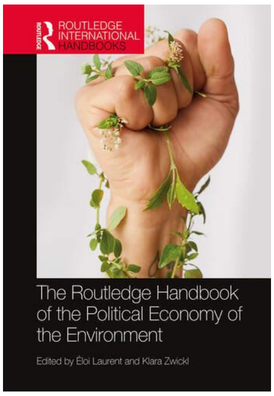

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AIR QUALITY CO-BENEFITS OF CLIMATE MITIGATION IN THE EUROPEAN UNION

Klara Zwickl and Simon Sturn[1]

Introduction

Climate policy is typically framed as distributional conflict between countries, with some countries free riding on the costly efforts of curbing carbon emissions of others, or between generations, with current generations living at the expense of future ones. These perspectives neglect that burning fossil fuels not only releases greenhouse gases into the atmosphere, but also emits several hazardous co-pollutants that negatively affect local air quality and harm the health of people living in proximity to the emitting sources. Climate policy thus not only reduces global climate change, but also improves local air quality.

Incorporating these air quality co-benefits into an assessment of climate policy is crucial to get a broader picture of its costs and benefits. A substantial number of studies show that the benefits of climate policy are vastly underestimated if co-benefits are omitted from the analysis. Co-benefits alone are typically found to justify a substantial increase in carbon prices, independent of any climate benefits. Incorporating co-benefits also highlights the importance of another distributional dimension that is often neglected; the one between rich and poor neighborhoods. Since socioeconomically disadvantaged neighborhoods are typically more exposed to co-pollutants – both within and across countries – they benefit more from air quality improvements through policies that reduce fossil fuel combustion. Climate policy can thus reduce environmental inequality in air pollution exposure.

This chapter discusses the role of air quality co-benefits of climate mitigation in the European Union. In section What are air quality co-benefits and why do they matter?, we describe what air quality co-benefits are and why they matter. In section Why are greenhouse gases and co-pollutants regulated separately?, we review the regulation of greenhouse gases and co-pollutants in the EU. In section How large are air quality co-benefits?, we give an overview of the literature on the magnitude of air quality co-benefits, and then in section What are implications of including air quality co-benefits in European climate policies?, we discuss implications of including co-benefits into EU climate policy. Conclusion section concludes the chapter.

What are air quality co-benefits and why do they matter?

Climate policy, or the lack thereof, is often viewed as a global coordination failure. Since a ton of carbon emissions has the same climate impact independent of the location of its release, in

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the absence of binding international agreements every country or region has incentives to free ride on others' carbon mitigation efforts. As a public good, climate stabilization will therefore be underprovided and carbon leakage – emissions shifting from regulated to unregulated locations – can undermine emissions reductions of single countries or regions (Fowlie 2009; Kuik and Hofkes 2010; Eichner and Pethig 2011; Aldy and Stavins 2012). Additionally, the long timespan between emissions releases and their climate effects imply a trade-off between generations, which requires balancing the economic costs of carbon mitigation today with the environmental benefits arising in the future (Arrow et al. 1996; Nordhaus and Boyer 2000).

Framing the optimal level of climate policy as distributional struggle across countries and generations overlooks the local and immediate benefits of carbon mitigation here and now. A growing body of literature on co-benefits of climate policy has emphasized the existence of large positive spillovers from fossil fuel abatement, ranging from environmental health improvements to positive fiscal impacts due to carbon revenues and reductions in fossil fuel subsidies (for an overview, see Parry et al. 2014, 2015). The most widely studied co-benefit of carbon mitigation is improved air quality, since the combustion of fossil fuels releases not only greenhouse gases, but also several hazardous co-pollutants, such as particulate matter, sulfur dioxides, and nitrogen oxides. Carbon mitigation consequently can simultaneously provide global climate stabilization, as well as local health benefits, through improved air quality.

