Authors:

Xavier Querol (Project coordinator)

A. Karanasiou
F. Amato
C. Vasconcelos
A. Alastuey
M. Viana
T. Moreno
F. Planas
N. Perez
M. Cabañas
R. Bartoli
S. Martinez
M. Sosa

E. Montfort
R. Harrison
K. Eleftheriadis
F. Lucarelli
C. Alves
V.L. Gianelle
C. Colombi
I. Celades
A. Escrig
V. Sanfelix
G. Holman
D. Beddows
M. Harding
L. Diapouli
S. Vratolis
M. Gini
E. Bairaktari
S. Dalaina
V. Galifianakis
S. Nava
G. Calzolai
R. Udisti
S. Vecagli
R. Traversi
M. Severi
S. Borselli
M. Giannoni
C. Pio
T. Nunes
L. Tarelho
M. Duarte
M. Cerqueira
E. Vicente
D. Custódio
H. Pinto
INDEX

1. INTRODUCTION ............................................................................. 3

2. IMPACT OF AIR QUALITY ON CLIMATE CHANGE ..................... 4

3. IMPACT OF CLIMATE CHANGE ON AIR QUALITY .................... 9

4. EU EMISSION REDUCTION TARGETS ........................................ 10

4.1. CLIMATE CHANGE .............................................................. 10
4.2. AIR QUALITY ............................................................... 10

5. EMISSION SOURCES AND TRENDS ........................................ 12

6. MITIGATION MEASURES FOR AIR POLLUTION AND CLIMATE CHANGE 14

6.1. ENERGY EFFICIENCY ........................................................ 17
6.2. FUEL SWITCHING .............................................................. 18
6.3. COMBUSTION TRADE-OFFS AND TECHNOLOGICAL MEASURES 18
6.4. STATIONARY COMBUSTION - TECHNOLOGICAL MEASURES 20
6.5. ROAD TRANSPORT ........................................................... 23
6.6. RESIDENTIAL HEATING .................................................... 27

7. SUMMARY AND CONCLUSIONS ................................................. 28

8. REFERENCES .............................................................................. 31
1. INTRODUCTION

Measures to improve urban air quality and mitigate climate change tend to be considered separately even though many of same pollutants affect both environmental impacts. The emission of the traditional air quality pollutants (AQP$^1$) either directly, or indirectly as a result of atmospheric chemistry, affect the concentrations of several climate pollutants$^2$; while the warming of the climate affects the concentrations of the AQP$s$. Some pollutants, such as ozone ($O_3$) are both an AQP and a greenhouse gas. These interactions are complex with the AQP$s$ both enhancing and mitigating global warming.

A large number of abatement measures are beneficial for mitigating both impacts; however there are some measures that may be beneficial for mitigating climate change but increase emissions of the key urban air pollutants, and vice versa.

This report aims at identifying policy measures that have co-benefits and where there are adverse side effects. This is not always simple to determine due to the different temporal and spatial scales of the impacts of emissions on climate change and public health$^3$.

The timescales for climate change are very long term whilst urban air pollutants are responsible for an estimated 0.5 million premature deaths each year in Europe today (EEA, 2015a). This report focuses on measures that could be introduced in the relative short term, i.e. in the next decade or so.

There are a very large number of potential policy measures. This report concentrates on the main pollutants emitted during the combustion of fossil fuels for electricity, heating and transport that affect both climate change and urban air quality. These are carbon dioxide ($CO_2$), methane ($CH_4$), nitrogen oxides ($NO_x$), sulphur dioxide ($SO_2$), and particulate matter (PM). Emissions of carbon monoxide ($CO$) and non-methane volatile organic compounds (NMVOC$s$) are also greenhouse gases (GHGs) through their role in the formation of ozone ($O_3$). The role of $CH_4$ is complex because it is both a GHG and an important precursor of $O_3$.

It is recognised that emissions from agriculture, forestry, and land use change also contribute to climate change. Indirect land use change can result in the release of carbon when natural

---

$^1$ Typically considered to be carbon monoxide ($CO$); nitrogen oxides ($NO_x$); sulphur dioxide ($SO_2$), particulate matter (PM). Also includes non-methane volatile organic compounds (NMVOC$s$) as an $O_3$ precursor and ammonia ($NH_3$) as a PM precursor.

$^2$ In this report the terms greenhouse gases (GHGs) and climate pollutants are used. GHGs are typically defined by the gases included in the Kyoto Protocol i.e. carbon dioxide ($CO_2$), methane ($CH_4$), nitrous oxide ($N_2O$), sulphur hexafluoride ($SF_6$), hydrofluorocarbons (HFC$s$) and perfluorocarbons (PFC$s$). Nitrogen trifluoride ($NF_3$) was added in 2012 (Doha amendment). The term climate pollutants includes aerosols.

$^3$ Poor air quality has a number of adverse effects including the eutrophication and acidification of ecosystems. The AIRUSE project is on improving urban air quality and therefore in this report we have focused on public health.
vegetation (e.g. forestry) is replaced by less carbon rich vegetation types. Direct emissions from agriculture and forestry also contribute to the formation of secondary organic and inorganic particulate matter (PM) which are AQPs. Mitigating these emissions has not been considered in this report as the AIRUSE project focuses on mitigation measures for AQPs, particularly PM, within urban areas.

The emissions of the other traditional GHGs, the fluorinated gases (also known as f-gases) are associated with specific industrial processes or products. These are also not discussed in this report.

Before discussing the mitigation measures this report provides a brief summary of the interaction between the AQPs and climate pollutants, albeit many are both, largely based on the descriptions provided by the 5th Assessment Report of the Intergovernmental Panel on Climate Change (IPPC) (Stocker et al., 2013). The section on the impacts of climate change on air quality is based on that provided by the Air Quality Expert Group (AQEG, 2007). The evidence of the many linkages between air quality and climate change is growing and both sections of this report have also drawn on the comprehensive and recent review of Fiore et al. (2015).

2. IMPACT OF AIR QUALITY ON CLIMATE CHANGE

The Earth’s long-term, globally averaged equilibrium temperature depends on the balance between the incoming solar energy and the outgoing radiation. Any perturbation to this balance is known as radiative forcing (RF) and can be positive (warming) or negative (cooling). It is generally assessed relative to the year 1750 and is globally integrated.

Recent advances in atmospheric modelling have allowed the combined impact from emissions, chemical transformation, aerosol-cloud interactions, and distribution over the atmosphere to be considered in the assessment of global climate change. However, results between models are not always consistent.

The contribution of GHGs, aerosols and their precursors over the period 1750 to 2011 is shown in Figure 1, taken from the 5th assessment report of the IPCC (Stocker et al., 2013). CO₂ is the largest single contributor to historical RF. It should be noted that the relative importance of some contributors varies markedly depending on whether changes in emissions or concentrations are considered. CH₄ emissions have a much larger forcing (0.97 Wm⁻² from 1750 to 2011) than CH₄ concentration increases (approximately 0.5 Wm⁻²) due to the effects of atmospheric chemistry. This reflects the additional RF due to tropospheric O₃, stratospheric water vapour and CO₂ produced by the oxidation of CH₄.
Figure 1: Change in radiative forcing of climate between 1750 and 2011 (Stocker et al., 2013)

Figure 1 Notes: Radiative forcing (RF) of climate change during the Industrial Era shown by components from 1750-2011. The horizontal bars indicate the overall uncertainty, whilst the vertical bars are for the individual components (vertical bar heights proportional to the relative uncertainty, with a total length equal to the bar width for a ±50% uncertainty). Best estimates for the totals and individual components (from left to right) of the response are given in the right column. Values are RF except for the effective radiative forcing (ERF) due to aerosol-cloud interactions (RFaci) and rapid adjustment associated with the RF due to aerosol-radiation interaction (RFari, Rapid Adjust). Note that the total RF due to aerosol-radiation interaction (-0.35 Wm⁻²) is slightly different from the sum of the RF individual components (-0.33 Wm⁻²). Secondary organic aerosol has not been included since the formation depends on a variety of factors not currently sufficiently quantified. The ER of contrails includes contrail-induced cirrus. Combining ERFraci -0.45 [-1.3 to 0.0] Wm⁻² and rapid adjustment RFari -0.1 [-0.3 to +0.1] Wm⁻² results in an integrated component of adjustment due to aerosols of -0.55 [-1.33 to -0.06] Wm⁻². CFCs = chlorofluorocarbons. HCFCs = hydrochlorofluorocarbons. HFCs = hydrofluorocarbons. PFCs = perfluorocarbons, NMVOCs = non-methane volatile organic compounds, BC = black carbon.

Some pollutants have had either a warming or a cooling effect, while others have a mixed effect, resulting in net warming, or net cooling, at the global scale. Emissions of CO₂, CH₄, nitrous oxide (N₂O), BC, CO, and NMVOC have warmed the atmosphere (+3.97 Wm⁻²). Emissions of SO₂, organic carbon (OC) and mineral dust have contributed to a cooling of the atmosphere (-0.80 Wm⁻²). Emissions of halocarbons have had both a warming and cooling effect, with a net warming, while emissions of NOx and NH₃ have also had both a warming and cooling effect, but the net result is cooling. Aerosol–cloud interactions have resulted in a
cooling of the atmosphere, although the contribution of individual compounds is currently unknown.

Climate change is a long-term process and the impact of individual pollutants depends on their atmospheric lifetime. The effect of emissions of a GHG or PM has typically been assessed through its global warming potential (GWP), which is a measure of its impact on global climate relative to that for CO$_2$ on a given time frame, for example 100 years.

