Multiple benefits from climate change mitigation: assessing the evidence

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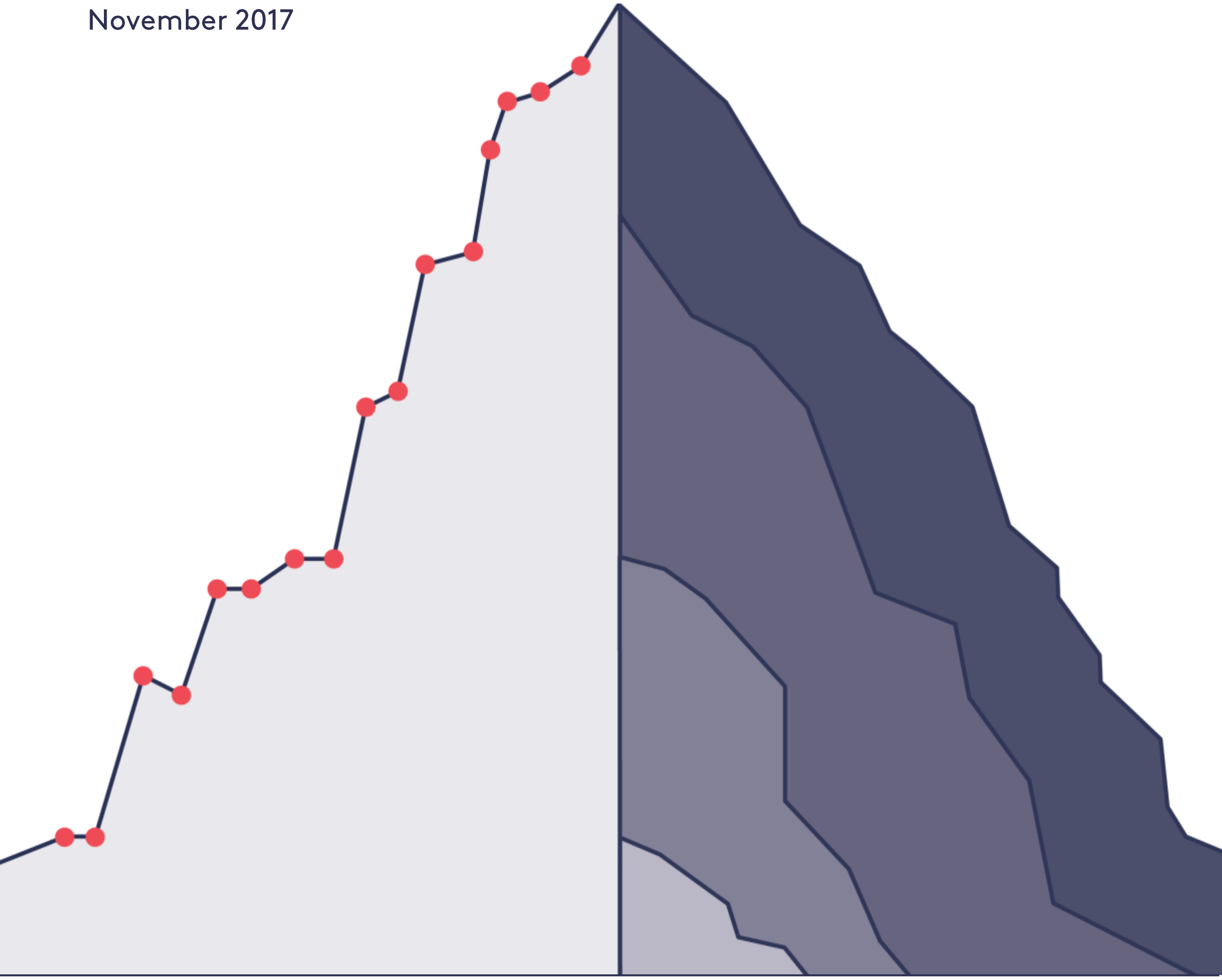

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Executive summary

Climate change mitigation yields multiple benefits – or 'co-benefits' – in addition to the reduction of greenhouse gas emissions. Many co-benefits, such as diminished local air pollution resulting from reduced use of fossil fuels, occur in the short run, are relatively certain to be achieved, and are primarily enjoyed by the country doing the abatement. This stands in stark contrast to the benefits of climate change mitigation, which are longer term, less certain, and accrue largely to *other* countries. As the world proceeds with the implementation of the Paris Agreement on climate change, a focus on the multiple benefits of mitigation can motivate a more ambitious approach to reducing greenhouse gas emissions.

Abating greenhouse gas emissions reduces harm to health caused by local and regional air pollution

Reductions in health damages from local and regional air pollution are by far the largest category of co-benefit arising from abating greenhouse gas emissions. Based on figures from the World Health Organization Global Burden of Disease, damages from ambient PM2.5 exposure (particles less than 2.5 microns in size) in 2010 were estimated to be over 9 per cent of GDP in China, over 5 per cent in India, and over 3 per cent in the United States. Per tonne of carbon dioxide equivalent (tCO_2e), these damages amounted to nearly US$70 in China, US$50 in India, and roughly US$200 in EU countries. Comparing these figures with the US government estimate of the social cost of carbon of US$36/$tCO_2e$ in 2015, local air pollution co-benefits can conceivably be larger than the costs of abating CO_2 in most major emitting countries.[1]

Co-benefits only occur if there are pre-existing distortions and inefficiencies in economies. Local and regional air pollution is the preeminent example: government policies generally fail to reduce emissions to the efficient level, where marginal costs and benefits are equated. At the same time, the causal flow regarding co-benefits can go in different directions. Investments in urban transport infrastructure, for example, can reduce greenhouse gas emissions. But these investments are likely to be driven by the large benefits available from reduced congestion and accident fatalities – in this instance climate change mitigation becomes a co-benefit of efforts to increase local wellbeing.

Estimates of the benefits of reducing PM2.5 exposure are based on an increasingly robust combination of epidemiology (relating exposure to health outcomes), physical modelling of pollutant dispersal, and economic analysis of the dollar value of reduced mortality. This progress in benefits estimation is being translated into integrated assessment models that can simulate climate mitigation over the course of this century.

There are variations in the control of local air pollutants and in the evidence for economy-wide benefits from climate change mitigation

The electric generation sector is a major source of both CO_2 emissions and emissions of local air pollutants, including PM2.5 and its precursors. The evidence across countries is that the degree of control of local air pollutants varies widely, being particularly weak in many large middle-income countries. Even in the US, however, there is huge variation in emission factors across generating facilities, with high emissions concentrated in relatively few generators. The degree of control of

1 World Bank (*The cost of air pollution: strengthening the economic case for action,* 2016) updates the data and methodology of air pollution damage valuation and provides recent cross-country estimates. The figures closely match the estimates reported in this document.

local air pollutants from electric power generation has a fairly direct effect on the potential for co-benefits from CO_2 mitigation.

The evidence for economy-wide benefits from climate change mitigation is mixed. Carbon taxes may incur a co-cost rather than a co-benefit, owing to tax interaction effects. Recent evidence[2] suggests, however, that recycling carbon tax revenues through reductions in labour taxes can yield a net positive tax interaction, increasing the overall efficiency of the tax system. Costs of climate mitigation can affect competitiveness, but the aggregate effects are likely to be small; some energy-intensive trade-exposed sectors, however, may need compensating policies. 'Green' job creation may or not be a net benefit, given that 'dirty' jobs will have to decline as mitigation is undertaken. Active labour market policies will be required as the energy transformation proceeds. While flexible climate policy instruments, such as taxes or tradable permits, do induce innovation, there is little evidence of a net GDP benefit from green innovation (the Porter Hypothesis). Investments in indigenous green energy (wind, solar, biofuels) may reduce reliance on fossil fuel imports which are subject to price shocks, but some types of green energy may decrease energy security to the extent that intermittency of supply is an issue.

Energy efficiency measures and actions to reduce short-lived climate pollutants are producing health- and food-related co-benefits

Increasing energy efficiency will be a major part of the energy transition and there is evidence that increased efficiency has reduced dependence on fossil fuel imports, as well as model results estimating the GDP growth resulting from increased efficiency. In addition, increased energy efficiency reduces the need for costly abatement of local air pollution. Viewed as a type of zero-emission energy supply, energy efficiency can also contribute to reductions in the health damages from local air pollution.

Actions to reduce emissions of short-lived climate pollutants (SLCPs), primarily methane and black carbon, will likely be driven by the potential for local benefits. Model results (Shindell et al. 2012) for 2030 suggest that health benefits from reduced ozone and PM2.5 exposure (a result of reducing black carbon emissions by 75 per cent and methane by 40 per cent relative to business as usual) could be as large as 5 per cent of global GDP, while many countries could also enjoy substantial increases in crop yields. The climate co-benefit of this degree of control of SLCP emissions could be a reduction of close to 0.5°C in projected warming in 2030.

The health co-benefits of CO_2 mitigation could be large

A series of model-based analyses of co-benefits examine the future potential for health benefits to offset the costs of climate change mitigation. A study of power sector interventions in India in 2030[3] suggests that the net costs of mitigating 80 per cent of CO_2 emissions could be negative. Two studies[4] compare broad climate mitigation scenarios to business-as-usual scenarios, measuring the difference in air pollution mortality. For targeted warming of 2.4°C the co-benefits of mitigation (measured very conservatively) exceed US$100 per tonne of CO_2 abated in selected high-income countries, compared with roughly US$50/t$CO_2$ in middle-income countries. For a 2°C warming target a second study estimates health co-benefits for the EU at over US$200/t$CO_2$

[2] Bento et al. (2013) 'Environmental Policy in the Presence of an Informal Sector', http://works.bepress.com/antonio_bento/25

[3] Markandya et al. (2009) 'Public health benefits of strategies to reduce greenhouse-gas emissions: low-carbon electricity generation' *The Lancet* 374

[4] Shindell et al. (2012) 'Simultaneously Mitigating Near-Term Climate Change and Improving Human Health and Food Security', *Science* 335; and West et al. (2013) 'Co-benefits of mitigating global greenhouse gas emissions for future air quality and human health', *Nature Climate Change* 3

abated in 2030, compared with nearly $500/tCO_2$ (using similar parameters for the calculation of benefits, and less conservative assumptions) for the 2.4°C scenario.[5]

Considering both the value of air pollution mortality per tonne of CO_2 emitted in 2010 and the model results for 2030, there is strong evidence to suggest that the health co-benefits of CO_2 mitigation could be large, conservatively estimated at over US$100/t$CO_2$ abated in high-income countries and US$50/t$CO_2$ abated in middle-income countries. The ranking between developed and developing countries is driven by the relative carbon inefficiency observed in developing countries. The relevant comparator for these figures is the carbon price (which puts a cap on marginal abatement costs), US$36/t$CO_2$e in the US in 2015, and rising to US$85 in the 2.4°C scenario analysed here.