In contrast to greenhouse gases, co-pollutants of fossil fuel combustion have direct and short-term impacts on air quality at the location of their release. In fact, air pollution improvements can have such immediate health effects that variation in pollution exposure at the same school over time has been found to significantly affect students' absenteeism (Currie et al. 2009), and variation in outdoor particulate matter exposure around the workplace has been shown to significantly affect workers' productivity (Chang et al. 2016). WHO (2019, p. 15) considers air pollution "the biggest environmental risk to health". Since the environmental health damages of co-pollutants are large, air quality co-benefits could change the incentive structure of countries and regions to contribute to global carbon mitigation, also in the absence of binding international agreements (Parry et al. 2014, 2015; Zwickl et al. 2021).

Given current climate and air pollution challenges, air quality co-benefits of climate mitigation are a straightforward way for policymakers to simultaneously address both problems in Europe. With a 10% share of global greenhouse gas emissions in 2017, the EU-28 is currently the third biggest global carbon emitter, following China and the US. Considering cumulative, historical emissions up to 2017, the EU's share increases to 22% (Ritchie and Roser 2020). At the same time, the EU faces a severe public health crisis due to high levels of air pollution. While air quality has improved over the last decades, especially particulate matter and ground-level ozone levels still exceed the World Health Organization (WHO) safety thresholds in wide parts of Europe. According to the EU air quality report (EEA 2020), 48% of the urban population was exposed to levels above the WHO threshold for PM_{10}, 74% above the $PM_{2.5}$ threshold, and even 99% above the O_3 threshold in 2018. Moreover, the health impacts of specific levels of pollution exposure are continuously corrected upwards, especially as the harmful cardiovascular and respiratory health effects of very fine particulates are better understood. A recent study suggests that air pollution reduces average life expectancy by 2.2 years and accounts for 129 deaths per 100,000 inhabitants in the EU-28 (Lelieveld et al. 2019), which is substantially higher than previous European Environmental Agency (EEA) and WHO estimates.

Air pollution exposure is distributed very unequally, both within and between countries. Globally, low- and middle-income countries face a disproportionately high air pollution burden (WHO 2019). Within countries, air pollution exposure disparities are also widespread. For the US, where regionally disaggregated air pollution as well as socioeconomic data have existed for more than two decades, many studies show that poor and minority neighborhoods are disproportionately exposed to high and unsafe levels of air pollution (for an overview, see Ringquist 2005; Zwickl et al. 2014). For the EU, where geographically fine-scale socioeconomic data do not yet exist, the first evidence from country-case studies (e.g. Rüttenauer 2018 for Germany) and regional EU-wide analyses (e.g. EEA 2018) suggests that air pollution exposure disparities are also substantial.

Moreover, within most countries and regions, air pollution is spatially very concentrated and clustered in pollution hot spots (for the US, see Boyce et al. 2016). For the EU, an assessment of air pollution damages from large industrial point sources regulated under the European Pollutant Release and Transfer Register (E-PRTR) found that the top 147 emitting facilities – roughly the top 1% of facilities included in the registry – contributed to 50% of the total industrial air pollution damage costs (EEA 2014). The large majority of these are thermal power stations and mineral oil and gas refineries; most of them also reported high greenhouse gas emissions (based on own calculations using E-PRTR data). Replacing energy from these sources with renewable ones would generate substantial climate as well as air quality co-benefits. Since many of these facilities are located in proximity to cities in Eastern European countries with already higher pollution levels, pollution abatement at these locations would also narrow regional and socioeconomic environmental disparities. For the US, Boyce and Pastor (2013) have shown that air quality co-benefits are highest in poor and minority neighborhoods. Climate mitigation, especially when targeting pollution hot spots, can thus reduce not only overall air pollution levels, but also disparities in pollution exposure.

Why are greenhouse gases and co-pollutants regulated separately?

Both climate change and air pollution largely stem from the same activity, the combustion of fossil fuels. Yet carbon emissions and local air pollutants are regulated separately. While the role of air quality co-benefits is increasingly emphasized by policymakers, the majority of existing policies are not designed to maximize positive spillovers from one to the other.

