Member States in 2010. Of the urban EU population, 21 % was exposed to values above the daily limit value (EEA 2012). The World Health Organization (WHO) has a stricter guideline based on the fact that no threshold is found below which no adverse health effects of PM occur. The WHO guideline value was exceeded in most of the monitoring stations in continental Europe (red, orange and green dots). A similar picture emerges for PM_{2.5} but there are fewer measurement sites.

The EU target value for O₃ of 120 µg m⁻³ (daily maximum of 8-hour running mean values not to be exceeded on more than 25 days per year, averaged over three consecutive years) was exceeded at a large number of stations across Europe in 2010 (dark orange and red dots in Fig. 16.4). However, there are few exceedances of the target value in North Sea countries and none along the North Sea coast. Although winter 2010 was particularly cold in this region, summer was normal or slightly warmer than normal (Blunden et al. 2011). Even so, O₃ levels were low compared to



Note: The red dots indicate stations reporting exceedances of the 2005 annual limit value ($40 \mu\text{g}/\text{m}^3$), as set out in the Air Quality Directive (EU, 2008c).
 The orange dots indicate stations reporting exceedances of a statistically derived level ($31 \mu\text{g}/\text{m}^3$) corresponding to the 24-hour limit value, as set out in the Air Quality Directive (EU, 2008c).
 The pale green dots indicate stations reporting exceedances of the WHO air quality guideline for PM_{10} of less than $20 \mu\text{g}/\text{m}^3$ but not in exceedance of limit values as set out in the Air Quality Directive (EU, 2008c).
 The dark green dots indicate stations reporting concentrations below the WHO air quality guideline for PM_{10} and implicitly below the limit values as set out in the Air Quality Directive (EU, 2008c).

Source: AirBase v. 6.

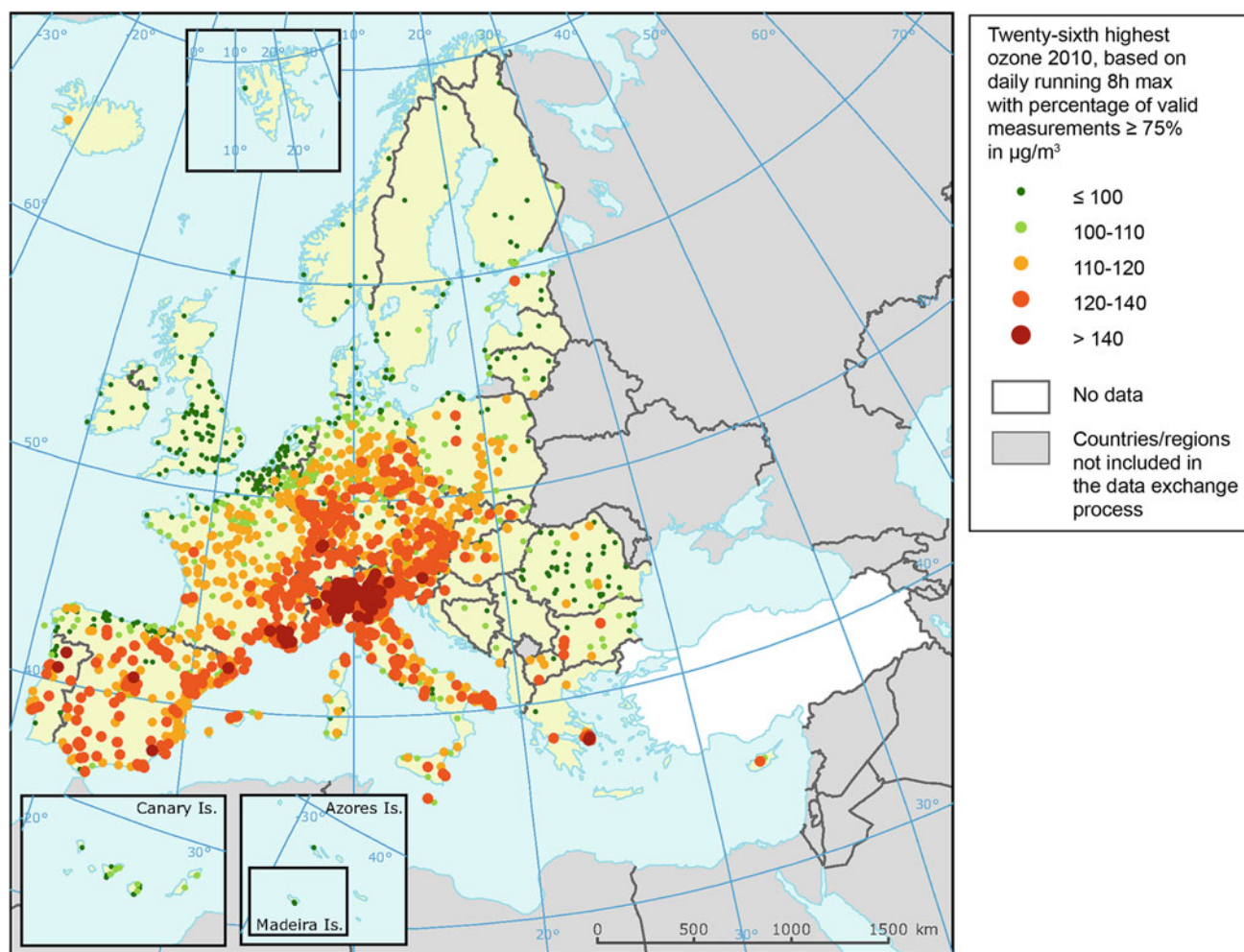
Fig. 16.3 Annual mean surface concentration of PM_{10} across Europe in 2010 (EEA 2012)

the long-term average (Fagerli et al. 2012). Ozone formation is dependent on sunlight and concentrations increase from north to south across Europe. Ozone is also non-linearly dependent on NO_x and VOC concentrations, and is produced and destroyed in a balance between VOC and NO_x , fuelled by solar radiation. In areas with very high nitrogen oxide (NO) emissions, O_3 will be depleted by the reaction with NO (NO_x titration). As the emission plume moves away from the source region, O_3 may be regenerated. Ozone concentrations are therefore generally higher in rural areas some distance from the main NO_x emission sources. The long-term objective for the protection of human health of $120 \mu\text{g m}^{-3}$ (daily maximum of 8-hour running mean values), is exceeded at several stations in all North Sea countries (EEA 2011, 2014). For the ambitious WHO guideline

($100 \mu\text{g m}^{-3}$ 8-hour mean) only two of the 510 rural stations, 3 % of urban background stations and 7 % of traffic stations would not exceed this level (EEA 2012). The EU information threshold ($180 \mu\text{g m}^{-3}$ 1-hour mean) is occasionally exceeded in Belgium, the Netherlands and Denmark but is rarely exceeded in other North Sea countries (EEA 2011, 2014).