The pollutants affecting urban air quality have short atmospheric lifetimes. These pollutants also have effects on climate change due their role in the formation of secondary PM and O$_3$, and the lifetime of other GHGs. There is now increasing focus on the global temperature change potential (GTP), which is based on the change in global mean surface temperature at a chosen point in time, again relative to that caused by CO$_2$, and thus takes into account the atmospheric lifetimes of different pollutants.

The use of GWP or GTP with different time horizons significantly affect the relative contributions of different sources and pollutants, particularly the short-lived climate forcing agents. For example, using GTP the sectors that have the greatest long-term warming impacts (energy and industry) lead to cooling in the near term (primarily due to SO$_2$ emissions), and thus emissions lead to opposite global mean temperature responses over short- and long-time scales. This can therefore affect the policy emphasis placed on abating short- and long-lived climate forcing agents.

It should be noted that there is greater uncertainty of the role of the short-lived climate pollutants than the long-lives ones, as indicated by the vertical bars in Figure 1. For PM and O$_3$ little is known of pre-industrial concentrations or long-term trends over time, and thus estimates of the RF from O$_3$ and PM rely on the results of models, which often yield in different RFs estimates, even when the same emission inventories are used (Fiore et al, 2015).

The IPCC (Stocker et al., 2013) has shown that near-term climate forcers, such as BC, SO$_2$ and CH$_4$ can have contributions comparable to that of CO$_2$ for short time horizons (of either the same or opposite sign), but their impacts become progressively less for longer time horizons when emissions of CO$_2$ dominate. The energy sector and industry will make the largest contributions to warming over the next 50 to 100 years, but to reduce emissions will require long-term efforts to de-carbonise energy. Reducing the precursors of O$_3$ and PM has a rapid impact on near term global warming.

There are other differences between urban air and climate forcing pollutants. For air quality the concentration of pollutants at the surface is the most important issue as this is where exposure occurs. Conversely, GHGs and aerosols have the greatest impact on climate high in the troposphere, where temperatures are lower, and the radiative effect greatest.

For CO$_2$, a long lived and well mixed GHG, the impact of emissions is independent of where the emissions occur, as concentrations are homogeneous, and the effects occur at the global level. This is very different to urban air quality where there are large variations in concentrations within a city, and the effect on public health is at a local scale.

There are a multitude of interactions between the pollutants in the air. This is illustrated in the following sections on CH$_4$, O$_3$ and PM.
Methane ($\text{CH}_4$)

Emissions of NOx, NMVOC and CO have a direct effect on the lifetime of CH$_4$. These air pollutants compete for hydroxyl radicals (OH), which is the major mechanism for removing CH$_4$ from the atmosphere. Growth in CO and NMVOCs emissions has increased the atmospheric lifetime of CH$_4$, and therefore its abundance. The rise in NOx emissions, however, shortens the CH$_4$ lifetime by increasing the production of OH. Increases in CH$_4$ emissions is thought to have resulted in a longer lifetime (Fiore et al, 2015). Increasing CH$_4$ emissions also have implications for the lifetimes of other GHGs such as hydrochlorofluorocarbons.

The net effect on climate forcing of NOx emissions on CH$_4$ concentrations may be slightly negative for ground level sources and slightly positive for aviation. CO and NMVOC emissions are virtually certain to have caused a positive forcing.

Most estimates of CH$_4$ impacts have not included the impact of CH$_4$ on stratospheric O$_3$ concentrations, which may increase the CH$_4$ RF (Fiore et al., 2015).

Ozone (O$_3$)

NOx, VOCs and CO are precursors of O$_3$. Tropospheric O$_3$ is one of the largest single components of the current radiative forcing of climate, after CO$_2$ and methane (CH$_4$)$^4$. NMVOC and NOx contribute to the formation of high O$_3$ episodes, while the oxidation of CH$_4$ is more important for influencing the long-term background O$_3$ concentrations. The impact of changes to NOx, CO and NMVOC emissions on the tropospheric O$_3$ burden occurs over short time-scales (days to months) whereas the fraction of O$_3$ produced as a result of CH$_4$ oxidation responds to changes on CH$_4$ emissions over a much longer time scale (decade or so). The atmospheric chemistry that produces O$_3$ also results in the oxidation of CO to CO$_2$. Therefore, emissions of NOx, CO and NMVOCs affect the atmospheric lifetimes CH$_4$, O$_3$ and CO$_2$.

The impacts are not globally homogenous. For example, there is a larger effect on RF of NOx and NMVOCs emitted in tropical regions. This is the result of the higher OH concentrations (due to the high radiation and water vapour) than elsewhere, which affects the CH$_4$ lifetime. The O$_3$ formed in this region can be lofted up to the upper troposphere where it is most efficient at trapping terrestrial radiation.

The O$_3$ produced from CH$_4$ is not strongly sensitive to emission location but does depend on atmospheric NOx concentrations, and therefore occurs in more polluted regions (Fiore et al, 2015). Observations indicate that O$_3$ concentrations have increased at many remote locations in Europe through the 1990s, but remained stable in the 2000s.

CO and NMVOC emissions exert a net positive RF by increasing tropospheric O$_3$ and CH$_4$, with a larger net impact from CO. Reducing NOx emissions lowers O$_3$ and nitrate concentrations but increases CH$_4$ long term, while CH$_4$ raises tropospheric O$_3$ and stratospheric water vapour.

---

$^4$ The RF of ozone is not explicitly shown in Figure 1 because it is a secondary pollutant, and it has been divided between its precursors.
Components of Particulate Matter (PM)

The effect of PM in the climate is very complex as particles can cause both positive and negative RF depending on their composition. BC, a product of incomplete combustion mainly emitted from the burning of solid fuels, including biomass for residential heating and from diesel vehicles in the European Union (EEA, 2015a), is a strong warming agent (+0.6 Wm⁻²). According to some estimates it is be more important than CH₄ but is subject to large uncertainties (see figure 1). NOₓ, SO₂, and NH₃ are all precursors of secondary inorganic aerosols, which have a negative radiative forcing effect (-0.62 Wm⁻²), as does OC particles (-0.2 Wm⁻²), and mineral dust (-0.1 Wm⁻²). Brown carbon (the light absorbing component of OC particles) absorbs radiation and warms the surface. BC and OC particles are often co-emitted and may, depending on the relative contributions result in a net cooling from some sources (Fiore et al., 2015).

PM may alter the climate by modifying cloud microphysical properties including making clouds brighter and more reflective, and by prolong the lifetime of a cloud before precipitation. These effects indirectly increase the scattering of solar radiation to space by clouds and change precipitation patterns. Aerosol-cloud interactions can influence the character of individual storms but, according to the IPPC there is no clear evidence for a systematic aerosol effect on storms or precipitation intensity (Stocker et al, 2013).

Estimating the RF of PM is challenging and is depending on the use of models. Very long term reliable observations are not available, and preindustrial concentrations can only be inferred from ice or sediment cores. There are also large sources of natural PM including sea salt, mineral dust and organic particles of biogenic origin. The distribution of PM, both natural and anthropogenic, in the atmosphere is not homogeneous and therefore the effects on climate vary regionally.

Secondary sulphate particles (-0.41 Wm⁻²) contributed most of the cooling of anthropogenic PM. Concentrations are greatest in the mid-latitudes of the northern hemisphere, and may have caused a southward shift in the jet stream. Global emissions of SO₂ have declined significantly in Europe and north America over recent decades and are back to the levels last seen in the 1960s. It may be that the rapid warming seen in the 1980s and 1990s reflect the decrease in SO₂ emissions in those decades.

As SO₂ emissions decline the role of nitrate particles may become more important. The formation of these particles required NH₃; with less sulphate in the atmosphere, the formation of ammonium nitrate may increase. Nitrate also has a cooling effect (-15 Wm⁻²).

BC and BrC have a direct impact on climate change by deposition onto snow and ice, which decreases surface albedo and increase absorption of incoming sunlight. If this results in increased melting and exposure of darker surfaces there is an additional radiative forcing.

Total aerosol effective radiative forcing is generally negative, but is positive over the Arctic and other snow-covered regions. According to the IPPC (Stocker et al., 2013) there is high confidence that aerosols have offset a substantial proportion of global warming. This has important implications for pollution abatement, as it implies that a reduction in secondary PM would increase global warming. Currently this cooling affect is not been well quantified. This has important implications because if the PM contribution to cooling is over-estimated the
warming will also have been over-estimate and visa versa. It therefore lessens confidence in future climate projections.

Air pollutants may also have significant effects on concentrations of \( \text{CO}_2 \) and \( \text{CH}_4 \) through their impacts on ecosystem sources and sinks. These include the effects of nitrogen (N) deposition increasing plant growth and thus carbon uptake, effects of sulphate deposition reducing \( \text{CH}_4 \) emissions from major natural sources, and effects of \( \text{O}_3 \) in reducing plant growth and carbon uptake.

### 3. IMPACT OF CLIMATE CHANGE ON AIR QUALITY

Climate change is expected to adversely affect air quality by changing weather patterns. Quantifying these impacts is difficult because of the year-to-year natural weather variability. However, it is considered likely that climate change will affect air quality through alternations to mixing, dilution, formation and removal processes, and affects both anthropogenic and natural emissions. If regional meteorology changes leads to a deterioration in air quality this would be a ‘climate penalty’; if on the other hand it reduces concentrations this would be a ‘climate benefit’ (Fiore et al., 2015).