Air pollution regulations have been established in many EU countries from the 1970s onwards and have been frequently updated and harmonized ever since. The "Ambient Air Quality Directive" currently in place is binding to its member states. It includes thresholds, limit values, and target values for each air pollutant, air quality plans, and short-term action plans for pollution hot spots, as well as the implementation of national bodies responsible for data collection and compliance. The Ambient Air Quality Directive is supplemented by the National Emissions Ceiling Directive, which sets country limits on air emissions, as well as additional regulations for vehicles and for large combustion plants (EEA 2016, 2020).

The EU's main climate policy is the EU Emissions Trading Scheme (EU ETS). Implemented in 2005, it is the first multinational emissions trading scheme, including all EU member states, as well as Iceland, Liechtenstein, and Norway. The EU ETS is a classical cap-and-trade program, where first an EU-wide cap is set to meet the EU's climate goals, and then the existing permits are allocated across countries, which in turn allocate them to the participating industrial firms and facilities. At the beginning, permits were allocated for free to the historical polluters; over time, the share of auctioned permits and thus also the carbon revenues are increasing (Ellerman et al. 2016; Carl and Fedor 2016). EU ETS installations are free to trade permits across participating countries and sectors. Trading with other emissions trading schemes (such as the Swiss ETS) and options for international credits and offsets are gradually introduced. Moreover, there currently exist binding national annual greenhouse gas emissions targets for sectors not covered under the ETS, as well as binding targets for an increase in the share of renewable energy sources. Some European countries, such as Sweden, additionally have more stringent national climate policies, such as CO_2 taxes on energy use (OECD 2018b).

Two differences between greenhouse gases and local air pollutants have shaped their regulation. The first is related to the spatial distribution of pollution damages. Greenhouse gases are uniformly mixed, implying that one ton of CO_2 has the same climate impact independent of the location of its release. Standard environmental economic models have suggested that pollution mitigation is most efficient where abatement costs are lowest. Since regulators do not know the exact abatement costs of each plant, market-based environmental policies, such as taxes or permit schemes, have been argued to be most efficient (Stavins 2003). Co-pollutants such as particulate matter, sulfur dioxide, and nitrogen oxides, by contrast, are non-uniformly mixed. The total environmental and human health damages strongly depend on the fate and transport of emissions, which include not only emission releases but also factors such as wind and weather patterns and the number of people exposed. Because location matters, for the latter type of pollutants, spatial variations in abatement benefits have to be considered along with abatement costs (Tietenberg 1995; Muller and Mendelsohn 2009; Boyce and Pastor 2013).

The second difference between greenhouse gases and co-pollutants is related to existing pollution control technology, especially the availability of end-of-pipe controls. The only way to currently abate CO_2 is to reduce fossil fuel use or to some extent to switch to other fossil fuels that emit less CO_2, such as natural gas (which however, might only be a temporary solution, and moreover its climate impact might be strongly underestimated since natural gas releases the powerful greenhouse gas methane, see Howarth 2014[2]). With the exception of technologies such as carbon capture and storage, which are currently still very expensive and cannot yet be implemented at large scale, no end-of-pipe technology for greenhouse gases currently exists. For co-pollutants such as nitrogen oxides and sulfur dioxide, by contrast, a variety of end-of-pipe controls are available that can filter out some of the pollutants during combustion. The two most well known are catalytic converters for cars and scrubbers for industrial power plants. Both have been mandated in most EU member countries for several decades. These policies have the distinct advantage that they are easy to implement and monitor. A disadvantage, however, is that they can only abate co-pollutants, while providing no positive spillovers on greenhouse gas mitigation. In fact, spillovers can even turn slightly negative when these pollution abatement technologies require energy, which in turn is generated through more fossil fuel use (Holland 2010).