These significant health benefits from reducing air pollution raise the question of whether countries should simply implement end-of-pipe pollution controls – an approach that cleans up contaminated flows of water or air at the point at which the pollution enters the environment – and set aside the climate policy question for later. This would not be the best policy approach, for two reasons. First, zero-emission technologies such as wind or solar have a structural cost advantage compared with fossil fuels with end-of-pipe controls – they avoid both pollution mortality and the need for pollution controls. The question of whether to 'abate or go green' needs to take this into account. Second, standard economic theory suggests that in the face of two externalities – local pollution and greenhouse gases in this case – the optimal policy is to use separate instruments for each, e.g. a local pollution tax and a carbon tax. At least one model analysis[6] (Bollen et al., 2009) shows that this could indeed produce higher global welfare over the course of this century than would individual policy interventions – either control local air pollution only, or control greenhouse gas emissions only.

Policy decisions should be made on the basis of social cost–benefit analysis

The rational basis for making policy decisions on local air pollution and greenhouse gas abatement is a social cost–benefit analysis (CBA), where costs include economic costs such as the health damages from local air pollution externalities. CBA has limitations for analysing optimal climate policy because of deep uncertainties about catastrophic outcomes at high greenhouse gas concentration levels. But it can be an appropriate tool for analysing co-benefits given their local, and more certain, characteristics.

More generally, social cost–benefit analysis needs to be embedded in a reformed public investment management system which covers the full project cycle from economic appraisal to post-implementation assessment of outcomes. Stronger public investment management systems offer permanent increases in growth. Here, however, the relevant increase is the growth in wellbeing, not necessarily growth in gross output. Similarly, governments need to set incentives for the private sector that decrease greenhouse gas emissions and externalities such as air pollution mortality, and therefore increase wellbeing.

Governments have taken a blinkered approach to climate policy, focusing on the financial costs of greenhouse gas abatement and worrying about the global public good nature of any benefits. By including the economic costs of local air pollution damages in the policy analysis, there is evidence that health co-benefits can change this dynamic. Certainly co-benefits need to be part of any honest appraisal of climate policy options. In making this argument, however, there is a risk

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5 In a more stringent climate scenario (compared with one that is less stringent), less fossil fuel is being burned relative to business as usual (BAU), leading to lower health damages, also relative to BAU, while CO_2 abatement relative to BAU is higher.

6 Bollen et al. (2009) 'Local air pollution and global climate change: A combined cost-benefit analysis.' *Resource and Energy Economics* 31

that countries focused on local benefits will feel less urgency about achieving a cooperative solution to the climate problem. From the wider perspective this would obviously be an error, because only collective action can solve the climate problem. Ensuring that the poorest countries in the world are sheltered, both from harmful climate change (through adaptation) and from the high costs of adapting and abating, must also be part of any collective action.

1. Introduction: Why focus on co-benefits?

Managing climate change is by far the greatest collective action problem the world has faced. No single country can solve the problem.

The Paris Agreement on climate change, adopted in December 2015 and entering into force in November 2016, has given renewed impetus to international efforts to mitigate climate change. It aims for the increase in global average temperature to remain well below 2°C above pre-industrial levels, with a commitment to pursue efforts to limit the increase to 1.5 °C above pre-industrial levels, recognising that this would significantly reduce the risks and impacts of climate change (United Nations, 2015).

While the Paris Agreement does not contain commitment targets with legal force, it represents the first time that virtually all countries, both developed and developing, have agreed to take actions to reduce the risk of climate change. The main mechanism for these actions is through nationally determined contributions (NDCs) for each signatory, which specify emission reduction benchmarks, targets and timetables. Collective progress on achieving the goals of the Agreement will be subject to five-yearly stock-taking, starting in 2023.

The costs and benefits of acting on climate will be very unequally distributed across countries. But governments need to recognise that many actions, particularly on climate mitigation, can have local benefits that change the cost–benefit calculus for acting on climate. Collectively, these local benefits can be termed the 'multiple benefits' or 'co-benefits' of climate action.

Co-benefits can be loosely defined as the ancillary or secondary benefits resulting from a given policy. In the context of climate change mitigation, they are a joint product of actions to reduce emissions of greenhouse gases. The classic example of a co-benefit is the reduction in local air pollution associated with climate change mitigation policies that reduce the use of fossil fuels.

While the literature in this field is largely focused on co-benefits, there are also trade-offs or co-costs for certain measures. One that has attracted much attention is an increase in emissions of fine particles that are damaging to health associated with increased combustion of biomass (as a 'greener' fuel). A full assessment of the impacts of climate change mitigation measures reveals a variety of other trade-offs. Consider, for example, options for replacement of coal-fired power generation. All likely options will lead to a significant reduction not only in greenhouse gas emissions, but also in emissions of other air pollutants, and in discharges of liquid effluent. However, the alternatives have their own impacts. Nuclear generation produces significant quantities of hazardous waste, there are risks in the extraction of natural gas, wind turbines may intrude on scenic landscapes, and so on. A knowledge of these trade-offs is clearly essential in order that they be mitigated or avoided. Fortunately, analysis indicates that these co-costs tend to be less significant than the co-benefits (see Forster et al., 2013 and chapter 3 below).

1.1 Benefits for policymakers and countries concerned about climate change

For policymakers dealing with climate change, the existence of local co-benefits has several attractive characteristics. A key concern for policymakers is that climate interventions only produce benefits over a long time horizon, a period of decades, and the size of these benefits is uncertain. In addition, the climate benefits produced by a country's own actions will largely accrue to other countries. In contrast, air pollution co-benefits are enjoyed in the near term when cleaner low-carbon energy sources are substituted for dirty ones, the benefits are large, they accrue mostly to the country itself and they are highly certain compared with those of climate change interventions (although the uncertainties in local air pollution damage estimation also need to be acknowledged). Co-benefits reduce the net cost of mitigating greenhouse gas emissions.[7]

There are other reasons why a focus on co-benefits may be attractive for countries concerned about climate change. These include:

- As the Paris Agreement shows, moving on climate change may require countries to act, independently or in coalition, to deal with climate change. *Taking a much broader view of benefits and costs can help countries establish which climate actions are in their own interest.*
- Countries are focused on the financial costs of climate mitigation, and assume (often rightly, in the case of small countries) that the *direct* benefits they will enjoy as a result of abating their own carbon emissions will be negligible. Local co-benefits can therefore help to motivate mitigation actions.

However, it will be important to ensure that individual country actions on climate, motivated by the multiple local benefits that can accrue to these efforts, do not undermine the case for the Paris Agreement. The only 'solution' to the climate problem is through collective action.

To motivate a focus on multiple benefits, Table 1.1 presents median deaths from ambient PM2.5 exposure from the WHO's Global Burden of Disease (GBD) 2010 (Lim et al., 2012), as well as a range of estimates (based on the 95 per cent confidence interval for deaths in GBD 2010) of the value of these deaths in ratio to GDP.[8] The countries selected are the 15 largest emitters of CO_2 in 2010.

Table 1.1. Value of mortality from ambient PM2.5 exposure, percentage of GDP, 2010

	CO_2 emissions (million tonnes)	Deaths	% GDP
China	8,287	1,233,890	9.7–13.2
United States	5,433	103,027	3.2–4.6
India	2,009	627,426	5.5–7.5
Russia	1,741	94,558	6.9–9.8
Japan	1,171	64,196	4.9–7.7
Germany	745	41,582	5.1–7.3
Iran	572	32,288	4.7–6.2
South Korea	568	23,036	4.6–7.1
Canada	499	7,171	2.0–3.2
United Kingdom	494	23,373	3.7–5.5
Saudi Arabia	464	8,550	3.4–4.4
South Africa	460	3,208	0.6–1.0
Mexico	444	20,496	1.9–2.5
Indonesia	434	63,826	2.8–3.9
Brazil	420	7,582	0.3–0.7

Sources: Lim et al. (2012); Hamilton (2014)

If no change in 'end-of-pipe' controls (cleaning up at the point at which the contaminants enter the environment) on PM emissions is assumed, then this table provides evidence of the potential

[7] This reduction in net costs of abatement has two consequences in the context of a global solution to the climate problem. First, the globally efficient level of a carbon tax in any given country would have to rise to reflect co-benefits. Second, the reduction in net costs of mitigation weakens the bargaining position of countries arguing for a mitigation subsidy.

[8] Note that GDP is used as a numeraire in many of the tables in this study. Reported percentages of GDP should therefore not be construed as lost production.

for air pollution co-benefits in the big emitters. The largest percentages are in the emerging economies, with Brazil being the notable exception owing to low fossil fuel dependence. While a proportion of these deaths are not preventable, given that they are the result of lagged or cumulative exposure in the past, reductions in fossil fuel use in 2010 could similarly prevent future deaths from lagged or cumulative exposure (although these would need to be discounted in any analysis of policies in 2010).

Table 1.1 presents indirect evidence to suggest the potential for co-benefits from reducing fossil fuel use is large in many of the large CO_2 emitters. Table 1.2 speaks more directly to this potential.

Table 1.2. PM2.5 mortality per tonne CO_2 emitted and carbon efficiency, 2010

	PM2.5 mortality (US$/tonne CO_2 emitted)	Carbon efficiency (GDP/tonne CO_2 emitted)
Brazil	17–34	5,106
Canada	64–103	3,234
China	69–94	716
Germany	228–325	4,433
India	47–64	850
Indonesia	46–63	1,634
Iran	35–45	739
Japan	232–363	4,694
South Korea	88–137	1,928
Mexico	45–59	2,370
Russian Fed.	60–86	876
Saudi Arabia	38–50	1,134
South Africa	5–8	794
United Kingdom	174–258	4,651
United States	87–126	2,753

Source: Hamilton (2014)

Table 1.2 shows the value of PM2.5 mortality per tonne of CO_2 emitted for the large emitters, as well as an explanatory indicator, carbon efficiency measured as GDP generated per tonne of CO_2 emitted. While the mortality figures per tonne of CO_2 are averages, they may not differ too greatly from the marginal benefits of policies to reduce fossil fuel use if an instrument such as a carbon tax is used in policy implementation. This is because a carbon tax will not create incentives to introduce end-of-pipe measures[9] to reduce emissions of PM2.5.