PM, for example, is removed from the atmosphere by precipitation and increased frequency and intensity of rainfall will reduce PM concentrations. The same conditions would reduce the mineral dust component, as rain is effective at suppressing dust emissions. Conversely, reduced rainfall would increase concentrations.

A reduction in periods of stagnant air, with increased wind speeds, would increase mixing, and dilution of emissions, and improve air quality; while conversely an increase in days with low wind speeds and a shallow mixing depth would lead to a deterioration in air quality.

Higher maximum temperatures and more frequent of summer heat waves will alter the atmospheric chemistry associated with \( \text{O}_3 \) and secondary aerosol formation, and increase the frequency of summer pollution episodes. The greatest effect is likely to be on concentrations of water vapour, which will lead to decreases in \( \text{O}_3 \) in remote areas but increases in more polluted regions where there are higher NOx concentrations. There could also be an increase in the flux of \( \text{O}_3 \) from the stratosphere to the troposphere, further increasing concentrations at the surface (Fiore at al., 2015).

Increases in temperature will also lead to increases in emissions of biogenic compounds such as isoprene, and hence the formation of \( \text{O}_3 \), secondary aerosols and other photochemical pollutants. Since emissions of VOCs vary between tree species, tree-planting schemes aimed at energy production or carbon sequestration should take account of the potential for emissions and their impact on the formation of \( \text{O}_3 \) and secondary organic aerosols (QUARG, 2007).

In addition, hot, dry summers reduce the uptake of \( \text{O}_3 \) through the stomata of plants. This decreases the damage to the plant but increases the concentration of ground level \( \text{O}_3 \) (QUARG, 2007). On the other hand it has been suggested that if the increased concentrations offset the \( \text{O}_3 \) uptake effect, and there is more \( \text{O}_3 \) damage to plants there could be a knock-on effect on the carbon cycle and atmospheric \( \text{CO}_2 \) (Fiore et al, 2015).
Finally, higher summertime temperatures have the potential for more frequent drought and more wildfires increasing emissions of PM, NOx and other air pollutants. Drought will also increase mineral dust emissions, which could worsen PM pollution.

4. EU EMISSION REDUCTION TARGETS

4.1. Climate Change

The EU’s initial climate change strategy was to achieve the so called 20/20/20 targets by 2020:

- Reduce GHG emissions by 20% from 1990 levels
- Increase the share of renewable energy to 20%
- Improve energy efficiency by 20% from 1995 levels

The GHG and renewable targets are expected to be achieved, but the energy efficiency target may be missed by a few percent, unless there is better implementation and enforcement of the current policies and measures in some Member States including Germany and Sweden. Seventeen Member States, including Greece, Italy, Portugal, and Spain are considered to be on track towards their 2020 targets (EC, 2014a).

The Commission has proposed a series of new targets to be achieved by 2030:

- 40% reduction in GHG emissions from 1990 levels
- Increase the share of renewable energy to 27%
- Improve energy efficiency by 30% from 1995 levels

The Commission expects all sectors of the economy to contribute to these targets.

EU leaders have also endorsed the long term objective of reducing the EU’s GHG emissions by between 80% and 95% by 2050, compared to 1990 levels (EEA, 2015b). In the long-term a shift from coal and gas to nuclear power and renewables will be required to meet these targets, which would also have significant air quality benefits.

The G7 nations, which include France, Italy Germany, and UK, are committed to decarbonising their economies this century (The White House, 2015).

4.2. Air Quality

National emission limits are set for the EU and individual Member States under the Gothenburg Protocol to the Convention for Long-range Transboundary Air Pollution (CLRTAP). These limits were implemented in EU legislation by the National Emissions Ceiling Directive (NECD; Directive 2001/81/EC). This Directive limits the total emissions in 2010 of the four pollutants responsible for acidification, eutrophication and ground-level O₃ (SO₂, NOₓ, VOCs and NH₃). The NECD leaves it largely to the Member States to decide which measures, in
addition to EU legislation for specific source categories, to take in order to comply. The protocol includes emission limits for certain sources.

The emission reduction limits for the European Community for 2010 from 1990 levels were as follows:

- SO2 – 75%
- NOx – 49%
- NH3 – 15%
- VOCs – 57%

In 2012 the Gothenburg Protocol was revised. It introduced new national emission reduction commitments applicable from 2020 for SO2, NOx, NMVOC, NH3, and fine PM (PM2.5), and new more stringent emission limits.

The EU is committed to reducing total emissions by 2020 from 2005 levels as follows:

- SO2 – 59%
- NOx – 42%
- NH3 – 6%
- VOCs – 28%
- PM2.5 – 22%

Whilst the original aim of the CLRTAP was to limit transboundary air pollution and its effects on eco-systems, the amended Gothenburg Protocol also addresses human health. These emission reductions targets will be implemented into EU legislation by a revised NECD. This revised Directive will also include additional national limits for 2030, which are currently being negotiated.

The proposed 2030 EU emission limits, from 2005 levels, were as follows:

- SO2 – 81%
- NOx – 69%
- NH3 – 27%
- NMVOCs – 59%
- PM2.5 – 51%
- CH4 – 33%

The revised NECD is due to be agreed at the end of 2015.

Other than the revised NECD the only proposed new legislation in the European Commission’s Clean Air Policy Packag’ (European Commission, 2013) is a Directive to reduce emissions from medium-sized combustion installations. The package has been criticised as un-ambitious. Amann, 2015 has estimated that the NECD targets could be nearly meet without any new emission control measures. Harrison et al. (2014) suggested that more ambitious emissions
limits should be accompanied by lower ambient air quality limit values to improve air quality. There are many sources of AQPs that are difficult to quantify reliably, including NOx and PM emissions from road transport, and therefore relying solely on emission targets is likely to result in some important sources remaining uncontrolled. In addition, as the development of emission inventories is not an exact science, there may be a temptation to alter the emission factors used to help meet the targets.

5. EMISSION SOURCES AND TRENDS

Table 1 shows the sources of GHG emissions in the EU in 1990 and 2013 as estimated by the member states and collated by the European Environment Agency (EEA, 2015c). All sectors have reduced emissions over this period except road transport and refrigeration/air conditioning. For road transport there is a significant difference between emissions of CO₂ (gasoline and diesel cars) and NOx (diesel vehicles) measured in official tests and real world driving conditions and any inventory that uses the former data is likely to under estimate emissions from road transport.

Emissions have also increased from international aviation, international maritime transport and international bunkers (included within energy use in the table). Energy and national transport is responsible for 77% of the CO₂ equivalent emissions, while agriculture is responsible for 11%. This is mainly due to N₂O, a powerful GHG, emitted as a result of fertilizer use, the grazing of animals and use of animal manure.

Emissions of GHGs have declined significantly since 1990 and in 2013 the EU’s emissions (CO₂e) were 21% below 1990 levels (EEA, 2015d). The 20% emission reduction target was achieved some six years before the target date of 2020. However, there has not been a reduction in all member states and a number of EU countries, most notably Spain and Portugal, have increased GHG emissions⁵ over this period, while most northern and central European countries have reduced emissions. Germany, the largest emitter in the EU, has reduced emissions by 24% over the same period (EEA, 2015c). The reduction in many central and eastern EU countries has also been very significant due to structural changes to their energy industry since joining the EU. Over the same time the EU economy grew by approximately 45%. EU countries produced more output and/or value (GDP) with less GHG emissions (EEA, 2015c).

⁵ Excluding land use, land use change and forestry.
Table 1: Emissions of Greenhouse Gases in the EU-28 by Sector (EEA, 2015d)

<table>
<thead>
<tr>
<th>Sector</th>
<th>Emissions CO₂ equivalent (million tonnes)</th>
<th>Change in emissions (% 1990 to 2013)</th>
<th>Contribution in 2013 (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>1990</td>
<td>2012</td>
<td></td>
</tr>
<tr>
<td>Energy (excluding road transport)</td>
<td>3633</td>
<td>2685</td>
<td>-26</td>
</tr>
<tr>
<td>Road transport</td>
<td>723</td>
<td>839</td>
<td>+16</td>
</tr>
<tr>
<td>Industrial processes and product use</td>
<td>511</td>
<td>360</td>
<td>-30</td>
</tr>
<tr>
<td>Agriculture</td>
<td>569</td>
<td>441</td>
<td>-22</td>
</tr>
<tr>
<td>Waste</td>
<td>244</td>
<td>152</td>
<td>-38</td>
</tr>
<tr>
<td>Total (without LULUCF)</td>
<td>5,689</td>
<td>4,481</td>
<td>-21</td>
</tr>
<tr>
<td>Land use, land-use change and forestry</td>
<td>-260</td>
<td>-318</td>
<td>+22</td>
</tr>
<tr>
<td>(LULUCF)</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Total (with net CO₂ emissions/removals)</td>
<td>5,429</td>
<td>4,163</td>
<td>-23</td>
</tr>
</tbody>
</table>

Table 2 shows the sources of AQPs in the EU-28 in 2013. The emission categories are slightly different to those in Table 1 due to the different reporting requirements of the UN conventions on transboundary air pollution and climate change. Energy production and use remains a major source of CO, SO₂, and NOx while virtually all the NH₃ emissions are from agriculture. The major sources of NMVOCs are industrial processes and product use.

This table shows the average estimated emissions across the EU. In reality there is significant spatial variation in the contributions of different sources. In terms of human exposure road traffic emissions of PM, NOx and CO are especially important because they occur close to ground level within cities. Most industrial emissions occur at some height, often at some distance from heavily populated areas and therefore may have less direct impact on public health.