16.2.2 Contribution from Emission Sectors and Regions

On a country-by-country basis, the ratio between the contribution to air pollution and deposition from domestic versus non-domestic (transboundary) sources varies



Note: The map shows the proximity of recorded O_3 concentrations to the target value. At sites marked with dark orange and red dots, the twenty-sixth highest daily O_3 concentration exceeded the $120 \mu\text{g}/\text{m}^3$ threshold and the number of allowed exceedances by the target value.

Source: AirBase v. 6.

Fig. 16.4 Twenty-sixth highest daily maximum 8-hour average surface ozone (O_3) concentration recorded at each monitoring station in 2010 (EEA 2012)

substantially, depending on pollutant, local source strength, proximity to major non-domestic sources, and the geographical size of the individual countries (see recent reports by the European Monitoring and Evaluation Programme; EMEP¹).

For $\text{PM}_{2.5}$, contributions to the overall national pollution level are dominated by transport from non-domestic countries (exceptions are Great Britain and Norway, which are heavily influenced by air masses originating over the Atlantic). As an example, the contributions to surface $\text{PM}_{2.5}$ and $\text{PM}_{\text{coarse}}$ ($\text{PM}_{\text{coarse}} = \text{PM}_{10} - \text{PM}_{2.5}$) are shown in Fig. 16.5 for the Netherlands (Gauss et al. 2012). The main contributions are clearly from neighbouring countries. The

contribution from volcanoes is due to the major volcanic eruption in Iceland in 2010.

The picture is more complicated for O_3 and O_3 -derived parameters such as SOMO35.² Countries around the North Sea are some of the highest NO_x emitters in Europe. High NO_x emissions in combination with limited solar insolation during winter at these latitudes results in inefficient photochemical O_3 production and substantial NO_x titration. As a result, the present levels of NO_x emissions will, at least as an annual average, decrease the O_3 burden in the North Sea

¹http://emep.int/mscw/index_mscw.html.

²SOMO35, defined as the annual daily sum of 8-hour running average O_3 concentrations over 35 ppb, is a measure of accumulated annual O_3 concentrations and used as an indicator of health hazards, see EMEP (2013) for details.

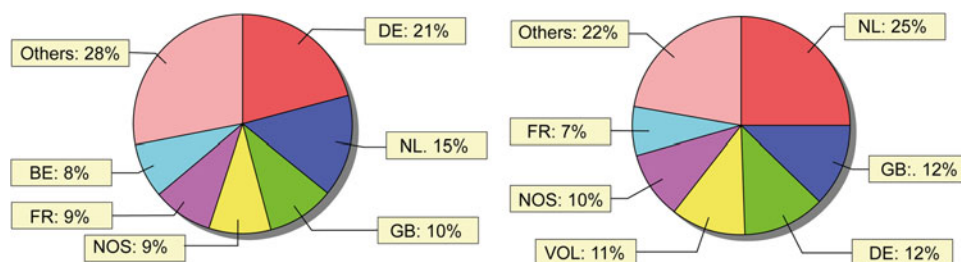


Fig. 16.5 Percentage contribution from individual countries to surface PM_{2.5} (left) and PM_{coarse} (right) in the Netherlands. BE Belgium, FR France, NOS North Sea, DE Germany, NL The Netherlands, GB Great Britain, VOL volcanoes (Gauss et al. 2012)

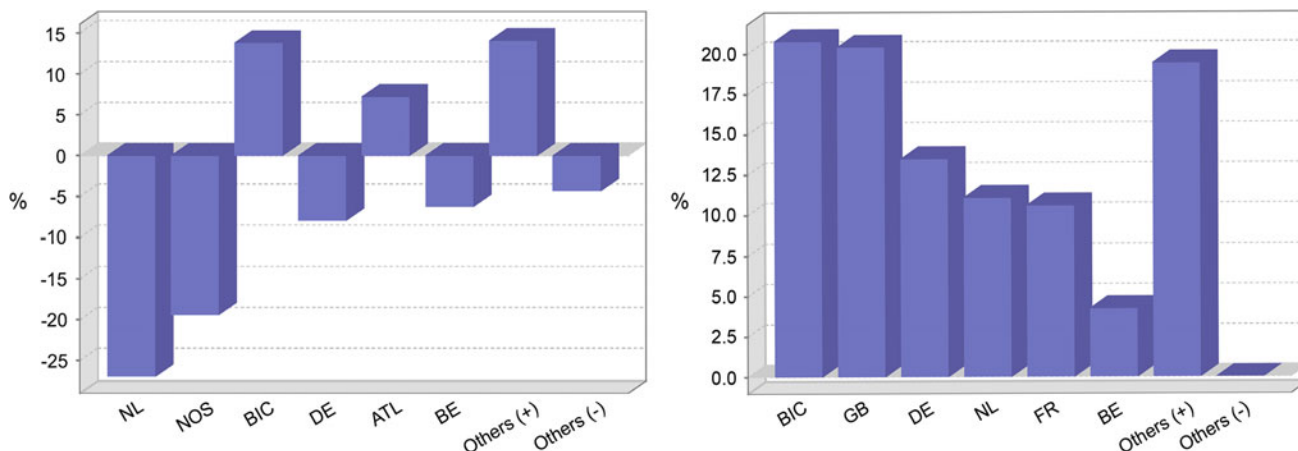


Fig. 16.6 Percentage contribution from individual countries to SOMO35 from NO_x emissions (left) and SOMO35 from nmVOC emissions (right) in the Netherlands. NL Netherlands, NOS North Sea,

DE Germany, GB Great Britain, FR France, ATL remaining Atlantic within model domain, BE Belgium. BIC is boundary and initial concentrations (Gauss et al. 2012)

region. On the other hand, regional VOC and CO emissions result in increased O₃ levels. This is illustrated for The Netherlands in Fig. 16.6 (see EMEP country reports for further examples³).

The negative contribution from NO_x from several countries is caused by titration. The contribution from BIC (boundary and initial concentration) is calculated by perturbing lateral boundary (and initial) concentrations separately for NO_x and nmVOC in the EMEP model.

The contributions from different domestic emission sectors to model-calculated PM_{2.5} concentrations in European countries are shown in Fig. 16.7. The countries are sorted according to the model-calculated contribution from Sector 1 (combustion in energy and transformation industries). The countries in the North Sea region are all to the right in the figure, with relatively small contributions from industry (Sectors 1–3; see figure caption for definitions).

These countries are characterised by a relatively large

share of the emissions from agriculture and transportation (road and shipping). In the North Sea countries, shipping represents a significant share of the transport-related impacts. Studying the external costs from all international ship traffic in relation to the other sources, Brandt et al. (2013a) estimated that ship traffic accounted for 7 % of the total health effects in Europe due to air pollution in 2000. The corresponding value for Denmark, which is surrounded by heavy ship traffic, is 18 %. In Denmark the relative contribution from international ship traffic is comparable to the contributions from road traffic or the domestic use of wood stoves (Brandt et al. 2013b).