A useful comparator for Table 1.2 is the latest US estimate of the social cost of carbon in 2015, US$36/t$CO_2$ (in 2007 dollars) (US Government Interagency Working Group, 2016). If a carbon tax at this level were administered globally, this would result in marginal abatement costs equalising at the same level. Average abatement costs would be lower, however, so the average figures on the value of PM2.5 mortality per tonne of CO_2 emitted are larger than average CO_2 abatement costs in most countries. In other words, co-benefits exceed abatement costs, making such a carbon tax an attractive policy purely in terms of local costs and benefits.

The general trend evident in Table 1.2 is that the potential co-benefits of reducing fossil fuel use are substantially higher in rich countries compared with emerging economies. This is largely driven

[9] Note that end-of-pipe measures will typically reduce PM emissions but have no impact on CO_2 emissions. The extent to which marginal and average values of PM2.5 mortality per tonne of CO_2 emitted are similar will, however, be affected by non-linearities in the concentration response function (discussed in Section 3.1 below).

by carbon efficiency, which is high in rich countries (Canada and the US excepted), and low in emerging countries (Brazil excepted).

The potential for wider benefits and costs associated with climate change mitigation immediately raises questions about the use of cost–benefit analysis (CBA). The quality of public investment management (PIM) systems in many developing countries is low, with the application of cost–benefit analysis being particularly weak. As a result, economic analysis of co-benefits may currently not be a factor in project choice, and countries may be foregoing significant local benefits from climate action. Cost–benefit analysis has limitations as a tool to define optimal climate policies for the world, owing in particular to the deep uncertainties about the climate system and its response to increased forcing from greenhouse gases. As a result, many analyses of global climate policy have opted for a risk-management approach, and the analysis of adaptation investments often considers robust approaches (as opposed to optimal approaches) in the face of climate uncertainty. But co-benefits are local and more certain in their effects, making cost–benefit analysis the preferred tool for analysis of co-benefits and co-costs.

1.2 Key questions and structure of the paper

This paper considers a number of questions about the co-benefits associated with action on climate change:

- How wide is the scope of co-benefits of greenhouse gas abatement? Examples could include environmental benefits, public health benefits, increased efficiency, induced technical change, employment, economic growth, and energy security.
- Does the causality also flow the other way? To what extent are there climate co-benefits derived from other development policies and projects? For example, are there local air pollution initiatives that also yield climate benefits?
- What are the key sectors where co-benefits can be realised? Focusing on sectors provides a direct link to policy interventions. Possibilities include the energy sector (including fuel switching, energy efficiency, and new green technologies), the transport sector, and agriculture and forest sector interventions.
- How solid is the analytical base for assessing the extent of co-benefits? Are there widely agreed analytical approaches and a substantial literature?
- How large are the co-benefits of climate actions likely to be, and which are the dominant benefits? Are co-benefits characteristic of particular categories of countries?
- What actions will foster the use of co-benefits estimation in the analysis of climate interventions?

Chapter 2 of the paper takes a closer look at the concepts and definitions relating to the co-benefit question. Following this, Chapter 3, the bulk of the study, assesses what is known about potential co-benefits across multiple domains: environmental, economy-wide, and sector-specific. Chapter 4 reports on empirical results on co-benefits, in particular the application of integrated assessment models (IAMs) to simulate co-benefits over the course of the century. Chapter 5 analyses a key policy question – if governments aim to reduce local air pollution damages, are there greater benefits from using traditional pollution control measures or moving to greener zero-emission energy sources? Chapter 6 deals with creating co-benefits through strengthening public investment management. The final chapter concludes and considers policy reforms that can foster the application of co-benefits analysis to development decisions.

Finally, a caveat concerning the assessment of co-benefits in Chapters 3 and 4. Box 1.1 summarises the comparative risk assessment of ambient exposure to fine particulate matter in air (PM2.5) carried out in GBD 2010 and GBD 2015. It is fair to say that this is the most comprehensive effort to date to arrive at estimates of deaths attributed to outdoor PM2.5 exposure, and the numbers are substantially larger than previous estimates. This is linked to improvements in both data and analysis of the data. The underlying methodology for the assessment of mortality risks was only published in 2014 (Burnett et al., 2014). As Box 1.1 highlights, GBD 2010, GBD 2015 and World Bank (2016) together provide the latest estimates of the welfare value of air pollution damages, country by country. But the great majority of previous estimates of air pollution mortality are out-of-date and biased downwards. This will be pointed out below in the assessment of evidence for co-benefits, but it is important to draw attention to this at the outset.

Box 1.1. The WHO Global Burden of Disease 2010 (GBD 2010) and updates to 2013 and 2015

GBD 2010 was a major step forward in the measurement of mortality from disease and injury worldwide. In addition to measuring outcomes, however, the study also carried out a comparative risk assessment in order to attribute these outcomes to 67 different risk factors. A key cluster of risk factors looked at mortality and disability-adjusted life years associated with exposure to ambient PM2.5 (particles in air less than 2.5 microns in size), indoor air pollution and ozone.

As described in Lim et al. (2012), the GBD 2010 exercise estimated ambient PM2.5 exposure worldwide on a 0.1x0.1 degree grid by combining model results on chemical transport in the atmosphere and satellite imagery on optical depth, and then building a regression model based on these data and ground observations of PM2.5 concentrations. When overlaid with population, this permits estimation of deaths from PM2.5 exposure using an integrated concentration-response function that is estimated using data on outdoor air pollution, indoor air pollution, second-hand tobacco smoke, and active smoking (Burnett et al., 2014). To test sensitivity, the estimates were simulated 1,000 times at the level of individual age, sex, country and year, resulting in median, 2.5 percentile and 97.5 percentile estimates of global deaths linked to PM2.5 exposure.

Global mortality from ambient PM2.5 exposure in 2010 was estimated to be 3.2 million (95% CI ±12.2%).

More recently, Cohen et al. (2017) published the detailed results on PM2.5 mortality prepared for the WHO Global Burden of Disease 2015. They arrive at a central estimate of global mortality from PM2.5 exposure in 2015 of 4.2 million deaths, and note that this very substantial increase on 2010 is due to population ageing, changing non-communicable disease rates, and increasing air pollution in low- and middle-income countries. In future updates of the estimates of the co-benefits of climate change mitigation, therefore, we can expect to see much larger values of the co-benefits from reductions in outdoor air pollution.

In addition, World Bank (2016) has published new estimates of the value of ambient PM2.5 mortality for 2013 using methodologies very similar to this report. The work is based on GBD 2013, where improvements in data and methodology resulted in a central global estimate of 2.8 million deaths from ambient PM2.5 exposure. The World Bank country estimates of the welfare value of PM2.5 mortality as a percentage of GDP for 2013 are comparable with the lower-bound values reported in Table 1.1.

2. Concepts and definitions

Co-benefits (and co-costs) can be defined as the ancillary or secondary benefits resulting from a given policy. If reducing carbon dioxide emissions is the primary purpose of a policy, then any associated reduction in local air pollution can be viewed as a co-benefit. Similarly, reducing emissions of **short-lived climate pollutants (SLCP),** which have a relatively short lifetime in the atmosphere, from a few days to a few decades, has a relatively direct co-benefit because a principal SLCP is black carbon, which in turn is a component of PM2.5, a major risk to health.

More formally, co-benefits (and co-costs) are the **externalities** associated with a given policy intervention. In an optimal economy all externalities would be internalised in the process of welfare maximisation (Hallegatte et al., 2012; Fullerton and Metcalf, 2001) – local pollution, in particular, would be reduced to the point where the marginal costs and benefits of pollution abatement are equated. The existence of pollution co-benefits is a reflection of living in a second-best world.

In a second-best world, a comprehensive approach to **cost–benefit analysis (CBA)** of projects and policies should include co-costs and co-benefits, and would often, therefore, result in different policy decisions. Limited or poor-quality CBA ignores co-benefits.

In a project cost–benefit setting, co-benefits are only realised when the alternative project is harmful, e.g. a source of pollution. Similarly, in scenario analysis the **'green scenario'** only yields co-benefits to the extent that there is a reduction in damages (e.g. from local air pollution) or an increase in benefits compared with the **business-as-usual scenario**. And energy efficiency projects must be assumed to have economic benefits that exceed economic costs before any claims can be made for the existence of co-benefits associated with increasing energy efficiency. In addition, co-benefits need to be measured relative to any existing policies (such as fuel taxes) that reduce emissions of both local pollutants and CO_2.

A key conceptual issue concerns the **opposite causal flow** – are there local investments that can yield climate benefits? While in principle the answer to this question is 'yes', it may be that the local project with the best benefit–cost ratio yields no climate benefits – e.g. end-of-pipe treatments for emissions of pollution, rather than switching to a cleaner fuel. More generally, efficient solutions to market failures require individual policy instruments to deal with individual failures – a single multi-purpose instrument is unlikely to be optimal.

By the same logic, interventions to reduce greenhouse gas emissions may not yield the socially optimal level of local pollution emissions. But the inclusion of co-benefits in the analysis of greenhouse gas-reducing projects offers the prospect of valuing the generation of local benefits, and project design decisions may increase these co-benefits.

While CBA is typically applied to projects in the public sector, governments will choose to mitigate greenhouse gas emissions by private actors as well. This involves a choice of public policies to set incentives for the private sector: regulations, market-based instruments, or information-based instruments such as eco-labelling. In this setting it may still be possible to analyse any co-benefits of the chosen policy, and governments can aim for policies that incur the lowest private costs and generate the greatest social benefits in the form of co-benefits.

With respect to **policy instruments**, the central tenet of environmental economics is that economic instruments (taxes or tradable quotas) are the most efficient instruments for pollution control and provide strong incentives for innovation. This is backed both by theory and a large body of empirical analysis. However, there are situations where economic instruments are not practical or viable – examples include emission of highly toxic substances, prohibitive monitoring costs (e.g. taxing end-of-pipe emissions from individual vehicles), or agency problems (e.g. the landlord who does not pay the heating bill, and therefore has no incentive to install an efficient furnace). In some instances, therefore, other instruments such as technical standards may be practical policy tools, even if they entail higher costs than the theoretically perfect instrument. But policymakers need to be concerned about the interaction between instruments such as a **cap-and-trade** system and policies such as **feed-in tariffs** (FIT), for example. If the electric power system is subject to cap-and-trade, the additional emission reductions associated with the FIT will simply lead to more emissions by other electricity producers – that is, the emissions cap that is binding (OECD, 2011).