Table 2: Emissions of Air Pollutants in the EU-28 by sector 2013 (EEA, 2015e)

<table>
<thead>
<tr>
<th>Sector</th>
<th>NH₃</th>
<th>CO</th>
<th>NMVOC</th>
<th>NOx</th>
<th>SO₂</th>
<th>PM₂.₅</th>
</tr>
</thead>
<tbody>
<tr>
<td>Energy production &amp; distribution</td>
<td>0</td>
<td>3</td>
<td>9</td>
<td>21</td>
<td>56</td>
<td>5</td>
</tr>
<tr>
<td>Energy use in industry</td>
<td>0</td>
<td>12</td>
<td>2</td>
<td>12</td>
<td>19</td>
<td>6</td>
</tr>
<tr>
<td>Non-road transport</td>
<td>0</td>
<td>2</td>
<td>2</td>
<td>7</td>
<td>2</td>
<td>2</td>
</tr>
<tr>
<td>Road transport</td>
<td>2</td>
<td>22</td>
<td>12</td>
<td>39</td>
<td>0</td>
<td>13</td>
</tr>
<tr>
<td>Commercial, institutional &amp; households</td>
<td>1</td>
<td>47</td>
<td>16</td>
<td>15</td>
<td>16</td>
<td>58</td>
</tr>
<tr>
<td>Industrial processes and product use</td>
<td>1</td>
<td>11</td>
<td>50</td>
<td>3</td>
<td>6</td>
<td>10</td>
</tr>
<tr>
<td>Agriculture</td>
<td>93</td>
<td>2</td>
<td>9</td>
<td>3</td>
<td>0</td>
<td>4</td>
</tr>
<tr>
<td>Waste</td>
<td>1</td>
<td>1</td>
<td>1</td>
<td>0</td>
<td>0</td>
<td>1</td>
</tr>
<tr>
<td>Other</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>% change 1990-2013</td>
<td>-27%</td>
<td>-66%</td>
<td>-59%</td>
<td>-54%</td>
<td>-87%</td>
<td>-39%</td>
</tr>
</tbody>
</table>
There have been large reductions in EU emissions of all the air pollutants since 1990, particularly SO₂. Reductions in NH₃ emissions have been much lower than for the other pollutants.

An investigation into the reasons for the decline in emissions of SO₂ in Western Europe over the 50 years from 1960 suggest it resulted from a combination of reduced energy intensity and improved fuel mix. For NOx end-of-pipe abatement measures played the dominant role in the reduction of NOx emissions over the same period (Rafaj et al, 2014).

Some sectors have, however, seen emissions increase in recent years. For example, emissions of PM₂.₅ and BaP from households have increased by 15% and 16% respectively over the last decade, mainly due to the use of wood and other solid fuel combustion for heating homes (EEA, 2015e). The use of biomass is growing in some countries due to government incentives/subsidies for renewable energy, rising costs of other energy sources, and/or public perception that it is a ‘green' option. In Sweden, for example, the use of biomass for district heating has grown from just a few per cent in the 1980s to nearly 50% of the district heating energy mix in 2010, due in part to the introduction of a carbon tax. In addition, some households have reverted to heating with solid fuels in response to economic hardship. This has happened recently in, for example, Greece and Ireland (EEA, 2015a). Residential heating also contributed 35% of BC emission in the EU in 2013, a similar proportion as road transport.

6. MITIGATION MEASURES FOR AIR POLLUTION AND CLIMATE CHANGE

In recent years most EU countries have put great emphasis on limiting climate change (e.g. UN, 2014), but there is no evidence of a similar focus on improving air quality (e.g. Williams, 2012). All EU countries have national climate change strategies, action plans and mitigation programmes at national, state/provincial, and municipal levels (UN, 2014).

Further mitigation of GHG emissions will, however, be required to meet the generally accepted international ‘target’ of limiting the average global temperature increase to +2°C compared to the pre-industrialised temperature. GHG emissions by will need to reduce by 40-70 % by 2050, relative to 2010 levels, and emissions levels by near zero by 2100 (Stocker et al., 2014).

Europe's air quality has improved markedly since 1990 but air pollution remains the principal environmental factor linked to preventable illness and premature mortality in the EU and still has significant adverse effects on much of Europe's natural environment (European Commission, 2013), yet for many years air quality has been regarded by many as a solved problem (e.g. Williams, 2012). This has resulted in a lack of political will to address the issue. This is perhaps best illustrated by the lack of ambition in the European Commission’s air quality strategy, ‘A Clean Air Programme for Europe’, published in December 2013 (European Commission, 2013). Despite the same pollutants playing an important role in atmospheric chemistry and climate change the emission targets in the proposed revisions to the NECD for 2030 will be almost met without any new emission control policies (Amann, 2015). In addition, this strategy did not tighten the ambient air limit values despite the evidence of the health effects below these levels (WHO, 2013).
Harrison et al. (2014) argue that to improve air quality both emissions and ambient air quality standards need to be simultaneously adjusted over time. One without the other will not result in good air quality in the future. The analysis of monitoring data, as part of the assessment of compliance with the EU ambient air quality limits, has identified the unintended effects of poorly designed vehicle emission standards (Carslaw et al., 2011). Without the ambient limit values these effects may not have been identified.

Policies to mitigate climate change are primarily based on meeting predetermined national emission targets. For example Germany (Umwelt Bundesamt, 2015) and Denmark (Danish Government, 2013) are committed to reducing GHG emissions by 40% by 2020 from 1990 levels; and the UK by 35% (Committee on Climate Change, 2015). These and other emission reduction targets, such as those agreed under the Kyoto Protocol and most recently by the Group of Seven Industrialised Nations (G7) in 2015 (The White House, 2015), have focused governments on determining the best way to achieve them. Policies have been determined largely based on the least cost of reaching an emission target with little, if any, consideration of the environmental effects on either air quality or the climate. This focus on emissions is not surprising as mitigating climate change requires co-operation at a global scale and the effect on the climate of reducing national emissions in isolation is marginal.

Despite the lack of air quality ambition amongst European policymakers there has been a growing recognition of the importance of analysing climate and urban air quality policies together over the last decade or so and a number of integrated assessment models have been developed. Williams (2012) defines these as “mathematical models incorporating atmospheric transport and chemistry with mitigation costs for all sources to arrive at an optimised pathway to meet a prescribed environmental target”. Whilst it is recognised that there are shortcomings in the use of economic models to identify the optimum package of measures, due to the difficulty in valuing all effects of climate change and air pollution (e.g. Kunreuther et al., 2014), this type of model is used to evaluate European and some national air quality policies.

The RAINS model has been used to develop cost-effective strategies to combat air pollution for both the European Commission and the CLRTAP since the 1990s. It was extended in 2006 to include GHGs and climate change. The renamed GAINS model now estimates historic and future emissions of 10 air pollutants and 6 GHGs for each country based on historic data from international energy and industrial statistics, emission inventories and data supplied by countries. It assesses emissions in five-year intervals up to 2050. For each country/region the effects of about 2,000 specific emission control measures and their costs can be assessed. GAINS calculates the effects on ambient air quality (PM$_{2.5}$, ground-level O$_3$, deposition of sulphur (S) and nitrogen (N)), and the subsequent impacts on human health, ecosystems and climate change (IIASA, 2015). However, there are limitations with this model, for example the coarse grid it uses (25 x 25 km) does not reflect the spatial distribution of urban air pollutants.

This type of air quality/climate change integrated assessment model has not, in general, been used in the development of climate change policy. Instead, as noted above, policy in this area has typically been evaluated in terms of the cost per tonne of GHG emissions mitigated, ignoring the many co-benefits of emission reduction (e.g. Nemet et al., 2010; William, 2012; Smith et al., 2014). This is because the policy objective is to reduce emissions and not explicitly to reduce damage. Policymakers need to identify the least cost package of measures that will enable the emission target to be met.
One advantage of assessing the co-benefits in these economic models is that it reduces the perceived cost to society of climate policies, making it more acceptable. In addition, the public may associate more readily with short term local health benefits than long term global climate benefits (e.g. Nemet et al., 2010; Smith et al., 2015).

Where the co-benefits of air quality have been assessed significant co-benefits have been found. However, this depends on the time scale of the emission reductions and the level of climate ambition. The greater the ambition to reduce GHG emissions, the greater the air quality benefits. This is illustrated in Figure 2, and is because more radical solutions to meet GHG targets will result in significant changes to energy supply (e.g. decarbonising electricity generation) and transport (e.g. electrifying road transport). Less radical solutions may involve opting for technologies or fuels, which may deteriorate urban air quality such as the burning of biomass in urban areas (William, 2012).

*Figure 2: Co-benefits of climate change mitigation for air quality (Stocker et al., 2014)*

Incorporating the co-benefits of air quality as well as other co-benefits such as energy security, traffic accidents, and wider environmental benefits, would further reduce the perceived costs of climate policies. For example, McCollum et al. (2013) estimated that the air quality and energy security benefits of stringent climate change policies would be 0.1-0.7% of global GDP in 2030. Smith et al. (2015) estimated that a wide range of public health and environmental co-benefits, including air quality, could reduce the cost of meeting the UK’s climate target from 0.6% of GDP to under 0.1%.

Determining the relative benefits of reducing emissions of different AQPs and GHGs is not straightforward given their different spatial and temporal effects. There is an argument that early priority should be given to abating the traditional AQPs because this would provide public health benefits, and reduce atmospheric concentrations of the short-lived radiative forcing pollutants $O_3$ and BC as well as $CH_4$. This approach would also allow sufficient time to restructure the global energy economy from fossil fuels dependence towards renewable energy.