The North Sea region is also affected by transport of air pollutants from other continents. Figure 16.8 shows the model-calculated effects on surface O₃ from intercontinental transport due to 20 % reductions in anthropogenic emissions in North America, East Asia and South Asia. As an average for the European continent the calculated contribution to surface O₃ from other continents is about half of the total European contribution (HTAP 2010). The intercontinental contributions show large seasonal (see Fig. 16.8) and geographic variability. Brandt et al. (2012) calculated the effects

³http://emep.int/mscw/index_mscw.html.

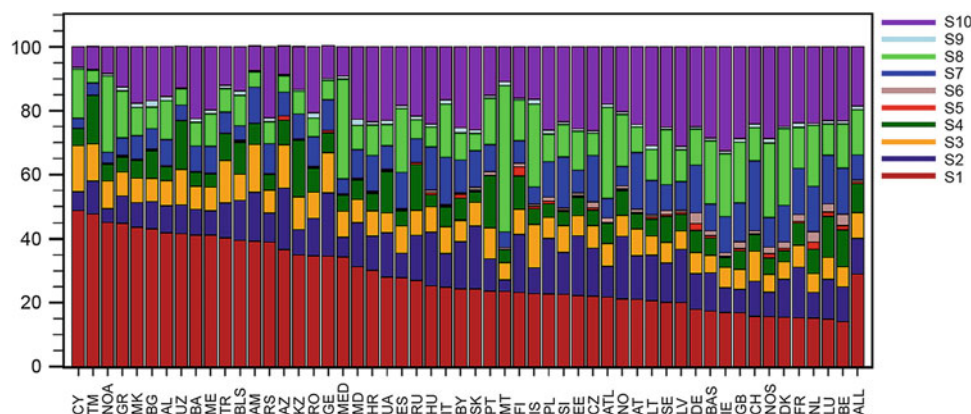


Fig. 16.7 Percentage contribution from individual sectors to $PM_{2.5}$ concentration in European countries: *S1* combustion in energy and transformation industries, *S2* non-industrial combustion plants, *S3* combustion in manufacturing industry, *S4* production processes, *S5*

extraction and distribution of fossil fuels and geothermal energy, *S6* solvent and other product use, *S7* road transport, *S8* other mobile sources and machinery (including shipping), *S9* waste treatment and disposal, *S10* agriculture

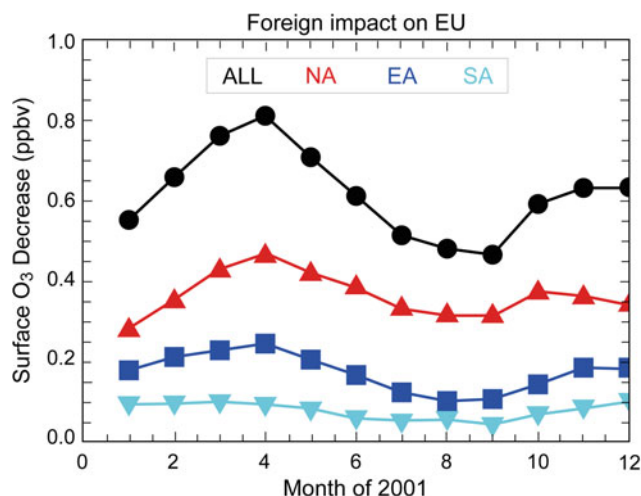


Fig. 16.8 Changes in European surface O_3 levels from 20 % reductions of anthropogenic emissions in North America (NA), East Asia (EA) and South Asia (SA). ALL is the combined effects of the contribution from the three foreign regions (adapted from HTAP 2010, see original report for more details)

of North American emissions on Europe using a tagging method. Their results are at the lower end of the HTAP estimates. The HTAP and Brandt et al. (2012) calculations are for different meteorological years. Located close to the western continental rim, the intercontinental contribution to the North Sea region is higher than the European average as shown by Jonson et al. (2006).

Particulate matter has a short residence time in the atmosphere, and as a result the intercontinental contribution to Europe is in general low. Calculating the ratio of the effect of other (than Europe) source regions to the effect of all source regions (including Europe), indicates that about 5 % of the PM surface concentrations in Europe can be attributed to intercontinental transport (HTAP 2010). However, the

temporal variability is large, and the contribution can be significant for specific episodes.

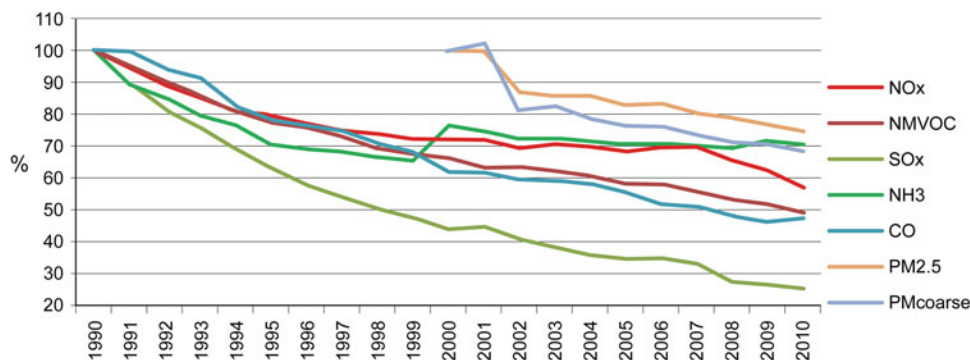
16.3 Recent Change

16.3.1 Emissions

For most air pollutants emission totals reached a maximum in the late 1980s or early 1990s. Since then emissions of air-polluting substances have decreased substantially in most European countries. For parties to the Gothenburg Protocol⁴ emission ceilings are set for 2010 for four pollutants: sulphur, NO_x , VOCs and NH_3 . For EU Member States these emission ceilings are largely integrated within EU legislation. This is illustrated in Fig. 16.9, showing the evolution of emissions from countries within the EMEP domain (i.e. Europe, large parts of the North Atlantic and the polar basin and parts of North Africa; see www.EMEP.int for definition) from 1990 to 2010. Emissions of PM have only been reported since 2000. There are large differences in trends between individual countries, with some countries even increasing their emissions for one or more species. Parties to the Gothenburg Protocol whose emissions have a more severe impact, and whose emissions are relatively cheap to reduce are obliged to make the largest cuts in emissions. The countries around the North Sea are among those that have had to make the greatest cuts in emissions. Uncertainty in the emission trends is significant. The large drop in PM emissions from 2001 to 2002 may reflect incomplete reporting prior to 2002.

⁴www.unece.org/env/lrtap/multi_h1.html.