To deal with this problem, in California the California Air Resources Board has the power to adjust CO_2 emissions allowances to account for the effects of other, more specific, policies such as its FIT programme. As a general rule, policymakers should calculate the implicit price of CO_2 emissions reductions under the various programmes aimed at reducing emissions, but this can be complex when multiple instruments interact.

Finally, while many climate mitigation efforts will be led by central governments, the experience to date suggests that cities and sub-national governments have been very active in promoting climate actions. While this represents an important commitment to climate action, it is also true that many of the co-benefits of climate action will be local – many cities and locales view climate mitigation to be in their self-interest. However, governments will need to be alert to any potential inconsistencies between national and local policies.

3. Assessing potential co-benefits and co-costs

The major co-benefits of climate action include:

- Reduced air pollution damages to health. There may also be distributional benefits associated with reduced pollution, since poor people are often the most exposed to air pollution
- Increased labour productivity linked to reduced air pollution exposure
- Increased enjoyment of environmental amenities linked to reduced pollution emissions
- Increased ecosystem service provision by conserving and increasing forest cover linked to reduced emissions from deforestation and forest degradation (REDD) and selected adaptation investments
- Reduced crop damage linked to ground-level ozone
- Rising real incomes and increased competitiveness generated by efficiency gains, particularly energy efficiency
- Benefits generated by green innovation
- Increased energy security
- Potential for 'green jobs'
- Decreased congestion costs from transport sector investments, including air pollution damages (while stuck in traffic), energy costs, the opportunity cost of time, and lost productivity; improved health from active transport (walking and cycling may also be significant)
- Agricultural productivity increases from conserving and building soil carbon
- Improved weather and land use information generated by monitoring systems for climate adaptation

Co-costs include:

- Negative impacts on the poor of rising food prices driven by biofuels production (although this is not technically an externality). In developing countries poor households are generally net buyers, rather than producers, of food
- Loss of environmental services from land clearance to produce biofuels
- Increased PM2.5 emissions from biofuels, and associated negative health impacts
- Loss of competitiveness in energy-intensive trade-exposed sectors
- Tax interaction effects of carbon taxes, which can decrease the efficiency of the overall tax system

As these lists suggest, many of the co-benefits are linked to environmental quality. A second cluster concerns economy-wide benefits and costs (efficiency, competitiveness, energy security, green jobs). The analysis that follows will show that multiple sector interventions and multiple pathways can affect each of these main co-benefits.

There is a strong argument for classifying co-benefits according to the sectoral interventions that lead to co-benefits. This would maintain a close link to policy actions that decision-makers can implement in individual sectors. But most sectoral interventions have connections to both environmental and economy-wide outcomes, in addition to idiosyncratic benefits particular to the sector (for example, the benefits of reducing congestion in the transport sector). This suggests that it is worth assessing the two cross-cutting themes – environmental and economy-wide impacts – separately, before turning to sectoral interventions.

3.1 Environmental benefits of climate change mitigation

Environmental co-benefits are a major component of the co-benefits of climate action. This is particularly the case for mitigation efforts that reduce the combustion of fossil fuels, which means that mitigation actions in multiple domains (electricity, transport, energy efficiency, buildings) will have an impact on environmental quality. The backdrop to this analysis of co-benefits is the widespread deterioration of environmental quality in middle-income countries. Table 3.1.1 shows the value of degradation in a variety of countries in the Middle East and North Africa (MENA) region.

Table 3.1.1. Cost of environmental degradation, selected MENA countries, % of GDP

	Water	**Land**	**Air**	**Waste**	**Coast**	**Total**
Algeria (1998)	0.8	1.2	1.0	0.1	0.6	3.7
Egypt (1999)	1.0	1.2	2.1	0.2	0.3	4.8
Iran (2002)	2.82	2.5	1.6	0.36	0.15	7.43
Iraq (2008)	3.5	1.0	1.5	0.4	0.01	6.41
Jordan (2006)	0.81	0.11	1.15	0.23	0.06	2.36
Lebanon (2000)	1.07	0.6	1.02	0.05	0.68	3.42
Morocco (2000)	1.23	0.44	1.03	0.49	0.52	3.71
Syria (2001)	0.9	1.0	1.3	0.1	0.1	3.4
Tunisia (1999)	0.61	0.52	0.58	0.13	0.26	2.1

Source: Croitoru and Sarraf (2010)

While heavy industrialisation and motorisation in middle-income countries, combined with limited environmental controls, is the major factor driving the figures seen in Table 3.1.1, a major study by the National Research Council (2010) in the United States also estimates significant damages from fossil fuel production, refining and use in the US, as seen in Table 3.1.2.

Table 3.1.2. External damages from fuel use in the United States, 2005

Fuel use by sector	**US$ bn (2007 dollars)**	**% of GDP**
Coal electricity	62.00	
Natural gas electricity	0.74	
Transport fuel	56.00	
Heating and process use	1.40	
Total	**120.14**	**0.9%**

Source: National Research Council (2010)

According to the National Research Council, the external damages from fossil fuel use amounted to almost 1 per cent of GDP in the US in 2005. However, as noted in the Introduction, the release of the Global Burden of Disease 2010 overshadowed many estimates of damages from air pollution in the literature, including the figures in Tables 3.1.1 and 3.1.2. As Table 1.1 shows, an estimate of PM2.5 damages in the US in 2010 amounted to 3.2–4.6 per cent of GDP.

What follows is an overview of the linkages between environmental quality and human wellbeing, and then the focus shifts to air pollution damages since these represent the major pathway for climate actions to yield co-benefits. The causal chain between air pollution emissions and damage to receptors of this pollution is explained, with particular emphasis on the ‘concentration response functions’ (CRFs) which allow the estimation of damages. Finally, economic valuation of the damages from pollution exposure is presented. Taken together, this is the analytical machinery that underpins our ability to value the co-benefits of climate mitigation via reductions in local air pollution.

The second major environmental co-benefit which could be considered is the provision of environmental services by natural forests. Forests are closely linked to climate change issues, both as a store of carbon above and below ground, and as a potential source of emissions when land is deforested. Afforestation, reforestation and efforts to reduce deforestation and forest degradation (REDD) all have climate benefits, and standing forests are in turn a source of ecosystem co-benefits.

As will be evident in Chapter 4 on model results, the vast majority of the empirical work on estimating co-benefits has focused on the reduction in air pollution damages associated with climate change mitigation policies. In the interest of brevity and focus on the major impacts, therefore, issues such as forest environmental services will not be considered further in this report. An excellent review article is Ferraro et al. (2012). A similar desire for focus means that other health factors, such as the health benefits of improved urban designs which increase walking and cycling, will be considered only briefly in Section 3.4 – for recent work on this topic see Woodcock et al. (2009) and Jarrett et al. (2012). More comprehensive studies of co-benefits include Smith (2013) and Forster et al. (2013) for the UK.

Environmental quality and human wellbeing

The environment is a direct source of natural resources, including land, water, minerals, energy, forests and fish. In addition, natural areas are also a source of ecosystem services that directly or indirectly contribute to human wellbeing. The Millennium Ecosystem Assessment (2005) provides a useful classification of the services provided by natural ecosystems:

- *Provisioning* services yield goods and services that can be consumed by humans, including bio-energy, food and fibre
- *Regulating* services include the regulation of climate, water flow, pollination, pollution dissipation and soil quality
- *Cultural* services include spiritual and cultural values, as well as recreation and amenity
- *Supporting* services include nutrient cycling, soil formation and primary production

From the perspective of climate mitigation and the co-benefits that it provides, the two most direct linkages of ecosystems and wellbeing are the provisioning and regulating of services. For regulating services, pollution dissipation as well as water and climate regulation are the primary sources of benefits to people, but bioenergy provision can also be important in some instances.

Air pollution co-benefits

Generally speaking, the largest co-benefits of climate change mitigation are associated with air pollution emissions. While the generation of electricity, for example, may yield water pollutants as well as air pollutants, the distinguishing characteristic of ambient air pollution is that it is difficult for people to avoid its effects. This is largely true for human exposure to air pollution, and is obviously true for air pollution impacts on crop yields. This contrasts with water pollution, where people can avoid a polluted local source of water by gathering or purchasing water from further afield. The focus in what follows, therefore, is on air pollution.

The causal flow linking pollution emissions to damages to people or productive processes is as follows:

Emission → dispersion → exposure → impact → valuation of impacts

From a climate and local air pollution perspective the main source of emissions is the combustion of fossil fuels for electricity production, motive fuels for transport, or fuels for heating and industrial processes. For each combustion technology there is an associated *emission factor* measuring the flow of a range of pollutants per unit of fuel burned, plus a *control factor* representing the degree of control of emissions (through scrubbers for sulphur dioxide [SO_2], for example, or carbon capture and storage for CO_2) which limits emissions to the environment.

Models of *dispersion* of air pollutants must account for local climate (particularly wind and temperature) and topography. In addition, dispersion processes also lead to interactions of emitted substances, which are also affected by climate variables. Key secondary products of pollution emission and dispersion include ozone (from the interaction of volatile organic compounds [VOC], carbon monoxide [CO] and nitrogen oxide [NOx]) and particulates. While there are primary emissions of particulate matter from fuel combustion, chemical interactions of SO_2, NOx, ammonia (NH_3) and VOC also lead to the production of secondary particulate matter.

Exposure to air pollution leads to damages to human health, crops, infrastructure and materials, and a variety of natural environments including water, soils, forests and fisheries. The empirical evidence is that the largest damages are to human health and crops. In order to estimate damages to human health, spatial information on pollution concentrations needs to be overlaid with spatial information on human populations. Since exposure to air pollution is generally highest in urban areas, demographic factors such as rates of urbanisation will play an important role in determining air pollution damages.

Measurement of the *impacts* of air pollution is based on concentration-response functions that detail the relationship between a given level of pollution exposure and a given end-point such as damage to materials (for example, the wood, metal, brickwork, concrete, glass, and so on exposed to pollution), damage to crops, or damage to human health.