Historically therefore, taxes, trading schemes, and attempts to achieve binding international agreements have been the main policy tools for greenhouse gas abatement, whereas emissions standards and technology requirements have been used for local pollution control. In the next section, we will discuss potential benefits when regulating the two together.

How large are air quality co-benefits?

Many studies find that air quality co-benefits are high enough to justify more stringent climate policies independent of their climate impacts. Most of these studies are simulation studies that compare a proposed or implemented climate policy to a business-as-usual scenario. They then compute air quality co-benefits per ton of carbon emissions and compare their magnitude to conventional social cost of carbon (SCC) estimates. If the monetized benefits of air quality improvements exceed those of the SCC, these studies conclude that carbon mitigation is beneficial independent of its climate benefits.

For example, Thompson et al. (2014), Driscoll et al. (2015), Shindell et al. (2016), and Buonocore et al. (2016) model different US climate policy scenarios and find that any of these climate policies is associated with air quality co-benefits of such a magnitude that their implementation would be beneficial to the US regardless of climate mitigation goals. West et al. (2013) model the global averted mortality resulting from a global carbon price limiting global temperature increase to 2.5 degrees until 2100 and find that air quality and health co-benefits are high enough to exceed carbon mitigation costs by far. In a global review of co-benefit studies, Nemet et al. (2010) find that co-benefit estimates range from 2–196 USD/ton of carbon with an average of 49 USD/ton of carbon. Moreover, with an average of 81 USD/ton of carbon, co-benefits are substantially higher in developing countries, compared to 44 USD/ton of carbon in developed countries.

Parry et al. (2014, 2015) analyze co-benefits of climate mitigation of the world's top 20 carbon-emitting countries for 2010. While they consider a broader set of co-benefits than most other studies, including reduced traffic accidents and government revenues through reduced fossil fuel subsidies, the largest co-benefits are found to be air quality improvements from reduced coal combustion. The average price across the 20 countries is USD 57.5 per ton of CO_2, which is substantially larger than the US social cost of carbon, estimated to lie at USD 35 in 2013. While there is strong heterogeneity in co-benefits across countries, depending on the fuel mix of the economy as well as existing environmental regulations, implementing 'nationally efficient' CO_2 prices (that correct for all of the negative externalities except for greenhouse gas mitigation) would result in a global reduction of greenhouse gas emissions by 10.8%.

A second important finding from the literature on air quality co-benefits is that they vary across place and across pollution sources. Buonocore et al. (2016) model health benefits of a proposed US environmental policy at the US county level. While they find positive co-benefits for every single US county, their magnitude varies substantially and depends largely on the share of the county's electricity coming from coal as well as the county's air quality prior to the policy. Muller (2012) assesses co-pollutant emissions per ton of carbon emissions for more than 10,000 point sources in the electric power generation and transportation sectors. The largest co-benefits of around 87 USD/ton of carbon (and carbon equivalent greenhouse gas) emissions can be found at bituminous coal-fired electric power generators, followed by other coal-fired power generators. Residual oil-fired electric power generators have co-benefits of around 49 USD/tCO_e, and natural gas-fired electric power generators have substantially lower co-benefits of 3 USD/tCO_e. In the transportation sector, co-benefits range from 11 to 23 USD/tCO_e, depending on the vehicle type (Muller 2012, p. 709f).