Fig. 16.9 Emission trends 1990–2010 (2000–2010 for PM_{2.5} and PM_{coarse}) (Fagerli et al. 2012)



The North Sea region is strongly influenced by emissions from shipping. Traditionally, emissions from shipping have been largely unregulated. However, recent policy decisions through the International Maritime Organization (IMO MARPOL Annex VI SO_x Emission Control Area SECA requirements) and the EU (Sulphur Directive) affect ship emissions of both SO_x and PM in the region. The former restricts the marine fuel sulphur content in SECAs to 1.0 % as of 1 July 2010 and 0.1 % from 2015 whereas the latter requires ships to use 0.1 % sulphur fuel in harbour areas from 1 January 2010. Prior to 2010 the maximum allowed sulphur content in SECAs was 1.5 % as opposed to the global fleet average of about 2.4 %. Emissions of SO_x, NO_x, VOC, CO and PM from international shipping, from 1990 to present, are listed in appendix B in EMEP (2013).

16.3.2 Air Pollution

There are no PM₁₀ or PM_{2.5} measurements extending over decades. It is only for the past decade that enough data exist to derive trends. The monitoring network for PM_{2.5} is sparser than for PM₁₀ giving larger uncertainties for the reported PM_{2.5} change. Over the period 2000–2009, Tørseth et al. (2012) found a decrease in PM₁₀ and PM_{2.5} concentration at about half of the European rural sites in the EMEP network. Average reductions in the annual means were 18 % (PM₁₀) and 27 % (PM_{2.5}). None of the stations in the network showed an increasing trend. Similar results were found by Barmapadimos et al. (2012) using selected EMEP stations corrected for meteorological variability. The trends roughly correspond to reported reductions in emissions of primary PM and precursors for secondary PM (Fig. 16.9). Figure 16.10 shows the change in PM₁₀ for the past decade as reported by the European Environment Agency (EEA 2012). These data also include stations in urban surroundings and near roads with heavy traffic. Most of the stations registering a trend showed a decrease, with only 2 % of stations recording an increase. For countries adjacent to the North Sea, moderate decreases are found at most stations although

some show no significant trend. There is also a reduction in the number of exceedances of the EU PM₁₀ daily limit value for most North Sea countries (EEA 2012).

Devasthale et al. (2006) used satellite measurements complemented by station data to investigate trends in air polluting particles and focused on the English Channel and the top three polluting harbours in Europe. For the period 1997–2002 they found increasing particle concentrations over harbours and coastal areas and decreasing concentrations over land areas. The different evolution is attributed to decreased emissions from land-based sources and increased emissions from shipping. Jonson et al. (2015) and Brandt et al. (2013a) modelled the effects of Baltic Sea and North Sea ship emissions in 2009 and 2011 (before and after the reductions in the sulphur content of marine fuels from 1.5 to 1 % from 1 July 2010). The calculations indicate clear improvements in PM concentration. These are however slightly offset by increasing NO_x emissions, affecting nitrate particle formation. This is particularly the case in and around major North Sea ports owing to partial economic recovery after the financial crisis.

Ozone is strongly coupled to meteorological variability both in terms of regional photochemical production and loss, and the contribution from intercontinental transport. Trends are therefore difficult to detect and long time series are needed. In some regions, lack of long-term data makes trend analysis impossible. Reductions in the highest O₃ values are found (Fig. 16.11) in England, Benelux and Germany (EEA 2009, 2012; Tørseth et al. 2012) for the period 1990–2010, but mainly the 1990s. The frequency of high values has also decreased, especially in the Netherlands, England and Ireland (Tørseth et al. 2012).

Studies report that despite a relatively large decline in anthropogenic emissions in Europe (Fig. 16.9) a corresponding reduction in O₃ concentration is not observed (Jonson et al. 2006; EEA 2009, 2012; Tørseth et al. 2012; Wilson et al. 2012). Models also generally struggle to reproduce some of the observed trends (Solberg et al. 2005; Jonson et al. 2006; Colette et al. 2011; Wilson et al. 2012; Parrish et al. 2014). Likely reasons include increased