Valuation of the impacts of air pollution requires the estimation of the economic value of a given change in the end-point of interest. For example, the market price of crops is the obvious way to value crop losses from pollution. But where the end-point is human mortality, then the relevant valuation generally uses some estimate of the 'value of a statistical life' (VSL).[10]

The **methods for estimating the impacts of pollution and the valuation of these impacts** are presented in Appendix 1. The key points to note are that:

- Impacts on human health dominate all of the estimates of the value of damages from pollution exposure, and air pollution is the largest contributor to these health losses.
- The WHO Global Burden of Disease Study 2010 and subsequent updates have made a major contribution to the estimation of the air pollution mortality impacts from exposure to PM2.5 globally, building on a new set of integrated concentration-response functions, as well as new data on levels of exposure.
- There is a large literature on estimating value of a statistical life, using both revealed preference and stated preference approaches. Meta-analyses of the literature provide both central values for the VSL and an assessment of the factors, particularly income, that affect the economic value ascribed to mortality across different countries.

[10] Note that the term 'value of a statistical life' is often misinterpreted. What is fundamentally being valued is the willingness to pay to reduce the risk of death over a given time period. See Appendix I.

One concern in mortality valuation, however, is that the great majority of this literature is based on studies in developed countries, which raises questions about the best way to 'transfer' values to developing countries. Using low values of the elasticity of income of the VSL (less than 1) probably inflates the value of mortality in developing countries (see Hammitt and Robinson, 2011; World Bank, 2016). Another concern is whether the VSL varies with age – if it declines then this will affect mortality valuation for air pollution because it is generally the elderly who are most heavily affected. Krupnick (2007) concludes, however, that 'the evidence for a senior discount [VSL that declines with age] is too thin to be relied on for policy purposes'.

3.2 Economy-wide benefits of climate change mitigation

While there is considerable evidence that reductions in air pollution will yield public health benefits, thus yielding co-benefits from reducing fossil fuel combustion, climate change mitigation policies also have economy-wide impacts as a result of changes in energy use. Our discussion starts with a potential co-cost – the tax interaction effects of carbon taxes. The costs of 'clean' energy are a major concern, and the discussion shifts to impacts on competitiveness. Green innovation is assessed as one response to rising energy costs. The co-benefit story in this instance is whether, as a response to competitiveness concerns, green innovation can actually increase economic efficiency. The focus then turns to green jobs. Finally, wind and solar energy build on indigenous sources of energy. The question arises, therefore, whether energy security increases as a result of expanding the use of clean energy – the same question can be posed for increasing energy efficiency.[11]

(i) Tax interaction effects – a potential co-cost

As Parry and Bento (2000) point out, the early literature on environmental taxation expressed the hope that environmental taxes combined with recycling of revenues through decreases in other tax rates could yield a double dividend: a cleaner environment and a more efficient tax system. However, as Bovenberg and Goulder (1996) and a large literature show, this hope overlooked the problem of tax interactions: adding an additional tax (the environmental tax) may increase the distortionary impact of the whole tax system sufficiently to offset the benefits of recycling revenues through tax reductions. The effect of tax interactions can be large. For example, Bovenberg and Goulder simulate an optimal carbon tax of US$30/t$CO_2$ (1990 dollars) when the assumed marginal damage of CO_2 emissions is US$50/t$CO_2$ – the difference is the net tax interaction effect (ibid).

From the perspective of climate mitigation through carbon taxes or tradable emission permits, this result from the public finance literature suggests a potential co-cost of climate policy: the total cost of the policy equals the financial cost of implementing the tax, plus the net tax interaction costs (after revenue recycling), minus other co-benefits such as reduced health damage from exposure to PM2.5.

It is clear, however, that this tax interaction co-cost depends on the details of both the overall tax system and the revenue recycling vehicle. A newer literature provides examples of where the tax interaction co-cost becomes a co-benefit owing to additional pre-existing distortions in the tax system. Parry and Bento (2000) point out that the US tax system includes subsidies to consumption of particular goods, the key example being mortgage interest deductibility from income taxes. This deduction creates a major inefficiency in the US income tax system, to the extent that recycling environmental tax revenues through income tax reductions creates a net positive tax interaction effect – the optimal pollution tax exceeds the marginal damage of pollution emissions.

[11] As OECD (2011) notes, however, under a cap-and-trade scheme total CO_2 emissions are constant (pending any changes in the cap). Installation of a wind generator frees up emission permits that will be used elsewhere in the economy to burn fossil fuels that may be imported.

In some of Parry and Bento's simulations the positive tax interaction effect exceeds the welfare gains from the environmental tax (ibid).

Similarly, Bento et al. (2013) examine another major inefficiency in the income tax system when there is an informal labour market. In this instance income taxes not only induce a shift towards more leisure but also increase the size of the informal sector that lies outside the tax system (the informal sector is estimated to be 16 per cent of OECD GDP on average, and much higher in developing countries). Recycling revenues from environmental taxation through reduced income tax rates can produce benefits by pulling informal labour into the formal market, thereby increasing the tax base.

On balance, therefore, the details of the existing tax system and its inefficiencies have a large impact on whether a carbon tax, combined with recycling revenues through reductions in taxes on labour, leads to a net tax interaction cost or to the realisation of a double dividend from carbon taxation. Policymakers will need detailed simulations of the tax system in order to examine this question. The existence of large informal sectors, or highly distortionary income tax deductions, however, will favour the 'tax and recycle' approach in many countries.

(ii) Competitiveness

Environmental regulation has long been accompanied by fears that it would harm the competitiveness of a country's firms. More recently, the opposite has also been asserted: that one of the co-benefits of environmental regulation is to spur firms to *increase* their competitiveness.

Competitiveness refers to the efficiency or productivity of one firm relative to another firm, or of a group of firms comprising an industrial sector in one country relative to the same sector in another country. The greater competitiveness of a firm or sector in this sense is likely to be reflected in a variety of outcome indicators, for example greater growth in profitability, sales, and, in particular, market share relative to other firms or sectors.

As Krugman (1994) has argued, it is not altogether meaningful to apply this sense of the term competitiveness to a country's economy as a whole. In the context of the national economy, the meaning of competitiveness undergoes a change, so that it refers mainly to overall national productivity. However, there are only limited circumstances in which the growth in an economy's overall productivity would be at the expense of productivity in other economies. On the contrary, most of the time higher productivity in one country will benefit other countries by providing them with a bigger market for exports and with cheaper and more innovative inputs.

Two other important general points need to be made by way of introduction.

First, while environmental policies may lead to some loss of international competitiveness for a country's pollution-intensive firms, there is no implication that this also reflects a loss in overall national welfare or in the productivity of the economy as a whole. A lack of environmental regulation is in effect a form of subsidy to polluting firms. Environmental policy improves overall economic efficiency and welfare by removing the implicit subsidy for polluting firms and causing a reallocation of resources towards cleaner activities.

Second, developing countries in particular may fear that environmental policies will hinder their industrialisation and that, consequently, it is better to 'grow dirty and clean up later'. Indeed, developing countries may well face coordination and market failures of various kinds that require active government intervention to address and to promote development. However, there is no reason to think that weak environmental policy is the best way to address such development problems. A combination of tighter environmental policy with more focused development interventions would likely yield a better outcome for both welfare and development.

Turning specifically to climate change, low-carbon energy sources are generally more costly than fossil fuel alternatives. To limit the impacts on competitiveness from climate mitigation, there are strong incentives for innovation to reduce these costs. This issue is taken up in sub-section 3.2 (iii) below on green innovation.

Ex-post studies of pollution regulation and competitiveness

In surveying the large existing literature on the effects of environmental policy on international competitiveness, Copeland (2011) notes that much of the theoretical literature starts with the premise that tighter environmental policy raises costs in the affected sectors, although whether or not this leads to a loss of international competitiveness will depend, he argues, on additional factors, such as whether or not pollution is generated during production or consumption, and on the extent to which environmental degradation destroys natural capital.[12]

With production-generated pollution, standard neo-classical theory predicts that environmental regulation drives up firms' costs relative to foreign competitors, causing some shift in pollution-intensive production to foreign countries via either trade or foreign investment (the 'pollution haven' hypothesis). Where pollution is consumption-generated (through car emissions, for example), however, the environmental regulation will apply to both domestic and foreign produced goods, and could conceivably increase the competitiveness of domestic firms if they are able to comply with the regulation more cheaply than foreign firms. Lastly, where economic activity in the presence of market failures causes destruction of natural capital, for example excessive depletion of a fishery, soil degradation or deforestation, environmental protection may reduce competitiveness in the short run but enhance it in the long run, by preserving the natural asset that domestic firms depend upon. A broader interpretation of this argument could point to long-term improvements in the health and productivity of the labour force due to more stringent protection of air and water quality.

Copeland (2011) notes that empirical study of the impact of environmental regulation on international competitiveness is hampered by lack of international data on pollution regulations and has tended to focus on the United States because of data availability. Much of this work studies some version of the pollution haven hypothesis, seeking to estimate the relationship between a measure of pollution regulation and some measure of economic activity such as trade, investment, output or plant location, either across jurisdictions or across sectors within one jurisdiction. Much of the early work tended to find little or no impact of environmental policy on economic variables. More recent work has adopted more sophisticated econometric strategies to address issues such as the possible endogeneity of environmental policy or unobserved heterogeneity. This work has tended to find some statistically significant negative impact of pollution regulation on various measures of competitiveness in the affected sectors,[13] although the economic magnitude of these effects is generally small, reflecting the fact that pollution abatement costs are only a small proportion of total costs in most industries.

There is much less empirical work on the impact of pollution regulation on competitiveness in developing countries. The few existing studies provide little overall guidance, with most finding little or no effect, with others find either negative or positive effects (see Copeland [2011] for references). Bassi and Zenghelis (2014) survey recent ex-post empirical studies of the UK Climate

[12] The Porter hypothesis (Porter and van der Linde, 1995) questions the premise that more stringent environmental policy will necessarily raise costs. This is discussed in detail below.

[13] OECD (2013), however, raises questions about the methodology employed in most of this literature.

Change Levy and the EU Emissions Trading System, most of which do not find these policies to have had any significant impact on competitiveness.

Ex-ante model-based studies on competitiveness

It might be argued that econometric or other studies of past environmental regulation are not especially relevant in the case of climate change because the scale of mitigation policies needed to address this problem is likely to be so much larger and more comprehensive than past environmental policies and experience. Nevertheless, even though an econometric approach appears infeasible, it is possible to throw some light on the potential cross-country effects of climate mitigation policies on competitiveness using multi-country, multi-sector computable general equilibrium (CGE) models.

The context for such studies is the assumption of mitigation actions that are undertaken by only a partial set of countries, the high-income countries. Even though it is well understood that the globally efficient policy is for all countries to undertake mitigation by adopting a common carbon price, such a global approach is unlikely for equity and political reasons. It then becomes important to understand how unilateral mitigation action by high-income countries might affect the competitiveness of their firms relative to those in developing countries.