Conversely, reducing $SO_2$ emissions would reduce the formation of secondary sulphate aerosols, which play an important role in reducing the overall warming. In the context of
European policy major progress has already been made in reducing emissions of this pollutant, which are now 13% of their 1990 level (EEA, 2015a), and will be further reduced to meet the Gothenburg Protocol target.

In the following discussion on the synergies and conflicts between air quality and climate change mitigation measures it should be noted that in reality it is very complex to determine priorities, and for climate change mitigation depends on the life cycle emissions which occur in different parts of the world and if decommissioning is considered, may occur over different time scales. The life cycle impacts are most important for the long-lived GHGs because the location of emissions is less important, whereas for urban air quality it is the location of the emissions and the presence of people that are most important.

There are several types of mitigation measure that can be introduced to reduce emissions from the energy and transport sectors in the relatively short term. In general, these can be divided into:

- Efficiency: minimise the use of energy and other resources and increasing the efficiency of energy production.
- Technological measures: reducing emissions typically ‘end-of-pipe’ technology.
- Fuel switching: replacing a high emitting fuel with a lower emitting fuel (e.g. replacing coal with natural gas, or using a cleaner version of the same type of fuel).
- Demand management: influencing the demand for a product or service.
- Behavioural change: changing the habits of individuals/organisations to reduce emissions.

These measures can be implemented through legislation, fiscal incentives, voluntary agreements, and public education programmes.

6.1. Energy Efficiency

The measures where there is clearly a co-benefit for both climate and air quality are those which result in a reduction in the demand for energy and transport (assuming no fuel switching) or those that enable the same activity to be carried out more efficiently. For these measures there will be a simultaneous reduction in both climate and air quality emissions. The significance of these reductions will depend on the magnitude of the source contribution to overall emissions, but there will clearly be a co-benefit.

Examples include energy conservation measures such as more energy efficient appliances and the reduction in road traffic through mode-switching. These reductions in demand, for energy and transport respectively, can, in reality, be difficult to achieve. For example, people may respond to the fitting of building insulation by increasing the temperature of their homes and not reduce their energy demand. Reducing the demand for car travel has proved to be very difficult except where there is very good public transport. Behavioural change can, however, be brought about with the use of consistent incentives, public education and legislation.
Technological innovation, such as smart phones coupled with intelligent information systems, can also affect behaviour, demand for services and products, and hence emissions.

6.2. Fuel switching

There is likely to be increasing dependence on electricity for heating and transport in the future. Fuel switching can have co-benefits for both urban air quality and climate change. For example, when a high carbon, fuel such as coal, is replaced by solar, wind or hydro power to generate electricity. It should be noted, however, that the production of photovoltaic cells are associated with emissions of NF₃, a very potent GHG. It has a GWP over 100 years 17,200 times that of CO₂, and atmospheric concentrations are increasing.

Wind and solar power are intermittent and it is likely that gas will continue to be an important fuel for power generation, to ensure availability of electricity at all times. In the future shale gas may become as important in Europe as it is in the United States, where it accounted for about one quarter of the total dry gas production in 2010. This may rise to almost half by 2035 (U.S. Energy Information Administration, 2012). There are significant uncertainties regarding the emissions from the production of shale gas in Europe not least because the lessons learnt from the US are likely to lead to tighter controls of fugitive emissions. One UK estimate of the life cycle emissions suggests, using a central estimate, that shale gas has a higher potential for the creation of photochemical oxidants, including O₃ than conventional gas sources. For acidification it is estimated to be a better option than coal power but is an order of magnitude worse than the other options (Stamford and Azapagic, 2014). Regarding GHG emissions shale gas is thought to be broadly comparable to conventional gas (MacKay and Stone, 2013). Although shale gas production is unlikely to take place in urban areas, there may be indirect impacts due to the increased formation of secondary PM which often contributes a significant proportion of urban PM₂.₅ concentrations.

Any increase in the use of natural gas risks increasing the emissions of CH₄ though leakages. A switch from coal to natural gas power generation would reduce CO₂ and sulphate emissions, but net CH₄ emissions might rise if there is substantial leakage in the production and distribution of natural gas that outweighs the decline in CH₄ release from reduced coal mining. While the reduction of sulphate is beneficial to health and welfare, it accelerates near-term warming, as would any additional CH₄ released to the atmosphere (Fiore et al., 2015).

Fuel switching does not always result in benefits for both urban air quality and climate change. Several EU member states have provided incentives to encourage biomass burning to reduce CO₂ emissions from the burning of fossil fuels. However, the combustion of biomass is a significant source of PM and BaP in a number of cities (e.g. AIRUSE, 2015a, 2015b).

6.3. Combustion trade-offs and technological measures

During combustion of fossil fuels in both stationary and mobile applications there are trade-offs between emissions of NOx, CO₂, and the products of incomplete combustion (PM, CO and VOCs). This is because NOx emissions are highest during efficient combustion when CO₂ emissions are at their lowest. There are three main approaches to these trade-offs. One approach is to control for low NOx (e.g. use low NOx burners) and accept a fuel efficiency penalty and possibly higher CO and VOC emissions. A second option is finding an optimal
balance between all emissions. A third option is to design for the lowest CO\textsubscript{2} emission and use ‘end-of-pipe’ technology to reduce the NOx, and possibly other pollutants.

NOx emissions can be reduced by changing the combustion conditions (known as primary abatement). These techniques include low excess air combustion, staged combustion, flue/exhaust gas recirculation, low NOx burners, and water injection to reduce the flame temperature. The use of these techniques depends on the fuel and application. They are associated with a reduction in energy efficiency and an increase in CO\textsubscript{2} emissions. The reduction in NOx emissions without this energy penalty requires the use of after-treatment (secondary abatement) using selective catalytic reduction (SCR), selective non-catalytic reduction (SNCR) or non-selective catalytic reduction. Again, the choice of technology depends on the application.

SO\textsubscript{2} emissions from stationary sources are typically reduced by fitting flue gas de-sulphurisation (FDG), which in simple terms uses an alkali (e.g. limestone or sea water) to convert the SO\textsubscript{2} to sulphate. CO\textsubscript{2} is also produced, and there is a further CO\textsubscript{2} penalty due to the power needed to run the FGD plant. Both wet and dry FGD systems are available with the former tending to be used for larger combustion plant. Dry scrubbers also require the PM formed in the process to be removed. Depending on the sulphur content of the fuel the CO\textsubscript{2} emissions may increase by 3\% or more (AQEG, 2007). Sulphur can also be removed from the fuel before combustion. Hydrodesulphurisation is typically used for petroleum products such as gasoline and diesel and takes place at the refinery. However, fuel switching from coal and heavy oil to natural gas, which has a low S content, has been responsible for much of past reduction in SO\textsubscript{2} emissions in Europe.

PM emissions can be reduced by increasing the combustion efficiency, but with increased NOx emissions. Industrial PM emissions are commonly reduced using electrostatic precipitators (ESP), fabric filters or wet scrubbers. These technologies will also have a small CO\textsubscript{2} penalty during their operation.

Abatement technologies have also been developed that provide multi-pollutant emission control, such as Regenerative Activated Coke Technology (ReACT\textsuperscript{TM}), which controls SO\textsubscript{2} and NOx in a multistage regenerative process from stationary combustion sources.

Whilst ‘end-of-pipe’ technologies are useful for reducing the AQPs emissions, there are few options for using this type of technology to reduce CO\textsubscript{2} emissions. Carbon capture and storage technology is currently in its infancy with few commercial plants operating anywhere in the world (CCS Association, 2015).

There are often many mitigation options available, some proven and others still in the development and demonstration phases. This report does not attempt to provide details of all possible options, but provide an indication of the possible conflicts and synergies.
6.4. Stationary combustion - technological measures

The electricity supply industry\(^6\) is the largest contributor to the EU greenhouse gas emissions, accounting for 27% of the total in 2012; while manufacturing contributed 19%. In the short to medium term Europe will continue to be dependent on fossil fuels for electricity and industrial processes, as well as other uses. The only method available to abate CO\(_2\) emissions is the use of carbon capture and storage (CCS). CCS can capture up to approximately 90% of CO\(_2\) emissions and store it in deep geological formations. Although it is not compatible with the EU’s long-term aim of decarbonising the electricity supply industry if proved viable it could have a role to play in reducing CO\(_2\) emissions in the medium term. New coal-fired power stations are being still being commissioned, particularly in eastern and central Europe, which will have long lifetimes and these, as well as future plants, could be retrofitted with CCS when the technology matures. EU legislation requires new plants to have the space available for fitting CCS in the future. The EU’s energy forecast assumes that there will be some demonstration CCS plants by 2020 but thereafter there will be no large increase in capacity up to 2030 (European Commission, 2010).

There are several methods to capture CO\(_2\). Post combustion capture is the preferred technique at present, which scrubs the flue gas with a chemical solvent. For coal and oil-fired power stations the impurities in the flue gas, which include SO\(_2\), NO\(_x\) and PM need to be removed. This process therefore requires that AQPs emissions are removed to operate efficiently. However, a large amount of energy is required to regenerate the solvent and to compress the CO\(_2\) for transport, which significantly reduces the net electricity output and efficiency of the plant. Although the reduced efficiency would not lead to a net increase in CO\(_2\) emissions other AQPs and GHG emissions would increase along the fuel chain simply because of the loss in overall efficiency at the power plant.