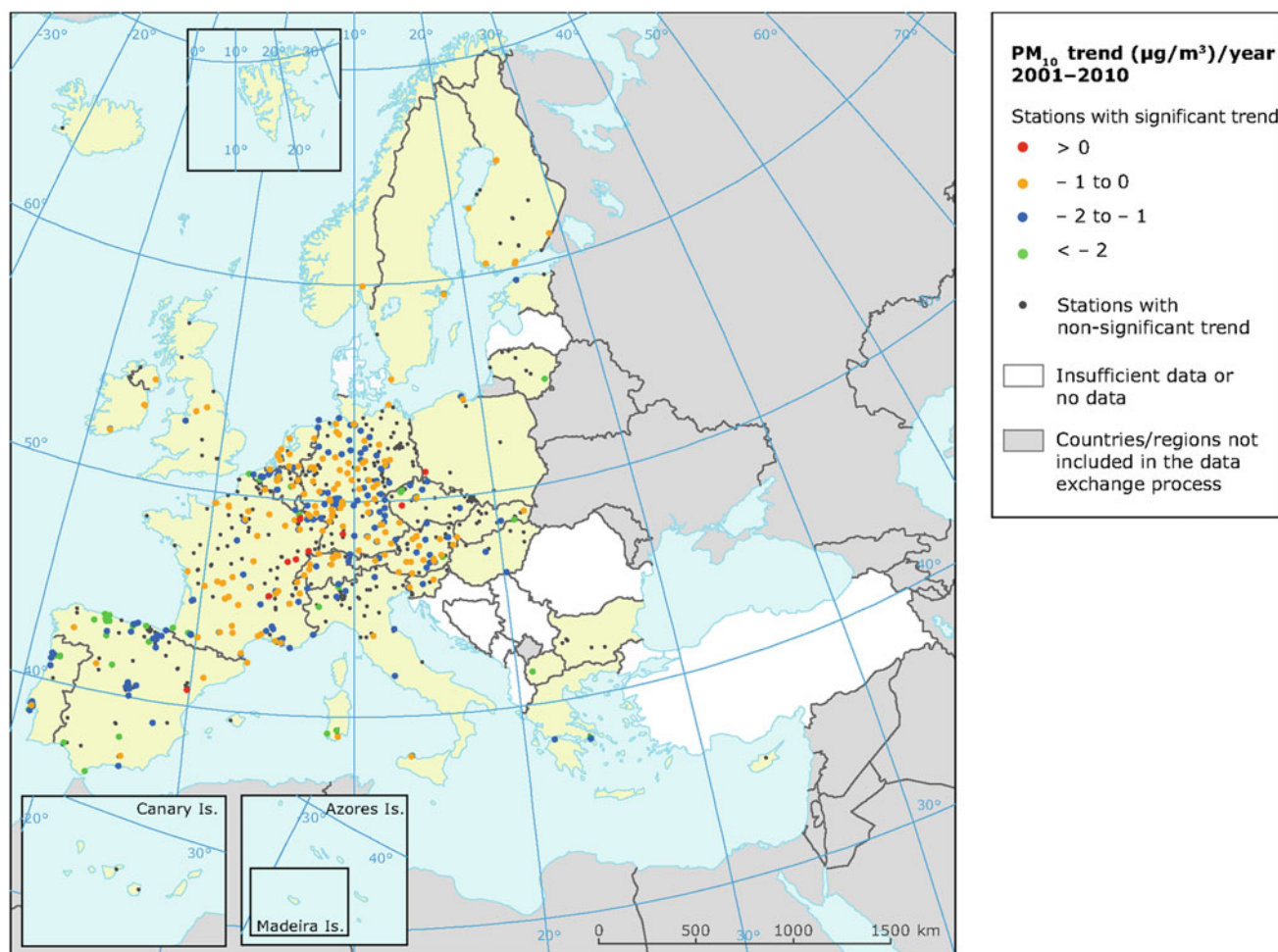


Fig. 16.10 Average annual change in surface PM₁₀ concentration for the period 2001–2010 (EEA 2012). Only stations with a statistically significant trend are shown

background (hemispheric) O₃ level—observational evidence suggests that the increase in background O₃ roughly doubled from 1950 to 2000 and then levelled off (Logan et al. 2012; Derwent et al. 2013; Oltmans et al. 2013; Parrish et al. 2013). Other possibilities are limitations in the understanding of photochemistry, coarse model resolution, uncertainties related to anthropogenic emission estimates, variation in poorly constrained natural emissions, and the small number of measurement sites with long-term data sets.

16.3.3 Contribution from Climate Change

Climate change influences air pollution levels through a number of factors (see HTAP 2010), including changes in temperature, solar radiation, humidity, precipitation, atmospheric transport and biogenic emissions. Using a model to compare current conditions (1990–2009) against a baseline period (1961–1990) Orru et al. (2013) found the largest

climate-driven increase in O₃-related mortality and hospitalisations to have occurred in Ireland, the UK, the Netherlands and Belgium where increases of up to 5 % are estimated. A decrease is estimated for the northernmost European countries. Hedegaard et al. (2012) compared the 1990s with the 1890s and found climate-driven decreases in surface O₃ in the North Sea region. However, the decrease is not statistically significant over the region as a whole.

Using an ensemble of global models, Silva et al. (2013) found the average number of premature annual deaths attributable to past (1850–2000) climate change in Europe to be 954 for O₃ (respiratory) and 11,900 for PM_{2.5}. But the magnitude and even the sign of the values varies between models. In a global study with one model, Fang et al. (2013) found a climate-driven contribution to cardiopulmonary and lung cancer mortality associated with industrial PM_{2.5} since 1860 of up to 14 % over Europe. Ozone was responsible for a small contribution. The calculation does not include the climate-driven effect on emissions of biogenic hydrocarbons.

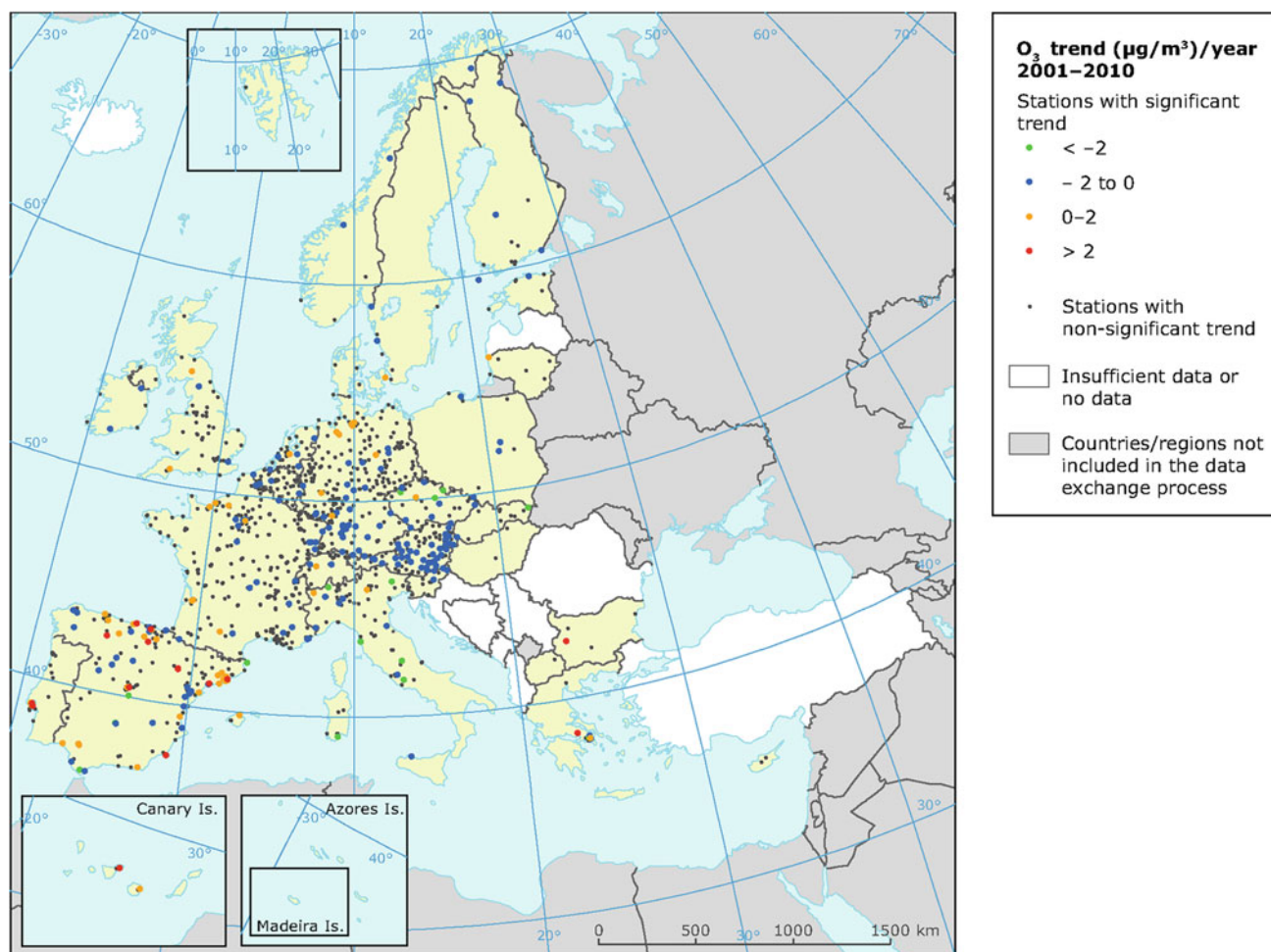


Fig. 16.11 Change in annual mean maximum daily 8-hour ozone (O₃) concentration in the period 2001–2010 (EEA 2012). Only stations (urban/suburban/rural) with a statistically significant trend are shown

Recent heat waves have been linked to climate change (e.g. Stott et al. 2004). Whether these anomalies are exceptional or a signal of changes in climate is still under debate. Summer 2003 was one of the hottest in the history of Western Europe, with surface temperature exceeding the average surface temperature reported for 1901–1995 by 2.4 °C. In fact, this summer was likely to have been warmer than any other back to 1500.