Mattoo et al. (2009) use the World Bank's ENVISAGE CGE model to study a scenario in which high-income [Kyoto Protocol] Annex 1 countries unilaterally implement a carbon tax of around US$240 per ton of carbon (about US$65 per tonne of CO_2) to reduce their carbon emissions in 2020 by 17 per cent relative to 2005 levels, or by about 30 per cent compared with business as usual (BAU) levels in 2020. In the base scenario this mitigation policy is undertaken without any offsetting border tax adjustment (BTA) on trade with developing countries.

A key finding from this scenario is that adverse impacts on high-income country competitiveness are highly concentrated in energy-intensive sectors that make up a relatively small share of these countries' output, with a relatively small impact on output as a whole. The effects are concentrated in the energy sector itself, which comprised 3–6 per cent of output in high-income countries in 2005 according to the database employed in the ENVISAGE model, and in energy-intensive manufacturing, which comprised 7–11 per cent of output.

Figures 3.2.1 and 3.2.2 below show the results of the simulation on output and exports respectively. As would be expected, output and exports of the energy sector in the high-income countries decline the most, by about 9 per cent. Output of energy-intensive manufactures also declines, ranging from a little under 2 per cent compared to BAU in Japan and the EU, rising to over 4 per cent in the US and close to 6 per cent in other high-income Annex 1 countries. The impact on services and other industries is much lower, however – generally less than 0.5 per cent – and, since these sectors generally comprise around 80 per cent of total output in high-income countries, the impact on total output is also limited, ranging from 0.3 per cent in Japan to 1.2 per cent in the US.

As would be expected, there is some increase in output and exports in energy-intensive manufacturing in developing countries. This is one channel for the phenomenon of 'carbon leakage', where policy-induced decreases in carbon emissions in countries undertaking mitigation are offset to some extent by increases in non-participating countries. Another is the so-called fossil fuel channel: the policy-induced decrease in demand for fossil fuels causes a steep fall in fossil fuel prices in the world market, inducing a shift towards more fossil fuel-intensive consumption and production methods in non-participating countries. The amount of leakage is quite limited, however: overall carbon emissions from developing countries rise 1 per cent compared with the baseline, quite a small amount compared with a close to 30 per cent fall in high-income country emissions.

The findings in Mattoo et al. (2009) are similar to those in other recent ex-ante model-based studies. OECD (2010b) notes the conclusions from OECD staff studies using the OECD ENV-LINKAGES model, for example Burniaux et al. (2010, in OECD 2010b). Table 3.2.1 summarises the results from a scenario in which Annex 1 countries reduce their 2020 emissions by 20 per cent compared with 1990, and 2050 emissions by 50 per cent. In this scenario, with more stringent mitigation targets than in the study by Mattoo et al. (2009), output of energy-intensive industries in Annex 1 countries is projected to fall a little over 3 per cent, with overall real income down 1.6 per cent. Carbon leakage is also expected to be a modest 6 per cent only.

Figure 3.2.1. Change in output (%) – unilateral mitigation in high-income Kyoto Protocol Annex 1 countries (no border tax adjustment)

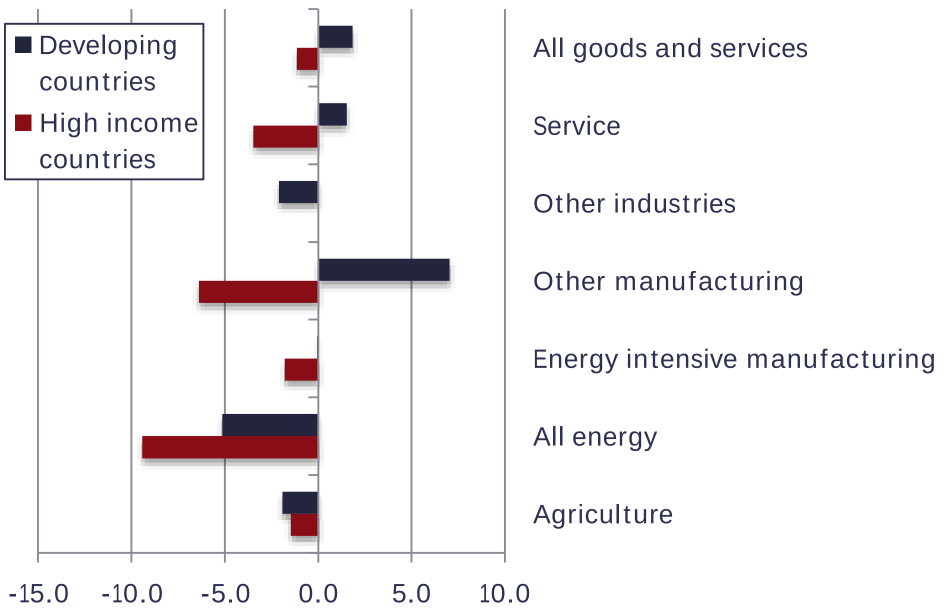
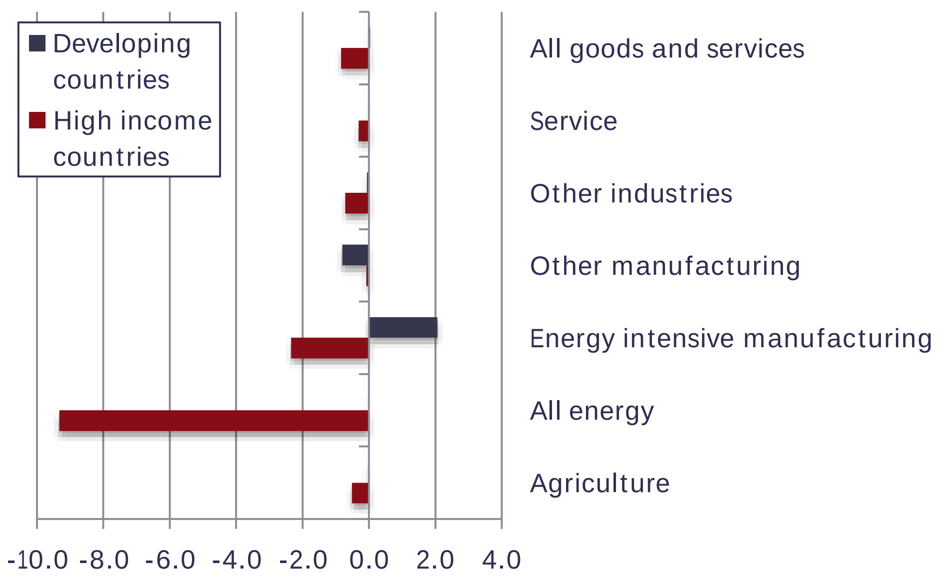


Figure 3.2.2. Change in exports (%) – unilateral mitigation in high-income Kyoto Protocol Annex 1 countries (no border tax adjustment)



Source: Mattoo et al. (2009)

Table 3.2.1. Impacts of alternative climate policy participation assumptions in 2030 (an emissions reduction pathway from 2005 of -20% in 2020 and -50% in 2050)

Policy scenario	% Change in 2030 from baseline of:							
	Output of energy-intensive industries			Real income [1]			World GHG emissions	Leakage rate (%)
	Acting countries	Non-acting countries	World	Acting countries	Non-acting countries	World		
Unilateral action:								
EU	-2.2	0.2	-0.2	-1.4	-0.1	-0.4	-2.4	7.9
USA	-4.6	0.6	-0.3	-1.2	-0.1	-0.4	-5.2	11.8
Japan	-1.4	0.1	0	-0.4	0	-0.1	-0.5	12.5
Coalition action:								
Annex I	-3.2	1.1	-0.9	-1.6	-0.5	-1.2	-13	5.9

Note. 1. Defined here as equivalent variation in income. Benefits from climate policy are not considered. GHG = greenhouse gases.
Source: OECD 2010 (based on Burniaux et al., 2010). © OECD 2010

(iii) Green innovation

Given the current high costs of low-carbon energy compared with fossil fuel alternatives, there is an obvious and urgent need for innovations to drive down the costs of cleaner energy. Without this, 'green growth' will remain a distant aspiration. Are there co-benefits from green innovation?

The co-benefits of climate action are defined to be the ancillary benefits associated with efforts whose primary purpose is to reduce greenhouse gas emissions. The first task in linking co-benefits to 'green innovation' (in this instance innovation aimed at reducing the cost of greenhouse gas abatement) is therefore to identify the associated co-benefits.

This is not entirely straightforward. Generally speaking, policies to abate greenhouse gases will increase costs of production and reduce profits in emitting sectors. One response on the part of producers is therefore to research and develop new, lower-cost, technologies for reducing greenhouse gas emissions. But because these cost reductions are an integral part of implementing greenhouse gas abatement policies, there is no logical reason to view them as co-benefits.

If, however, new green technologies can actually reduce costs and increase profitability while reducing greenhouse gas emissions, then the incremental increase in profits could conceivably be counted as a co-benefit. This is not a particularly tidy definition of co-benefits, but it links directly to the very large literature on the 'Porter Hypothesis' (Porter, 1991; Porter and van der Linde, 1995). Porter poses the question of whether environmental policies, in some instances, can actually stimulate greater economic efficiency.

Ambec et al. (2013) take stock of the literature on the Porter Hypothesis 20 years after its publication. Drawing on Ambec et al., the following discussion considers: definitions, theoretical underpinnings, empirical tests of the hypothesis, and policy design.

The Porter Hypothesis: strong, narrow or weak?

In the intervening 20 years, the literature has settled on analysing three possible variants on the hypothesis, first suggested by Jaffe and Palmer (1997):

- The *weak* Porter Hypothesis suggests that regulation spurs innovation as firms react to the new restrictions on their activities.
- The *narrow* Porter Hypothesis suggests that flexible regulatory approaches, such as economic instruments, create stronger incentives for firms to innovate than more restrictive regulations such technology mandates.
- The *strong* Porter Hypothesis suggests that regulation can actually be beneficial, in the sense that profitability increases after firms act to comply with regulation.

Much of the empirical literature on the hypothesis tests some combination of these particular benefits from regulation. The weak hypothesis raises the question of whether regulation serves to induce innovation. The narrow version corresponds to one of the principal arguments in favour of using economic instruments to deal with environmental problems, since that creates a financial incentive to drive down the cost of abatement through innovation. The strong version corresponds to the tentative definition offered above of a co-benefit from green innovation.