‘End-of-pipe’ abatement technologies have reduced emissions of SO\(_x\), NO\(_x\) and PM emissions for many years. They are standard on most large combustion plant in the EU because of the emission limits in the Large Combustion Plant Directive (2001/80/EC) and the Industrial Emissions Directive (2010/75/EU). Those power plants that do not comply with the emission limits will need to reduce emissions and eventually be shut or retrofitted over a transition period. If these abatement techniques were to be replaced by CCS there may be a small reduction in the overall energy penalty.

There are currently no EU emission limits for medium combustion plant. AQPs emissions from medium combustion plants (1-50 MW) are regulated in most European countries, although the UK only has emission limits for plant in the 20-50 MW size range. These plants can be a significant source of SO\(_2\), NO\(_x\) and PM emissions in urban areas. They are used for a wide variety of applications including electricity generation and providing building heating/cooling, as well as heat/steam for industrial processes (AMEC, 2014).

The current proposals would set emission limits for existing 1-50 MW plant generally the same as those for larger plants (50-100 MW), but with less stringent limits for NO\(_x\). These emission limits will be less stringent than those currently applied in some member states. This leniency is because the cost of after-treatment is considered too great for these plants in most instances

\(^6\) Which also includes gas, steam and air conditioning supply
as they tend to be operated by medium and small enterprises. Instead, these plants are likely to use low-NOx burners with a small CO$_2$ penalty to meet the emission limits. New and existing plants where EU ambient air quality standards are not met would, however, be expected to meet more stringent emission limits.

Table 3 provides a summary of the technologies and their potential abatement efficiencies, summarised from the European Commission assessment of options for controlling the emissions for medium combustion plant (AMEC, 2014). The abatement efficiencies are dependent on the fuel used and the size of the plant. The AMEC report does not provide details of the CO$_2$ impacts of the technologies.

Table 3: Illustrative Efficiency of Abatement Techniques (adapted from AMEC, 2014)

<table>
<thead>
<tr>
<th>Technology</th>
<th>SO$_2$ (%)</th>
<th>NOx (%)</th>
<th>PM (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Cyclone</td>
<td></td>
<td>65%</td>
<td></td>
</tr>
<tr>
<td>Fabric filter</td>
<td></td>
<td>95-99%</td>
<td></td>
</tr>
<tr>
<td>Electrostatic precipitator</td>
<td></td>
<td>97%</td>
<td></td>
</tr>
<tr>
<td>Combustion modification</td>
<td>30-50%</td>
<td></td>
<td></td>
</tr>
<tr>
<td>SNCR</td>
<td>35-45%</td>
<td></td>
<td></td>
</tr>
<tr>
<td>SCR</td>
<td>70-90%</td>
<td></td>
<td></td>
</tr>
<tr>
<td>NSCR</td>
<td>94%</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Water injection</td>
<td>40-60%</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Wet FGD</td>
<td>94%</td>
<td></td>
<td>99-100%</td>
</tr>
<tr>
<td>Dry FGD</td>
<td>64-80%</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Co-fired biomass</td>
<td>20%</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

The need to increase the efficiency of power generation is leading to greater use of the waste heat and there are growing numbers of medium sized combined heat and power (CHP) plants and boilers supplying heat to residential areas. CHP can increase the overall efficiency of the fuel used to more than 75% compared with around 40-50% from conventional electricity generation (AQEG, 2007). In addition, locally generated electricity can also reduce transmission and distribution losses. It requires a local demand for heat such as an industrial process or residential district heating, and therefore many plants are located where there is public exposure to the AQP emissions. District heating is more efficient in terms of both the climate pollutants and the AQPs than a large number of small individual heating systems, and given the size of the plant, it is likely to be economically viable to fit abatement for the AQPs. However if the emissions are not well abated CHP and or district heating could lead to local pollution hotspots.

Large-scale electricity generation tends to be located in rural areas to reduce public exposure to the emissions. Moving power stations back into urban areas may increase net local emissions depending on the feedstocks and location. Even burning relatively clean natural gas may have a detrimental local impact on ambient NO$_2$ concentrations where this is an additional emission source.

One of the drivers for the proposed medium combustion plant Directive is the conflict between promoting solid biomass as a renewable energy source to reduce GHG emissions and the higher PM emissions during its combustion compared to gas. The combustion of pre- and post-consumer waste wood can increase the heavy metals in the PM emissions. This is due to the
use of these metals in surface coatings and preservatives, particularly in the furniture industry. However, emission limits for all plant over 1 MW should ensure that suitable abatement is fitted to reduce PM emissions.

New EU minimum energy efficiency requirements and emission limits for space heaters and boilers were agreed in 2013 under the Ecodesign Directive (2009/125/EC). These will be introduced over the period 2015-2022. The following are included:

- Space heaters under 400 kW (excluding solid fuels) (Regulation 813/2013)
- Water heaters under 400 kW (excluding solid fuels) (Regulation 814/2013)
- Solid fuel boilers with a rated heat output of 500 kW or less applicable from the beginning of 2020 (Regulation 2015/1189)

Further Regulations have been agreed for domestic and small commercial space heaters. These are discussed in the domestic heating section (6.6) of this report.

Fuel switching can be an effective means of reducing the AQPs. For example, co-firing biomass with other solid fuels or switching from a high to a low sulphur coal could reduce SO\textsubscript{2} emissions. Switching from solid fuel to natural gas would remove all SO\textsubscript{2} emissions and reduce NOx by approximately 50% and PM by 99% (AMEC, 2014). However changing from a solid to a gaseous fuel would require major changes to existing plant.

Overall electricity consumption will grow in the future due to the increasingly demand for its use in transport and for heating, but significant changes in the mix of fuels used to generate electricity are likely. According to the European Commission (2010) in 2030 renewable energy is likely to produce more than one third of electricity while the use of solid fuels will decline from 30% in 2005 to 21% in 2030. The share of gas generation will decrease from 21% to approximately 18%, although in absolute terms it will increase slightly. Gas will continue to play an important role in balancing renewable electricity. Nuclear energy is also predicted to reduce in terms of share, but in absolute terms (TWh) nuclear electricity in 2030 is likely to be at almost the same level as in 2005. These forecasts assume that the two relevant EU targets for 2020 (20% renewable energy and 20% GHG reductions) will be achieved (EU, 2010).

The EU energy forecasts do not provide a prediction of the impact of their scenarios on the AQPs but the move from solid and gaseous fuels towards more wind, solar and hydro power will reduce emissions. Over the period 2005 to 2030 the carbon intensity of power generation is forecast to reduce by approximately one third. Reducing the amount of fossil fuels burnt in power stations will also reduce CH\textsubscript{4} emissions from deep-mines and from the extraction of natural gas.

This section discusses stationary combustion but much of it is also applicable to shipping, although there are unique issues for ocean going vessels due to the harsh conditions and greater space constraints. Further information on the abatement of shipping emissions is provided in the AIRUSE report ‘The Control of Shipping Emissions’.
6.5. Road Transport

The increasingly stringent AQP emission limits for road vehicles over recent years has resulted in the adoption of a number of end-of-pipe technologies i.e. catalytic convertors and particle filters. These have generally resulted in small CO$_2$ emission penalties. For example, it has been estimated that fitting an oxidation catalyst and particle filter to a Euro IV light duty diesel engine increased fuel consumption by 7% (Ricardo, 2005).

Associated with these technologies has been a need to reduce the sulphur content of gasoline and diesel such that fuels used for road transport and in off-road mobile machinery is now sulphur free (i.e. maximum 10 ppm). Reformulation of fuel often leads to an increase in energy consumption at refineries, and a consequent increase in CO$_2$ emissions. It has been estimated that the hydrodesulphurisation of road fuels from 50 ppm to 10 ppm increased CO$_2$ emissions from EU refineries by 6.5% (CONCAWE, 2000). These refinery emissions, however, may have been be offset by the adoption of more energy efficient vehicle technologies that sulphur free fuels have enabled.

Sulphur free diesel enabled selective catalytic reduction (SCR), and lean NOx traps (LNT) to be used to abate NOx emission, which in turn has enabled diesel engines to be tuned for lower fuel consumption. The use of SCR has reduced fuel consumption from diesel engines by approximately 3-5% (e.g. Iveco, 2015). In addition, the requirement for S free gasoline has enabled fuel efficient direct injection gasoline engines fitted with NOx abatement$^7$ to be produced which offer additional fuel savings.

One disadvantage of the use of SCR is that emissions of NH$_3$ and N$_2$O may increase. These emissions can, however, be reduced by using an oxidation catalyst to remove the NH$_3$ and choice of SCR catalyst to ensure minimal N$_2$O formation.

To reduce PM emissions diesel vehicles are fitted with diesel particle filters (DFPs) in the EU. In addition, a large number of older heavy-duty vehicles (HDVs) have been retrofitted with DPFs, particularly those that operate in urban areas. These filters need to be regenerated periodically to avoid the buildup of back-pressure in the engine. There are several methods for doing this but these also have small fuel consumption penalties and associated increase in CO$_2$ emissions.

Virtually all commercial vehicles and buses use diesel as the engines are long lasting and more fuel efficient than the equivalent gasoline engines. However, for passenger cars there is a choice between gasoline and diesel. Slightly more new diesel cars are sold in the EU than gasoline cars, but in some countries three quarters of new cars are diesel (ICCT, 2014). Less than 3% of new cars use other fuels, of which approximately 1% was hybrid, 2% natural gas and less than 1% electric vehicles (including fuel cells and plug-in hybrids) in 2013. Most (85%) of the gas vehicles were sold in Italy where there has been a long history of promoting the use of gas for both light- and heavy-duty vehicles. Elsewhere the use of natural gas is low.