Fischer et al. (2004) estimated that almost half of the excess deaths in the Netherlands during the 2003 heat wave were due to increased air pollution (O₃ and PM₁₀). Stedman (2004) estimated that the same air pollutants were responsible for 21–38 % of excess deaths in the UK. Doherty et al. (2009) found the overall number of deaths attributable to O₃ in England and Wales to be slightly greater than that attributable to heat for the 2003, 2005 and 2006 summers. Several studies have described the high pollution levels observed in 2003 (Vautard et al. 2005; Solberg et al. 2008; Tressol et al. 2008).

The high temperatures during the 2003 heat wave influenced summer O₃ because of its link with high solar radiation, stagnation of the air masses and thermal decomposition of peroxyacetyl nitrate (PAN). Availability of solar radiation favours photolysis yielding radical formation with subsequent involvement in O₃ production. Stagnation of air masses allows the accumulation of pollutants in the planetary boundary layer (PBL) and in the residual layer during the night. Increased temperatures and solar radiation favoured biogenic emissions of isoprene with a potential for enhanced O₃ chemistry in the PBL. High temperature and a spring to summer precipitation deficit reduced dry deposition of O₃. The high temperatures and exceptional drought led to extensive forest fires on the Iberian Peninsula which contributed to the peak in ground level O₃ observed in western central Europe in August (Solberg et al. 2008; Tressol et al. 2008).

16.4 Future Impacts

Future air quality will be affected both by changes in air pollutant emissions and by changes in climate. A large span in emission scenarios and degree of detail in climate simulations are used in different studies. As a result, the studies referred to in the following sections are not directly comparable.

16.4.1 Emission Scenarios

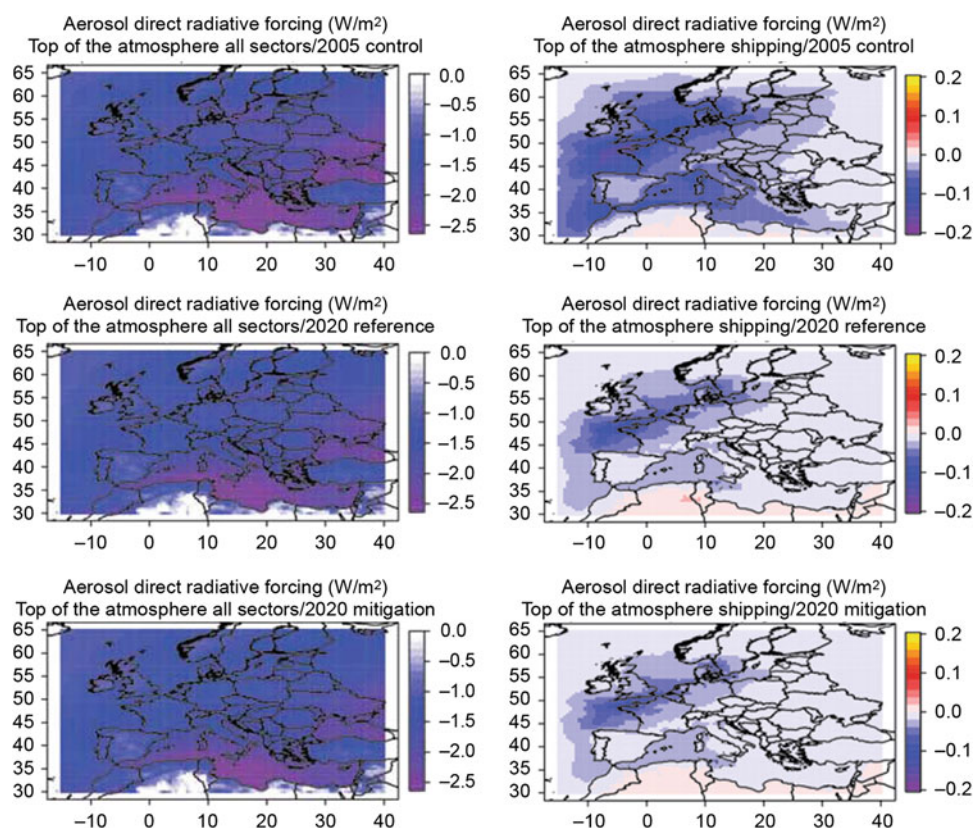
Emissions of air pollutants have significantly reduced in recent decades (see Fig. 16.9). Even though economic activity in Europe is expected to increase, air pollutant emissions from all land-based sectors and regions in Europe are still expected to decline following current legislation. For example, emissions from the transport sector are expected to decrease owing to the penetration of vehicles with stricter emissions standards (Euro 5 and Euro 6), and the expected transition to renewable energy in Europe should drive a decline in emissions from the energy sector. After five years of negotiation, a revised Gothenburg Protocol was finalised in May 2012. The revised protocol specifies emission reduction commitments from base 2005 to 2020. It has also been extended to cover PM_{2.5}. Most states decided only to

accept emission reduction obligations for 2020 that are even less ambitious than—or at best largely in line with—business-as-usual, that is, reductions expected to be achieved anyway solely by implementing existing legislation.

Overall, EU Member States' commitments to the revised protocol mean that from 2005 to 2020 they shall jointly cut their emissions by 59 % (SO₂), 42 % (NO_x), 6 % (NH₃), 28 % (VOCs) and 22 % (PM_{2.5}). According to the IMO MARPOL regulations the maximum sulphur content allowed in marine fuels will be reduced to 0.1 % from 2015 in SECAs (Sulphur Emission Control Areas). Both the North Sea and the Baltic Sea are accepted as SECAs. Further measures may also be implemented for NO_x in these seas. This may impose a shift from extensive use of heavy fuel oil to marine distillates, or a switch to liquefied natural gas (LNG), or using heavy fuel oil in combination with scrubber technology.

In sea areas outside SECAs, sulphur emissions have continued to rise and these emissions also affect the North Sea area. From 2020, the sulphur content in marine fuels outside SECAs should be reduced to 0.5 % globally, but depending on the outcome of a review to be concluded in 2018 as to the availability of the required fuel oil, this date could be deferred to 2025. However, the EU Sulphur Directive obliges ship owners to use 0.5 % fuel in

Fig. 16.12 Aerosol-induced direct radiative forcing at the top of the atmosphere in 2005 and for two projections: reference (2020) and mitigation (2020) (*left column*); Contribution attributed to shipping activities (*right column*). The reference scenario includes all current implemented and planned air quality policies. The so-called mitigation scenario in addition includes further climate policies leading to a stabilisation of global warming to not more than 2 °C in 2100. Both scenarios include mitigation measures for shipping that correspond to MARPOL regulations for NO_x and SO₂ (EEA 2013)



non-SECA EU sea areas from 1 January 2020 regardless of the outcome of the IMO review.