Population density also matters for the health impacts of co-pollutants. Muller and Mendelsohn (2009) find for the US that the damages per ton of SO_2 and $PM_{2.5}$ emissions vary by more than 100 times depending on whether the emissions are released in a densely populated city or in a rural area. Boyce and Pastor (2013) calculate co-pollutant intensity ratios (co-pollutants per ton of CO_2 emissions) for 1,540 large industrial point sources in eight industrial sectors and compare inter- and intra-sectoral variation in these ratios. In addition to the three most widely used co-pollutants, SO_2, NO_X, and particulate matter, they include toxic air pollutants from EPA's Risk Screening Environmental Indicators model (RSEI). Moreover, they report results with and without population weighting to illustrate the role of population density on total co-pollutant impacts. They find that ranks in co-pollutant ratios by sector vary strongly depending on the co-pollutant and depending on whether population weighting is applied. For example, although co-pollutants per ton of carbon emissions are higher for power plants than for petroleum refineries, population weighted damages are 3–10 times higher for refineries because they are located in areas with higher population density. Boyce and Pastor (2013) are also the first study to assess co-pollutant impacts by race/ethnicity and the level of income of the neighborhood. For air toxics, which have been found to disproportionately affect people of color (Ash and Fetter 2004; Zwickl et al. 2014), they find that petroleum refineries, chemical manufacturers, and power plants rank highest in disproportionately affecting people of color. Refineries and chemical manufacturers most disparately affect low-income people (Boyce and Pastor 2013). Overall, they find that co-pollutant damages are highest for facilities located in minority and low-income neighborhoods, and conclude that climate policy that incorporates air quality co-benefits could provide substantial equity – in addition to efficiency – improvements.

Finally, the literature discussed so far assesses co-benefits based on a unit elasticity assumption, where a 1% reduction or increase in CO_2 leads to a similar sized change in co-pollutants. However, this unit-elasticity assumption does not need to hold for various reasons. For example, end-of-pipe controls such as scrubbers can reduce co-pollutants, while at the same time requiring electricity. Increasing the combustion temperature in natural gas-fired power plants reduces CO_2 per unit of output but increases NO_X emissions. Since natural gas has lower sulfur content, fuel switching from coal to natural gas reduces both CO_2 and SO_2 emissions. Zwickl et al. (2021) empirically estimate co-pollutant elasticities for industrial point sources in Europe based on data from 2007 to 2015 and document a substantial heterogeneity across economic activities. Looking at climate policy–induced co-pollutant elasticities in the energy-producing sector they find elasticities of 1.2 to 1.8 for SO_X, 1.1 to 1.5 for NO_X, and 0.8 for PM_{10}, suggesting that previous studies might have been underestimating the magnitude of co-benefits. Monetizing the health effects of policy-induced co-pollutant emissions in the energy sector using co-pollutant damage costs from the EEA, they obtain air quality co-benefits of about 60 to 160 EUR per ton of CO_2 (converted into today's EUR), which is substantially higher than EEA or other conventional estimates of climate damage costs.

Summing up, the literature on air quality co-benefits has two main conclusions for EU climate policy, which will be explored in the next section. First, a substantially higher carbon price can be justified independent of its climate benefits. Second, variation in the magnitude of co-benefits across sectors and space suggests that a spatially differentiated carbon policy may provide benefits over a uniform one.

What are implications of including air quality co-benefits in European climate policies?

The literature on air quality co-benefits of climate mitigation has two important implications for current EU climate policy, especially the EU ETS. First, air quality co-benefits justify raising the price of burning fossil fuels substantially. The monetized damage costs of co-pollutants amount to more than 60 EUR/tCO_2 according to the mean estimate from various studies for all sectors in European countries reported by Nemet et al. (2010, Table A.1, converted into current EUR). Zwickl et al. (2021) quantify co-benefits in the European electricity sector to lie at about 60 to 160 EUR/tCO_2 in current euros. While the EU ETS price has been gradually increasing in the past years and has reached an all-time high of 31 EUR for a ton of carbon at the end of 2020 (Shepphard 2020), it is only at about 50% of the level that would internalize the costs of co-pollutants, completely ignoring any climate benefits. To fulfill the climate goals of the Paris Agreement of limiting the global average temperature increase to 1.5–2 °C above pre-industrial levels, a carbon price across countries and sectors of 33–66 EUR (40–80 USD) per ton of CO_2 in 2020 and 41–82 EUR (50–100 USD) per tCO2 by 2030 is required (Carbon Pricing Leadership Coalition 2017).[3] Also, this magnitude tends to be below the monetized co-benefits of improved air quality.