---

$^7$ Further information on NOx abatement from road vehicle is provided in the AIRUSE report ‘Abatement of nitrogen oxides (NOx) emissions from vehicles’.
and diesel and gasoline are likely to remain the main fuels for road transport for the foreseeable future.

One of the reasons for the growth in the diesel market share has been the use of fiscal incentives for low CO₂ emitting cars to mitigate climate change. For example, 20 EU member states have CO₂ based taxes on vehicles (ACEA, 2015). In addition, the motor industry has promoted these vehicles to enable them to meet the EU CO₂ standards (Regulation 443/2009/EC as amended). Each car manufacturer has an average CO₂ emissions standard depending on their individual mix of products sold. The aim of these manufacturer standards was to give a new car fleet average across the EU of 130 gCO₂ km⁻¹ by 2015. This fleet average target has been revised to 95 g CO₂ km⁻¹ by 2021. The average emissions of a new car sold in 2014 was 123.4 g CO₂ km⁻¹, well below the 2015 target (European Commission, 2015).

Diesel cars are more fuel efficient than equivalent gasoline cars but due to the different densities of the fuels the CO₂ benefit is less than the fuel consumption benefits. There is some controversy as to whether the dieselisation of the car fleet has led to a significant reduction in CO₂ emissions. On a like-for-like basis (e.g. power and size) diesel cars have lower CO₂ emissions than gasoline cars. However, it has been suggested that the CO₂ benefits of diesel cars have been marginal (e.g. Zachariadis, 2013; Cames & Helmers, 2013; Schipper & Fulton, 2008)⁸ because the purchasers of diesel cars tend to choose larger, heavier and more powerful vehicles to replace gasoline cars; and they are driven 40-100% further. Between 2001 and 2011 the average power of new gasoline cars increased by 7.5%, while over the same period the power of diesel cars increased by 22% (ICCT, 2014).

In addition, the official new car CO₂ emission data has under-estimated the real world CO₂ emissions with the gap widening from approximately 8% in 2001 to 40% in 2014 for all new cars. For diesel cars the gap has been higher than for gasoline cars since 2010, and is now approximately 5% (ICCT, 2015). This suggests that average new car CO₂ emissions may have declined by only 8% compared with official 27% reduction over the period 2001 to 2014 (ICCT, 2015). Given the higher PM and NOx emissions from diesel cars compared to conventional gasoline cars, there is likely to have been even less net benefit in terms of climate change. Furthermore, modelling of the European refineries by the JEC Research Programme, shows that it is more energy intensive to produce diesel than gasoline (JEC Research Programme, 2011).

Liquid petroleum gas (LPG) and compressed natural gas (CNG) cars are generally converted from gasoline-fuelled cars, either by the original manufacturer or by an aftermarket converter. Most LPG and CNG cars are bi-fuel, which can run on either gasoline or the gaseous fuel. CO₂ emissions from LPG vehicles tend to be between those of gasoline and diesel, while emissions from CNG vehicles tend to be lower, typically comparable with that of diesel vehicles. AQP emissions performance of well-engineered LPG and CNG vehicles is similar to that of a gasoline vehicle (VCA, 2015). Only Italy and Poland have a significant proportion of

---

⁸ This is discussed further in the AIRUSE report ‘Taxation and pricing strategies to discourage the use of diesel vehicles’
gas vehicles in their national fleets. In Poland they are mainly conversations of older gasoline cars.

The main air quality advantage of natural gas vehicles is that they have virtually no exhaust PM emissions and may also have lower NOx emissions. The disadvantage is that their use increases CH$_4$ exhaust emissions. Furthermore, CH$_4$ emissions from gas leaks during extraction, distribution and storage may increase if gas demand increases. In addition, there are CO$_2$ emissions associated with the energy needed to compress the gas.

The Renewable Energy Directive (Directive 2009/28/EC) requires transport fuels to meet a 10% renewable energy target by 2020 in the EU. This applies to all types of transport. The Fuel Quality Directive (2009/30/EC) requires 6% reduction in the GHG intensity of fuels traded in the EU by 2020, and applies to fuels used in road vehicles, and a range of non-road mobile applications. There is also a 7% limit on the use of biofuels grown on land (known as first-generation biofuels). This is to reduce the indirect GHG emissions arising from the increase in the land under cultivation.

Bioethanol and biodiesel are commercially available across all EU Member States and are widely used in fuel blends for vehicles. In the EU, road transport biofuels account for 4.2% of transport energy consumption in 2012, with more biodiesel sold than bio-gasoline (European Union, 2014). At the low bioethanol blends currently used in gasoline there is unlikely to be a significant impact on air quality, however at higher blends there may have a negative impact on O$_3$ in southern Europe (Hubbard et al., 2014).

The future expansion of the use of biofuels in road transport depends on the development of advanced, second generation biofuels, made from alternative feedstocks such as waste and algae. Commercialisation has been slower than expected (e.g. JEC Research Programme, 2014). It also depends on the development of engines able to use higher blends of biofuels and/or the development of biofuels with properties 'near-identical' to their fossil fuel counterparts. For road transport, the most promising biofuels include the use of 7% biodiesel (fatty acid methyl ester; FAME), bioethanol at 10% and 20% blends, lignocellulose fuels and hydrotreated vegetable oil, as well as pure plant oil for use in fleets.

It has been suggested, however, that the use of biofuels results in a rebound effect. That is an increase use of biofuels reduces oil demand, which in turn results in a decrease in the oil price. This leads to higher demand for oil, which causes oil consumption to decrease less than the increase in biofuel use (on an energy content basis). The rebound effect in the biofuel producing region is usually negative, which means that biofuels are effective in reducing domestic GHG emissions and increasing energy security. However, there is a positive rebound effect elsewhere. Further, a shift from costly first-generation biofuels to cheaper second generation biofuels may mean higher rebound effects, which may partially undo the reduction in the indirect land use emissions (Smeets et al., 2014).

There is a wide range of biofuel feedstocks, which have different impacts on AQP emissions. The emissions from biofuels depend on whether neat or blended fuels are used, the vehicle tested and the degree to which the vehicle has been optimised for the use of these fuels.

An important benefit of biodiesel blends is that they can reduce the emissions of PM, as well as CO and NMVOCs, but they can also increase NOx emissions (e.g.; Hajbabaiei et al., 2014;
Mizushima et al., 2014; Hajbabaeei et al., 2012; Maunula & Kinnunen, 2009). Mizushima et al. (2014) found, under real-world driving conditions a significant difference between biofuels with FAME blends performing the worst. NOx emissions with hydrotreated vegetable oil HVO were equivalent to those of zero sulphur diesel.

Hybrid vehicles combine an electric motor with either a diesel or gasoline internal combustion (ICE) engine. They come in several different forms. A mild hybrid cannot propel the vehicle on its own; instead, it saves fuel by shutting off the engine when the vehicle is stopped, braking or cruising. Some mild hybrids can also capture mechanical energy during braking. These are essentially fuel saving devices, and as such used on a number of conventional vehicles. They may benefit both AQP and GHG emissions, provided the manufacturing emissions remain similar. However these is some evidence that they work well under the official test conditions, but less so under real driving conditions (ICCT, 2015), and in reality mild hybrids may offer no air quality or climate benefits.

Hybrid vehicles can run on either an internal combustion engine (ICE) or a battery powered electric motor or both. The majority use an ICE aided by a small electric motor. The size of the motor limits the range of the vehicle when operating in electric mode.

Alternatively, plug-in hybrids are powered primarily by the electric motor with a small ICE generator producing electricity to run the electric motor. Mains electricity can also be used to recharge the batteries. The ICE essentially acts as a range extender, which is useful while the electric recharging network remains sparse, compared to conventional filling stations.

In a full hybrid the ICE is designed to run at maximum efficiency. Extra power is provided from the battery, when necessary, such as during acceleration. When the car is decelerating or standing still, the batteries recharge. While a conventional hybrid vehicle may travel short distances in pure-electric mode, plug-in hybrids are designed to travel extended distances with little or no assistance from the ICE. Full hybrid vehicles are heavier than conventional vehicles as they need an ICE, an electric motor and additional batteries, and therefore use more energy to move and accelerate. However, in urban areas, particularly where there is heavy congestion, these vehicles have lower GHG and AQP emissions. This is because hybrids tend to use electricity under these conditions as little or no power is required, while ICEs are most inefficient under these driving conditions.

Electric vehicles have zero exhaust emissions at the point of use (there are non-exhaust PM emissions), and therefore provide benefits in terms of reduced AQPs but their life-cycle emissions depend on the fuel used at the power station, and until power generation is decarbonised these will not be truly zero emission vehicles. Intelligent charging strategies are being developed that will enable EVs to be recharged whilst managing electricity demand and the fluctuations inherent in renewable electricity generation, particularly wind energy. A ‘smart grid’ can allow battery EVs to be re-charged according to electricity production patterns, but requires a completely new way of organizing the interface between the transport and energy sectors.

It should be noted that none of these technologies eliminate PM emissions from road traffic as non-exhaust emissions, such as from the brakes, tyres, and resuspension of road dust, can contribute a significant proportion of the road transport PM (e.g. Harrison et al, 2011).
Finally hydrogen ($H_2$) can be used as a road fuel either directly in an internal combustion engine or in a fuel cell. When combusted directly, it results in zero emissions of $CO_2$ and very low emissions of the other AQPs. When used in a fuel cell, the only emission is water vapour. $H_2$ offers significant advantages over other road fuels, and its use would be beneficial for urban air quality. The key issue is how it is produced, stored and distributed. There are numerous ways it can be produced and consequently a very large range of air pollution impacts. In the short term, natural gas is the only viable route to large-scale $H_2$ production but it only has a net benefit for road vehicles if used in a fuel cell. Producing $H_2$ using electrolysis from the existing EU electricity mix would increase the GHG emissions.