Efforts to improve air quality will undoubtedly influence climate. An interesting example is shown in Fig. 16.12. Large direct and indirect aerosol effects lead to a current global net cooling impact from the shipping sector. The aerosol direct and indirect (not shown) effects are likely to be substantially reduced over the North Sea in 2020. Refraining from air pollutant mitigation to favour a (potential) net cooling effect of the shipping sector is however risky from a health and environmental perspective and given large reported uncertainties, especially for the size of the climate impact of shipping aerosols.

To date, there are no NO_x Emission Control Areas (NECAs) in Europe. Hammingh et al. (2012) evaluated the potential impact of establishing a North Sea NECA. This would require new ships to emit 75 % less NO_x, from 2016 onward. A NECA in the North Sea would reduce total premature deaths due to air pollution in the North Sea countries by nearly 1 %, by 2030. This value would approximately double when all ships met the stringent nitrogen standards, a situation expected after 2040. Health benefits would exceed the costs to international shipping on the North Sea by a factor of two.

16.4.2 Impacts on Air Quality

16.4.2.1 Impacts from Emission Changes

Air quality in the North Sea region should improve as a result of expected reductions in emissions. The reductions in emissions should cause a decrease in PM levels. Based on a parameterisation of the HTAP source receptor calculations for the main source regions in the northern hemisphere, Wild et al. (2012) calculated future O₃ trends following the Intergovernmental Panel on Climate Change RCP (representative concentration pathway) emission scenarios. The calculations demonstrated that substantial annual mean surface O₃ reductions can be expected for most RCP scenarios by 2050 over most regions, including Europe. However, as

discussed in Sect. 16.2.2 parts of the North Sea region are characterised by extensive NO_x titration of O₃. Unlike most other regions in Europe, reductions in NO_x emissions here are likely to result in increased levels of surface O₃, at least in the short term (Fig. 16.13). Colette et al. (2012) concluded that air pollution mitigation measures (present in both scenarios in Fig. 16.13) are the main factors leading to the net improvement over much of Europe, but an additional co-benefit of at least 40 % (depending on the indicator) is due to the climate policy. However the climate policy has little impact in the North Sea region (Fig. 16.13).

The total health-related external costs in Europe due to the total air pollution levels from all emission sources in the northern hemisphere are calculated to be EUR 803 billion year⁻¹ for 2000 decreasing to EUR 537 billion year⁻¹ in 2020 (Brandt et al. 2013a). The decrease is due to the general emission reductions in Europe provided that the revised Gothenburg Protocol is implemented and given the regulation of international ship traffic by introducing SECAs in the North Sea and Baltic Sea. For Denmark the external costs are estimated to be EUR 4.54 billion year⁻¹ for 2000, decreasing to EUR 2.53 billion year⁻¹ in 2020.

Using a baseline emission scenario, Amann et al. (2011) calculated that loss in statistical life expectancy attributed to exposure to PM_{2.5} would decline between 2005 and 2020 from 7.4 to 4.4 months in the EU-27. There are significant improvements for the North Sea region (Fig. 16.14). The improvement in mortality due to ground level O₃ is about 35 % in EU Member States (Fig. 16.15) (Amann et al. 2011), with significant improvements in all North Sea countries. With commercially available emission control technologies, European emissions could be further reduced from baseline by 60 % (SO₂), 30 % (NO_x), 65 % (primary PM_{2.5}), and about 35 % (NH₃ and VOC). The measures would cut the loss in statistical life expectancy by 50 % (or another 2.5 months) compared to the baseline case in 2020.

However, the improvements come at a cost. Full implementation of the additional measures would increase costs for air pollution control in Europe in 2020 from EUR₂₀₀₅ 110 billion year⁻¹ to EUR₂₀₀₅ 192 billion year⁻¹, i.e. from

Fig. 16.13 Difference in surface O₃ between 2030 and 2005 for two scenarios: a reference case (left) and a sustainable climate policy case (right) (Colette et al. 2012)

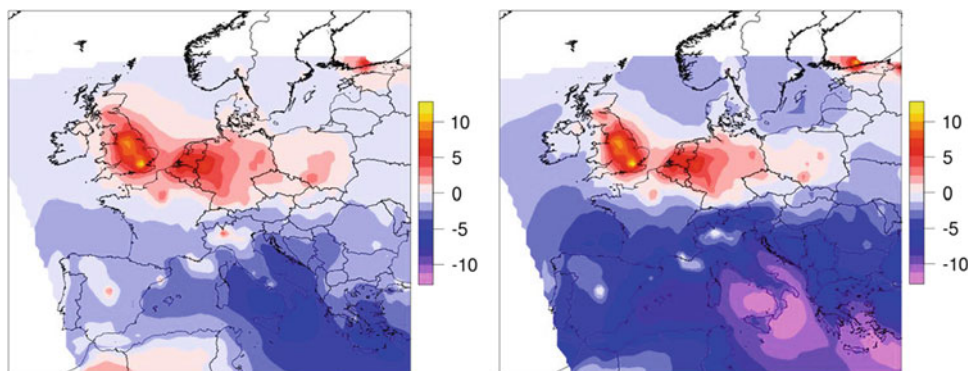


Fig. 16.14 Loss in statistical life expectancy attributable to exposure to $\text{PM}_{2.5}$ from anthropogenic sources. 2005 (*left*) and baseline projection for 2020 (*right*) (Amann et al. 2011)