The Porter Hypothesis: theoretical underpinnings

On the face of it, the strong hypothesis appears to conflict with the assumption that firms are profit-maximisers, as argued by Palmer et al. (1995). Rather than simply assuming that this is the case, consideration has gone into theoretical reasons why the strong hypothesis could hold in practice, at least in particular circumstances.

First, if we assume that there are profitable investments in reducing pollution emissions that are not being taken up, the explanation my lie in behavioural economics. Kennedy (1994) suggests that while such investments may exist for the firm, individual managers may be risk averse and so unwilling to make a risky investment; regulation provides the impetus to overcome this risk aversion.

A variety of market failures can also explain why regulation could increase profits. One example is market power (Mohr and Saha, 2008). If emission permits are the chosen instrument for controlling pollution, then handing out free permits to incumbent firms ('grandfathering') can create barriers to entry into the sector, since potential market participants will have to buy costly permits. As a result, grandfathering can increase profits in the sector by limiting competition. Of course, this is a negative result in the sense that these profits would presumably disappear under more optimal policy.

R&D spillovers reduce incentives to innovate, and these can apply to environmental innovation as well. Profitable innovations may not be pursued owing to incomplete appropriation of the benefits of innovation. Mohr (2002) argues that environmental regulation can overcome this barrier and lead to levels of innovation that are closer to the social optimum.

Xepapadeas and Zeeuw (1999) argue that regulation may create incentives to shut down older capital stock where compliance with the regulation would be very expensive. As a result the average vintage of the capital stock in a sector would decrease, potentially increasing average productivity. They argue, however, that this will not necessarily increase overall profits.

The Porter Hypothesis: empirical findings on green innovation

The weak version of the hypothesis, that regulation has induced innovation, has been supported in a large literature looking either at R&D expenditures or patents granted for environmental technologies. An early contribution was by Jaffe and Palmer (1997), who found correlations between total pollution abatement costs and total R&D expenditures. More recent work by Johnstone, Hascic and Popp (2010) finds that in addition to inducing innovation, different renewable energy policies have distinctive effects on investment in different types of renewable energy. This wide literature is summarised in more depth in Ambec *et al*. (2013).

The strong version of the hypothesis, that profits actually increase under environmental regulation, tended to be strongly rejected in early empirical work. For example, Gollop and Roberts (1983) estimated that introducing SO_2 controls reduced US productivity growth by up to 43 per cent in the 1970s. More recent work, however, is more positive, with Berman and Bui (2001) finding that petroleum refineries in the Los Angeles area were more productive than similar refineries in regions with less stringent pollution regulation. Alpay et al. (2002) found that the productivity of the Mexican food processing sector increased as more stringent environmental regulations were introduced.

Dynamics may also be an issue in the empirical work. Lanoie et al. (2011) argue that there is a lag between introduction of a regulation and the measurable effects of any innovation on productivity. By introducing three to four year lags into the analysis of manufacturing firms in Quebec, they find modest increases in productivity after the imposition of environmental regulations.

Johnstone et al. (2010) exploit a survey data set covering 4,000 firms in seven OECD countries, which permits them to test the three variants of the Porter Hypothesis. They find strong support for the narrow and weak versions – that regulation induces innovation and that the effect is stronger with more flexible regulatory policies. Environmental innovations do increase firm performance, but not enough to support the strong version of the hypothesis. The net effect of environmental regulation on performance is negative.

Finally, the question of whether the strong version of the Porter Hypothesis applies to countries as a whole, rather than firms, has been the subject of empirical research on the 'pollution haven' hypothesis – whether firms will tend to invest in jurisdictions with relatively weak environmental regulations. While early empirical work supported this hypothesis, Copeland and Taylor (2004) in a review article noted significant econometric deficiencies in this work, including endogeneity and missing variable bias. They concluded that environmental regulatory costs are in any event too small for most firms to be a primary determinant of where to invest.

The Porter Hypothesis: conclusions on green innovation

The first general conclusion from the assessment of Ambec *et al*. (2013) is that strong empirical evidence exists for both the weak and narrow variants of the Porter Hypothesis. Environmental regulation does induce innovation aimed at reducing compliance costs, and more flexible regulatory instruments increase the level of induced innovation. The second conclusion is that the evidence for the strong version of the hypothesis is decidedly mixed.

If the strong Porter Hypothesis held true in general, then the whole question of the theoretical foundations for the finding would need further research. As it stands, there are selected results in the literature that do support the strong hypothesis, and a suggestion in Ambec *et al*. (ibid) that the empirical literature may have overlooked the necessary lag between the introduction of a regulation and the application of the results of R&D aimed at reducing implementation costs. Returning to the question of co-benefits, measured as the net increase in profitability as a result of green innovation, the limited theoretical foundations of the strong Porter Hypothesis and the mixed empirical evidence suggests that governments should not assume that green innovation on climate change will yield co-benefits, although there may be individual circumstances under which this will hold true.

(iv) Green jobs

Linked to concerns about competitiveness are fears that environmental policies in general or climate mitigation policies in particular will significantly increase unemployment. Others have argued that such policies will, on the contrary, be a source of 'green jobs'.

Defining 'green jobs'

Efforts to define the concept of a 'green job' have attracted a good deal of effort. One approach has focused on specific occupations with an environmental focus or on particular sectors whose products are deemed to be of environmental benefit. OECD/Eurostat (OECD, 1999), for example, define green jobs as comprising 'activities which produce goods and services to measure, prevent, limit, minimize or correct environmental damage to water, air and soil, as well as problems related to waste, noise and eco-systems. This includes technologies, products and services that reduce environmental risk and minimize pollution and resources.' Bowen (2012) argues that on this definition green jobs comprise a small but not insignificant 1.7 per cent of total employment in Europe. The proportion of green jobs in world employment on this definition will likely be much smaller than in Europe, since most jobs with these features have mostly come into being in developed countries.

Other definitions attempt something more fluid and comprehensive. UNEP et al. (2008), for example, defines green jobs as 'work in agricultural, manufacturing, research and development (R&D), administrative, and service activities that contribute substantially to preserving or restoring environmental quality. Specifically, but not exclusively, this includes jobs that help to protect ecosystems and biodiversity; reduce energy, materials, and water consumption through high-efficiency strategies; de-carbonize the economy; and minimize or altogether avoid generation of all forms of waste and pollution.' This has the merit of focusing attention not on a particular occupation or product but rather on such functions as reducing energy consumption, which occurs in all industrial sectors rather than only in specific green sectors.

A further approach is to avoid the definitional debate altogether and instead to simply focus on the employment impacts of green policies, considering the net impact on overall employment levels, shifts in employment across sectors, and changes in occupations and skill levels within sectors.

Empirical methods and estimates

Surveys of green jobs studies include GHK (2009), Global Climate Network (2010) and Wei et al. (2010). The studies surveyed use a range of methods reflecting the different definitions of green jobs and differ according to the sectors or technologies covered and the extent to which they focus on gross or net job effects, among others.

While it is hard to draw broad conclusions because of the heterogeneity of the studies, the most common approach used is a 'partial' one that looks at the potential employment impacts of development and deployment of one or a number of related technologies, for example clean energy technologies. This approach typically makes use of engineering or firm-level data to provide an estimate of 'direct' employment created in the 'design, manufacturing, delivery, construction/installation, project management and operation and maintenance of the different components of the technology, or power plant, under consideration' (Wei et al., 2010). Such direct estimates are often complemented by estimates of 'indirect employment' created in upstream and downstream industries because of input–output linkages, and 'induced employment', which calculates the economy-wide employment enhancement due to the demand multiplier effects of spending by direct and indirect employees. Bacon and Kojima (2011) review measurement and methodological issues with this class of models.

Bowen (2012) notes that, 'the impression given by many of the findings is that climate-change policies in general and renewable energy in particular can generate considerable extra employment. However, apart from the fact that there is a considerable uncertainty regarding engineering estimates of labour coefficients, estimates of which vary widely across sources, most of these studies did not attempt to estimate the general equilibrium effects of expansion in green sectors, for example the potential for job-destruction in competing "non-green" industries (with their own attendant indirect and induced job effects), or other general equilibrium effects.' Fankhauser et al. (2008) observe that the higher labour intensity per unit of energy in green energy sectors implies lower labour productivity than in conventional energy production, which, combined with high capital requirements, shorter plant lives and more intermittent energy production, tends, for the present, to push green energy prices higher than conventional sources. The impact of higher energy prices on reducing real wages and raising costs and reducing employment in energy-using sectors are among significant general equilibrium effects.

Surveying economy-wide model-based studies of climate mitigation, Cambridge Econometrics et al. (2013) conclude that, 'There is no clear consensus about whether the overall net impact on employment (defined as number of jobs) will be positive or negative, but in almost all cases the impacts are small at macroeconomic level.' Cambridge Econometrics et al. (2011) study climate mitigation policies in the EU using an econometrically estimated multi-sector model and conclude that with appropriate recycling of tax revenues such policies could lead to a small net increase in employment of 0.1 to 0.3 per cent compared to the baseline, mainly due to higher employment in capital goods-producing sectors, resulting from the large levels of investment required in renewables and energy-efficient equipment, including the increased transmission capacity and generation backup requirements needed by low-carbon technologies. Many studies conclude that the switch to low-carbon technologies will have a high capital cost. It is not clear how well the referenced studies evaluated the possibility that significantly higher investment in the clean power sector might crowd out or be offset by lower investment in other sectors.

While employment increases in capital goods sectors in the study by Cambridge Econometrics et al. (2011), it declines in sectors that supply fossil fuels and some intensive users of energy. Nevertheless, the amount of 'churn' or job destruction and job creation linked to climate mitigation is expected to be quite small – perhaps 0.5 per cent of total employment – compared to the overall churn that normally occurs in a market economy, which is on the order of 20 per cent of total employment (Cahuc and Zylberberg, 2004; Cambridge Econometrics et al., 2011).

Broadly similar results are obtained by Congressional Budget Office (2010) for the United States, using a number of computable general equilibrium (CGE) models, with small net employment effects and more significant shifts of workers across sectors.