### 6.6. Residential Heating

Direct GHG emissions from homes contributed 9% to the total EU GHG emission in 2012 (EEA, 2015b). This excludes the emissions due to electricity production and from district heating schemes. Local space heaters (typically open fireplaces and solid fuel stoves) and boilers are significant energy users, and solid fuel appliances in particular contribute to emissions of PM.

In recent years there has been growing evidence of an increase in the burning of biomass within homes in urban areas across the EU. This is through to be due to a combination of it being perceived as a ‘green fuel’ and being more economical than fossil fuels, especially in areas where financial incentives are offered to encourage its use. In addition, wood is burnt to provide ‘ambience’ rather than just heat; so called recreational burning.

Wood burning is widespread in Scandinavia and the Alpine regions and is known to be associated with local air pollution. Elsewhere in northern and central Europe smoke emitted from domestic wood burning was believed to be not significant. However, there is recent evidence that wood smoke can make a significant contribution to PM concentrations during winter in the centre of cities in North West Europe, and can exceed traffic emissions (e.g. Crippa et al., 2013; Fuller et al 2013; Fuller et al., 2014; Wagener et al., 2012).

To reduce these emissions new EU regulations were agreed in 2015 for:

- Solid fuel local space heaters (e.g. fireplaces and wood burning stoves) under 50 kW (Regulation 2015/1185);

- Gaseous and liquid fuel local space heaters. These apply to domestic heaters up to 50 kW and commercial heaters up to 120 kW (Regulation 2015/1188).

For solid fuel, there are emission limits for PM, NOx, organic gases, and CO, while for gaseous and liquid heaters limits have been set only for NOx. These requirements will be introduced over the period 2020-2022, and it is likely to be many years before large benefits of these regulations accrue due to the slow replacement of these products.

It has been estimated that PM emissions from local space heaters could be reduced by 9-14 % in 2020 and 36-42 % in 2030 by introducing emissions limits and the labelling of appliances. Local space heaters also release emissions of NOx although there is insufficient information available to quantify the benefit of controlling these emissions (European Commission, 2015b).
7. **SUMMARY AND CONCLUSIONS**

Comparing air quality and climate change impacts and the different synergies and trade-offs is problematic because there is no common metric available for comparison. For climate change, it is important to consider fuel cycle or life cycle emissions, while for air quality the location of emissions (i.e., distance to centres of population) and altitude are of prime importance. The timescales are also different. The climate adjusts slowly to reductions in emissions of the main GHGs whereas air quality responds rapidly. The interactions between GHGs and the AQPs in the atmosphere are very complex and non-linear.

Policies to reduce climate change and improve urban air quality have generally been considered in isolation, with more prominence being given to the mitigation of climate change than urban air quality over recent years. In the long term, large reductions in both the AQPs and GHGs are needed to minimise climate change and improve public health. Priority should be given to measures where there are clear co-benefits such as energy conservation measures. However large emissions reductions from this type of measure can be difficult to achieve in reality, and there will continue to be a need to use legislation to force the adoption of low AQP emitting technologies despite some CO₂ penalties. Fuel switching to renewable fuels offers a huge potential for co-benefits, with only biomass and biofuels being problematic in terms of indirect GHG emissions from land use changes and higher emissions of PM from solid biomass and gaseous pollutants from some liquid biofuel blends.

Many EU governments have policies that can inadvertently lead to the deterioration of air quality. These include car taxation based on CO₂ emissions and fuel taxation that favours diesel over gasoline. These measures have led to the increasing share of diesel cars. Similarly, fiscal incentives for biomass for stationary combustion, without good PM control, has lead to increasing AQP emissions. Simple, but misleading, public education campaigns have resulted in many people believing that diesel and biomass are ‘cleaner’ than their alternatives. The CO₂ benefit of diesel cars is at best marginal but real world emissions of PM and NOx are significantly higher. Life cycle emissions of CO₂ from the combustion of biomass can be significantly lower than from fossil fuels, provided it has not been grown on previously uncultivated land, but emissions of PM and BaP are significantly higher compared to gas and oil fired domestic heating systems.

To reduce emissions from road transport requires a large shift to electric vehicles, which are more efficient in urban driving conditions than internal combustion engines, and have no exhaust emissions at the point of use, coupled with an increasing proportion of renewable electricity. These vehicles should be prioritised for fiscal incentives to increase their market share, and there should be a move away from graduated CO₂ taxation of passenger cars based on misleading ‘official’ CO₂ emissions (which are on average 40% higher than when driven on the road). This system has failed to address urban air quality.

Emissions of PM and BaP from residential biomass burning have increased over the last decade as policies to promote renewable energy have taken effect. Ambient concentrations are particularly high where biomass burning is widespread. Modern automatic pellet and wood chip appliances are significantly more energy efficient and have lower emissions than traditional fireplaces and wood log stoves, but most residential biomass burning is likely to take place in inefficient and polluting appliances.
AQP emissions limits for combustion plant of various sizes are proposed or already agreed for introduction over the next few years. This includes plants with a thermal input in the range 1-50 MW (proposed medium combustion plant directive); solid fuel boilers up to 300 kW (Eco-design Regulations); and space heaters up to 50 kW (Eco-design Regulations). There are no proposed limits for boilers in the 300 kW to 1 MW range; and in countries where there are a significant number of these plants affecting urban air quality, national governments may consider the introduction of emission legislation to fill this gap.

The emission limits in the Eco-design Regulations are not as stringent as those currently adopted by some member states. The lowest PM limits have been adopted in Germany, and there is a case for other member states to follow Germany’s lead and introduce similar standards as soon as possible, although a correction factor of up to 5 should be applied in order to account for the condensable fraction.

In the meanwhile Member States are encouraged to promote the use of appliances that meet Eco-design benchmarks (e.g. the Blue Angel and Nordic Swan schemes) through a certification scheme, or by linking fiscal incentives to appliances which achieve the benchmarks, accompanied by an effective public education campaign. Only high quality certified pellets should be allowed to be sold. Wood logs should have a maximum moisture content and the use of contaminated waste wood should be banned for domestic use. Users should be provided with advice on how to store and use biomass effectively.

National and regional governments could support voluntary replacement/retrofit programmes to encourage households to replace older appliances with safer, more efficient, cleaner burning technologies. Financial incentives may be necessary to assist with the capital costs. In addition, in areas where high concentrations of PM are forecast domestic biomass burning in old inefficient appliances could be banned, except where this is the only source of heating in a home.

Many of the options that are likely to be important in the longer-term to mitigate climate change (e.g. such as increased use of electricity for transport and heating) result in a shift of emissions from the point of use to that of fuel production. In terms of urban air quality this is likely to be of benefit, provided the majority of the fuel production uses clean renewables such as solar and wind power or takes place outside urban areas. Thus in the long term there are likely to be co-benefits for both urban air quality and climate change.

In the short term, i.e. the next decade or so, there are inherent difficulties in comparing the impact of measures to reduce emissions of pollutants of concern from an air quality perspective and those that have direct or indirect impacts on the climate due to the different spatial and temporal impacts of the emissions.

Table 4 provides a summary of the likely co-benefits and disbenefits of a range of generic abatement measures to mitigate climate change and improve urban air quality.
Table 4: Impact of generic mitigation measures on climate change mitigation and urban air quality in the short term (+ positive effect; - negative effects; = no effects; ? currently not clear)

<table>
<thead>
<tr>
<th>Generic measure</th>
<th>Likely Positive Impact on:</th>
<th>Notes</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Climate Change</td>
<td>Urban Air Quality</td>
</tr>
<tr>
<td>Energy conservation</td>
<td>+</td>
<td>+</td>
</tr>
<tr>
<td>Reducing demand for a product or service</td>
<td>+</td>
<td>+</td>
</tr>
<tr>
<td>End–of-pipe technologies</td>
<td>-</td>
<td>+</td>
</tr>
<tr>
<td>De-sulphurisation of fuels</td>
<td>-</td>
<td>+</td>
</tr>
<tr>
<td>Dieselisation of the car fleet</td>
<td>?</td>
<td>-</td>
</tr>
<tr>
<td>Biofuels</td>
<td>+</td>
<td>?</td>
</tr>
<tr>
<td>Hybrid vehicles</td>
<td>+</td>
<td>+</td>
</tr>
<tr>
<td>Increased use of electricity e.g. for heating</td>
<td>?</td>
<td>+</td>
</tr>
<tr>
<td>Domestic biomass burning</td>
<td>+</td>
<td>-</td>
</tr>
<tr>
<td>CHP</td>
<td>+</td>
<td>?</td>
</tr>
<tr>
<td>District heating</td>
<td>+</td>
<td>?</td>
</tr>
<tr>
<td>Solar, wind, tidal electricity</td>
<td>+</td>
<td>+</td>
</tr>
<tr>
<td>Hydrogen</td>
<td>?</td>
<td>+</td>
</tr>
</tbody>
</table>
8. REFERENCES


AMEC Environment and Infrastructure UK Limited, 2014. Analysis of the impact of various options to control emissions from the combustion of fuels in installations with a total rated thermal input below 50 MW. London.


AQEG, 2007, Air Quality and Climate Change: A UK Perspective. Published by Defra, London.