The overall carbon price could not only be increased by raising the price in existing carbon pricing schemes, such as the EU ETS, but also by extending regulatory coverage to previously un- or underpriced sectors. The carbon pricing gap in the EU, the percent of the country's emissions not priced at least 30 (60) EUR per ton of CO_2 in 2015, ranges from 30% (33%) in Luxembourg to 71% (80%) in Estonia (OECD 2018b). The EU ETS covers roughly 45% of the EU's emissions.

Finally, to increase the price of carbon, fossil fuel subsidies, which are still widespread, might be eliminated. In their quantification of global fossil fuel subsidies, Coady et al. (2017, 2019) distinguish between pre-tax and post-tax subsidies. The narrower indicator, pre-tax subsidies, measures the difference between consumer fuel prices and the opportunity cost of fuel supply (mostly applying to petroleum, natural gas, and electricity). Global pre-tax subsidies declined from 0.77% to 0.36% percent of global GDP from 2012 to 2016, which is in line with national and international efforts of fossil fuel subsidy reforms (Coady et al. 2019). While the majority of global pre-tax subsidies are provided in developing and emerging countries, a novel OECD (2018a) database on the Inventory of Support Measures of Fossil Fuels documents that various instruments, such as energy tax reliefs or exceptions for specific sectors or for heating fuels, are still widely in place across EU countries. Post-tax subsidies provide a broader measure, reflecting the difference between consumer fuel prices and prices necessary to internalize environmental costs and meet fiscal revenue requirements like the costs of climate change, local air pollution, broader vehicle externalities, and forgone consumption tax revenues. This broader measure thus also captures air quality co-benefits of carbon mitigation. In contrast to pre-tax subsidies, post-tax subsidies increased from 5.4% to 6.5% of global GDP from 2010 to 2016 (Coady et al. 2019). While post-tax subsidies are also substantially higher in most developing and emerging countries, they amount to relevant economic magnitudes also in European countries (for example, they amounted to 2.1% of GDP in Germany and 1.4% of GDP in France in 2015).

The second important implication of air quality co-benefits for current EU climate policy is that climate policy also has consequences on environmental inequality. As outlined earlier, especially poorer neighborhoods suffer from high air pollution. Evidence from the US further shows that the benefits from cleaner air due to fossil fuel abatement are more pronounced in poor and minority neighborhoods (Boyce and Pastor 2013). Environmental benefits from fossil fuel abatement also vary substantially across locations, activities, and sectors (see also Zwickl et al. 2021 for European evidence). Climate policy thus might be expected to have the largest benefits in poorer areas and areas with air pollution hot spots resulting from industrial activity. Without a consideration of co-benefits, carbon mitigation might not be prioritized in such areas. In fact, California's cap-and-trade program has not been found to reduce environmental disparities in co-pollutant exposure between 2011 and 2015 (Cushing et al. 2018). The latter is not surprising when emissions can be traded freely across locations and sectors, with no incentives for pollution abatement in places with the highest overall environmental damages.

For the EU, currently too little is known about the impact of permit trading in the ETS on co-pollutant emissions, as well as on the role of generating and reinforcing existing pollution hot spots. At the same time, however, various initiatives are set to increase the flexibility of EU ETS trading, such as allowing trading with other emissions trading schemes (such as the Swiss ETS as of 2020) and including carbon crediting mechanisms and offsetting (World Bank 2020). While the latter can bring some important benefits in minimizing the costs of climate policy, granting polluters more flexibility in how and where they want to abate emissions might also increase pollution hot spots. Moreover, sectoral coverage of the EU ETS has been steadily increasing, hereby also increasing flexibility of trading.