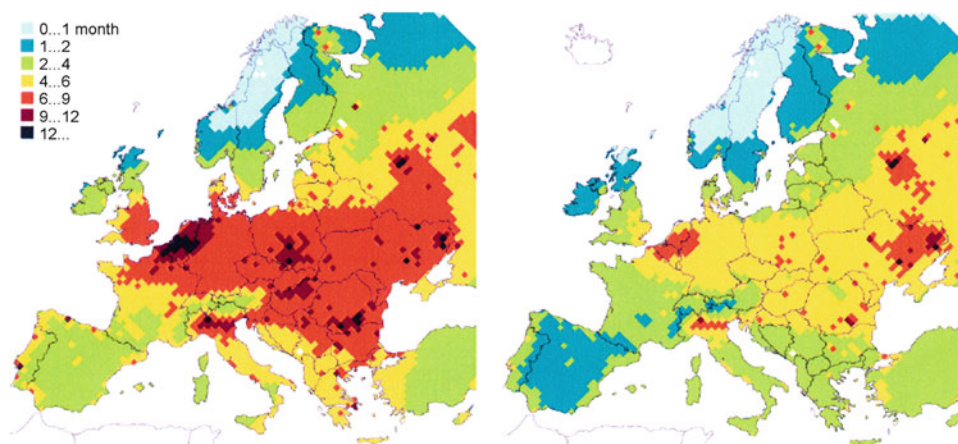
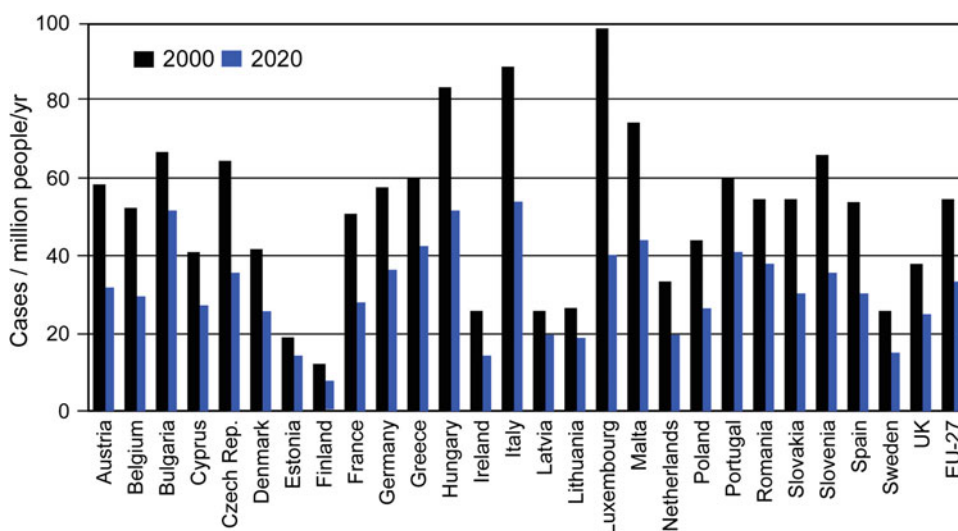


Fig. 16.15 Mortality rates attributable to exposure to ground-level O_3 (Amann et al. 2010)



0.66 % to 1.15 % of the GDP envisaged for 2020. At the same time, some of the measures achieve little environmental improvement. Experience shows that a cost-effectiveness analysis can identify portfolios that realise most of the potential improvements at a fraction of the costs of the total portfolio.

16.4.2.2 Impacts from Climate Change

Climate impacts on air pollution are summarised by HTAP (2010). Future changes in climate are expected to increase local and regional O_3 production and reduce O_3 in down-wind receptor regions. Factors contributing to O_3 increases near emission regions include increased O_3 production due to higher water vapour leading to more abundant hydrogen oxide radicals (HO_x) which leads to increased O_3 production at high NO_x concentrations. Increased global average temperature increases photochemistry rates and decreases net formation of reservoir species for NO_x , leaving more NO_x available over source regions. This promotes local O_3 production. In a warmer climate natural emissions of VOC and

NO_x (biogenic, lightning) are expected to increase. Such increases will depend on uncertain changes in soil moisture, cloud cover, sunlight, and biogenic responses to a more CO_2 -rich atmosphere.

Although there are many factors affecting PM levels, changes in cloud amount and precipitation are the major parameters as wet removal is a major sink for PM. Despite several pathways by which climate change may influence air quality, most model simulations show air pollutant emissions to be the main factor driving change in future air quality, rather than climate or long-range transport (Andersson and Engardt 2010; HTAP 2010; Katragkou et al. 2011; Langner et al. 2012a, b; Coleman et al. 2013; Colette et al. 2013). Hedegaard et al. (2013) found emission changes to be the main driver for PM changes but that climate change is equally important for O_3 in many North Sea countries.

Orru et al. (2013) compared O_3 -related mortality and hospitalisation due to climate change for a baseline period (1961–1990) and the future (2021–2050). Increases in O_3 -related cases are projected to be greatest in Belgium, France,

Spain and Portugal (10–14 %), whereas in most Nordic and Baltic countries there is a projected decrease in O₃-related mortality of the same magnitude. Overall there is an increase of up to 13.7 % in O₃-related mortality in Europe, which corresponds to 0.2 % in all-cause total mortality and respiratory hospitalisations.

The sensitivity of simulated surface O₃ concentration to changes in climate between 2000–2009 and 2040–2049 differs by a factor of two between the models used in a study by Langner et al. (2012a), but the general pattern of change with an increase in southern Europe is similar across different models. Changes in isoprene emissions from deciduous forests vary substantially across models explaining some of the different climate response. In northern Europe, the ensemble mean for mean and daily maximum O₃ concentration both decrease whereas there are no reductions for the higher percentiles indicating that climate impacts on O₃ could be especially important in connection with extreme summer events such as experienced in summer 2003 (see Sect. 16.2.2). Some regional climate modelling studies suggest that conditions such as those of summer 2003 could become more frequent in coming decades (Beniston 2004; Schär et al. 2004).

Colette et al. (2013) and Hedegaard et al. (2013) found that climate change in the North Sea region would constitute a slight benefit for PM_{2.5} concentrations. Other studies show both small increases and decreases of PM within the region (Nyiri et al. 2010; Manders et al. 2012). The spread of precipitation projections in regional climate models constitutes a major challenge in reducing the uncertainty of the climate impact on PM (Manders et al. 2012). Nevertheless, some conclusions can be drawn from the different climate model projections for the North Sea region (Jacob et al. 2014). Winter precipitation is expected to increase over the coming century, while summer precipitation is expected to decrease over much of the region. Heavy precipitation events are expected to occur more often in all seasons.

16.5 Conclusions

Climate and air quality interact in several ways. Emitted air pollutants could directly impact climate or could act as precursors for components acting both as harmful pollutants and climate forcers. On the other hand, air quality is sensitive to climate change since it perturbs the physical and chemical properties of the environment. Climate policies imply energy efficiency and technical measures that change emissions of air pollutants. Reciprocally, air quality mitigation measures affect greenhouse gas emissions. Mitigation optimised for a climate or air quality target in isolation could have synergistic or antagonistic effects.

In the North Sea area, the effects on air quality of emission changes since pre-industrial times are stronger than the effects of climate change over this period. Despite several pathways by which climate change may influence air quality, model simulations show air pollutant emissions to be the main factor driving change in future air quality in the North Sea region, rather than climate. The variation in climate simulations in different studies results in significant uncertainty in the impacts of climate change on air quality. This is particularly the case for PM where the spread of precipitation projections in regional climate models constitutes a major challenge in narrowing the uncertainty.

Climate impacts on air quality are substantial in connection with heat waves, such as that of summer 2003. Some regional climate models suggest that heat waves could become more frequent in the coming decades. If the anticipated reductions in emissions of air pollutants are not achieved, extreme weather events of this type may cause severe problems in the future.

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