Monitoring the distributional effects of carbon trading and co-pollutant impacts is therefore of great importance. If regressive distributional effects and pollution hot spots from emissions trading and emissions offsetting can be identified, various policy instruments can be implemented within or in addition to the EU ETS. Within the ETS, spatially or sectorally differentiated pricing, trading zones, or trading ratios could be implemented if high co-pollutant costs can be found in specific locations or sectors (Boyce and Pastor 2013). Moreover, revenues from permit auctions could be used to specifically address co-pollutant damages and accelerate fossil fuel abatement in places with the highest overall environmental damages (e.g. coal-fired power plants in densely populated areas) or investment in renewable energy sources. Besides the ETS, conventional regulations, such as stricter air quality ceilings, can also target the adverse effects of co-pollutants and pollution hot spots.

The current discussion on whether climate policy hurts the poor neglects an important dimension if co-benefits are omitted. This discussion centers on the regressive effects of rising carbon prices through consumption effects (since low-income households spend a larger share of their incomes on energy, especially for transportation and heating), broader macroeconomic effects (such as a shift in factor prices and sectoral employment shifts), and the role of redistributing carbon revenues in ways to avoid harming the poor (Rausch et al. 2011; Boyce 2018; Metcalf 2019). Since climate policy tends to reduce co-pollutant exposure and increase air quality co-benefits especially in poorer neighborhoods, co-benefits provide an additional mechanism through which climate policy can benefit the poor.

Conclusion

Framing climate policy as a distributional conflict across countries and generations overlooks important reasons for carbon mitigation here and now. Fossil fuel combustion emits not only carbon, but also hazardous co-pollutants with large direct and immediate environmental and health effects. These air quality co-benefits of carbon mitigation are high enough that carbon mitigation should be "in a country's own interest" (Parry et al. 2015).

Considering air quality co-benefits in climate policy in the EU, which currently has the world's largest and only multinational emissions trading scheme, could justify a substantially higher carbon price, which also would increase the chances of meeting the Paris goals of limiting global temperature increase to 1.5–2 °C compared to pre-industrial levels.

Moreover, considering air quality co-benefits also adds a new dimension to debates on the distributional effects of climate policy, since socioeconomically disadvantaged neighborhoods are usually more exposed to co-pollutants, both within and across countries, and benefit more from air quality improvements caused by fossil fuel abatement. Variations in the magnitude of co-benefits across sectors, activities, and locations suggest that co-pollutant damages should be carefully monitored, especially as the European carbon market is getting more and more flexible due to an increase in the geographical and sectoral coverage, as well as new laws that allow carbon crediting mechanisms such as offsetting. The more flexible the market gets, the more likely it is that hot spots of co-pollutants are generated as a side effect. The latter, however, could be addressed by a variety of measures, including differentiated pricing, supplementary regulatory measures, and carbon revenue use that offsets adverse co-pollutant impacts.

Since high levels of co-pollutants and pollution hot spots tend to be disproportionately located in socioeconomically vulnerable areas, neglecting the role of air quality co-benefits while increasing the flexibility of trading in existing carbon pricing schemes could widen socioeconomic environmental and health disparities. Monitoring co-pollutant impacts of carbon trading and considering air quality co-benefits will therefore be of increasing importance to narrow environmental inequality.

Notes

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2 While natural gas releases substantially less carbon dioxide than coal or oil, it also releases another greenhouse gas, methane. Compared to carbon dioxide, methane is a far more effective greenhouse gas, but it also stays in the atmosphere for a shorter time span (12 years, compared to a century or more for CO_2). As a consequence, a comparison of the climate effect between the two has to consider a specific period. When choosing a short period, such as 20 years, which is the period currently most relevant to avert the most severe damages from climate change, and when considering the full life-cycle of greenhouse gas emissions of natural gas versus coal or oil (from mining/drilling to its end use), natural gas is not found to have a lower greenhouse gas impact (Howarth 2014).

3 The High-Level Commission on Carbon Prices emphasizes, however, that prices might have to be substantially lower in developing countries, implying conversely that they have to be higher in the EU.

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