Labour market imperfections and macroeconomic context

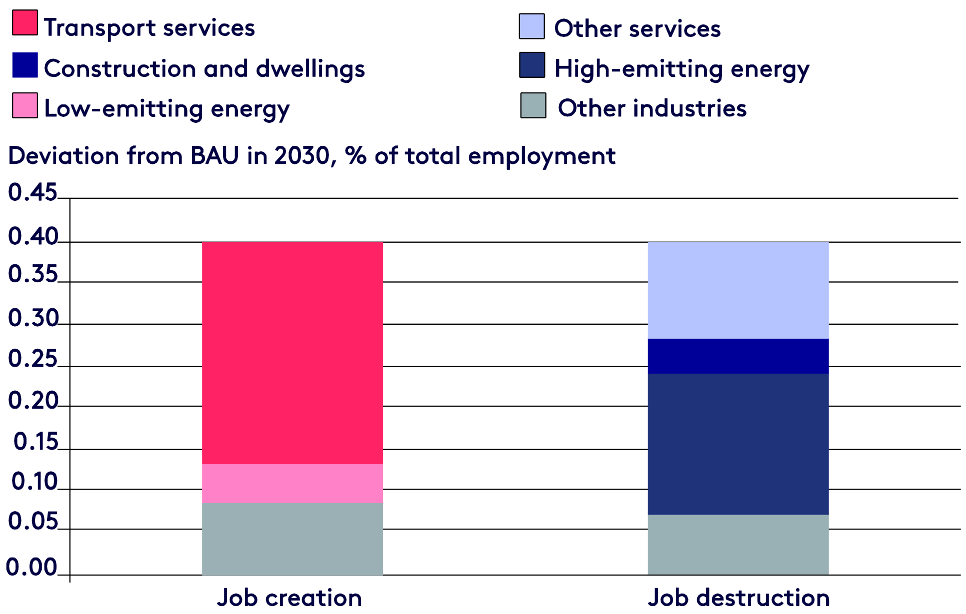
While net employment effects of climate mitigation are generally expected to be small, a number of studies note that more significant impacts are possible if there are significant labour market imperfections that hamper labour mobility, or depending on the macroeconomic policy framework.

Chateau et al. (2011) illustrate these issues in a study using the OECD's ENV-Linkage CGE model. They first analyse a central scenario in which the OECD implements climate mitigation policies (an emissions trading system), resulting in a 50 per cent reduction in CO_2 emissions by 2050 compared with 1990 levels. In this scenario it is assumed that carbon revenues are distributed to citizens through lump sum transfers. There is perfect labour market flexibility, so that all adjustment occurs in real wages. There is a GDP loss compared to the baseline in 2050 of 1.7 per cent due primarily to higher production costs and some loss of competitiveness relative to the non-OECD, although the GDP decline is modest compared with the substantial increase in GDP that occurs in the baseline. With weaker labour demand there is a 2.5 per cent decline in real wages compared to baseline, although with perfect labour market flexibility, there is no change in overall net employment; as expected, there is job creation and destruction across sectors. Figure 3.2.3 illustrates the labour market 'churn' in this scenario. Job creation and destruction as a result of climate mitigation each add up to only about 0.4 per cent of total employment (compared with total churn of around 20 per cent). Interestingly, quite large proportions of both job creation and destruction occur outside the clean and dirty energy sectors, reflecting general equilibrium effects.

Chateau et al. (2011) go on to study the impact of labour market imperfections and different modes of recycling emissions trading system (ETS) revenues.

Labour market imperfections are modelled in a simple way through a wage equation that assumes that real wage adjustment accounts for only a fraction of shocks to labour demand caused by climate mitigation, with the rest absorbed by job losses. With low labour market imperfections real wages absorb 80 per cent of the adjustment, with medium imperfections 50 per cent and with high imperfections 20 per cent. Table 3.2.2 shows the impact of OECD mitigation on employment from 2015 to 2030 under the assumption that ETS revenues are recycled through lump sum transfers. With medium labour market imperfections, for example, employment is 0.8 per cent below the baseline. Chateau et al. (2011) show that with high labour market rigidity employment falls to 2.2 per cent below the baseline. The latter would be enough to reduce OECD employment growth in 2012–2030 from 7.8 to 5.5 per cent.

Figure 3.2.3. Simulation results: sectoral composition of job creations and job destructions



Note: BAU = business as usual. Source: Reproduced from Chateau et al. (2011). © OECD

Table 3.2.2. Employment impact of an OECD-wide emissions trading scheme (ETS) for different recycling options of ETS revenues and an intermediate degree of labour market imperfections (% deviation from baseline)		
Year	Lump sum transfers	Lower labour taxes
2015	-0.05	0.12
2020	-0.29	0.59
2030	-0.75	0.80

Source: Chateau et al. (2011)

These results suggest that the institutional and other characteristics of the labour market will be quite significant for the employment impacts of climate mitigation policies, and that policymakers will need to pay close attention to labour market policies and reforms as a complement of climate mitigation.

Table 3.2.2 also illustrates the importance of the macroeconomic framing of climate policies, in particular the mode through which climate mitigation revenues are recycled. As noted, by 2030 climate mitigation results in a 0.75 per cent fall in employment with medium labour market imperfections and when revenues are recycled through lump sum transfers. When revenues are used to cut labour market taxes, however, employment increases by 0.8 per cent compared to the baseline. In other words, in a scenario with medium labour market imperfections, different fiscal policies for recycling mitigation revenues can make for a 1.6 per cent swing in employment outcomes.

(v) Energy security

The concern with energy security can be traced back to the 1973 oil crisis when supplies of oil to selected high-income countries were curtailed by the Organization of the Petroleum Exporting Countries (OPEC), resulting in a large shock to the global economy. In recent years events such as the oil price spike in 2008, increased price volatility in the crude oil market, political tensions over energy supply, and concerns with climate change have brought new focus to energy policies and revived the energy security debate.

The response to climate change has many potential impacts on energy security, both on the supply side – renewable energy sources usually come from indigenous sources, and on the demand side, where energy efficiency improvements reduce the demand for energy. This raises the question of whether improved energy security can be considered to be a co-benefit of climate policy.

Owing to its context-dependent nature, energy security has no universally accepted definition; the International Energy Agency defines energy security as 'uninterrupted availability of energy sources at an affordable price' (IEA, 2012b). The IIASA Global Energy Assessment (GEA, 2012) adopted 'sovereignty, robustness and resilience' as benchmarks for evaluating energy security. The OECD emphasises the importance of energy supply resilience and defines energy security as 'the resilience of the energy system to unique and unforeseeable events that threaten the physical integrity of energy flows or that lead to discontinuous energy price rises, independent of economic fundamentals' (OECD, 2010). Manipulation of energy supply for political purposes, as has happened in Ukraine, is part of the subtext for this definition. Linked to energy security, although not identical, is the concern with energy price volatility, and oil price volatility in particular.

Economic impacts of oil price volatility

Fossil fuel, dominated by oil, accounts for 90 per cent of the global energy mix and is arguably one of the major drivers of the world economy. As documented by Ebrahim et al. (2014), oil price volatility is much greater than the volatility of other commodity prices, due to three factors: a very tight market (highly inelastic oil supply and demand), speculation in the oil market, and inadequacies in oil data. While the effects of speculation in financial markets have been challenged (Kilian and Murphy, 2014), Ebrahim *et al*. document the many shortcomings in oil supply and use data that limits the ability of economic agents to plan for oil market movements.

Oil price volatility is a concern for macroeconomic policy because of a large literature showing that volatility in general (not just concerning oil) has negative impacts on growth. Aizenman and Pinto (2005) found that this is exacerbated in poor countries and suggest that this is linked to weak institutions and financial sectors. An inability to run counter-cyclical fiscal policies appears to be particularly damaging. Hnatkovska and Loayza (2005) found that a one standard deviation increase of volatility of GDP growth in their sample of countries leads to decreased growth of 1.3 per cent per year. Crises, which represent extremes of volatility, are particularly damaging to long run growth.

Turning to oil shocks specifically, Rasmussen and Roitman (2011) examined the long run (1970–2010) economic impacts of oil price increases in 144 countries, including both developing and developed, and found oil shock episodes have been generally associated with lagged and relatively small impacts on growth in oil importers. The study quantifies the short-term oil shock impacts on GDP growth at various degrees of oil dependence and found that given a 25 per cent increase in oil prices, countries spending more than 4 per cent of GDP on oil imports will lose 0.8 per cent of GDP compared with a 0.3 per cent GDP loss for a typical oil-importing country.

Rasmussen and Roitman (2011) suggest that the reason for this moderate impact is that oil exporters will generally increase their imports as export revenues boom, which adds to growth in oil importing countries. Blanchard and Gali (2007) argue that oil shocks have been less damaging in recent years because labour markets are more flexible and monetary policy more credible compared with earlier shocks.

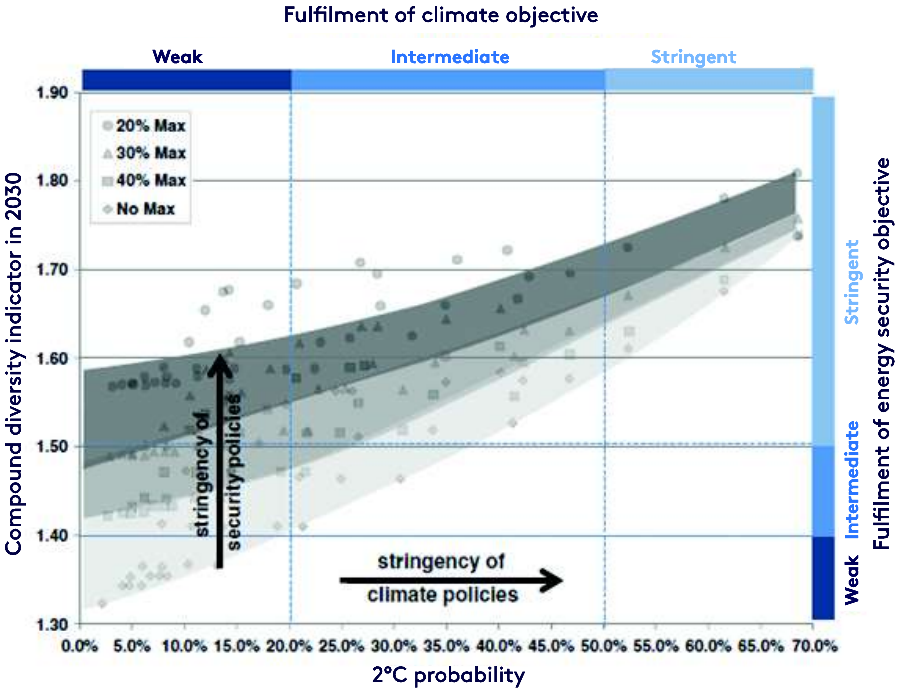
Is there an energy security co-benefit of climate policy?

The Global Energy Assessment’s definition of energy security emphasises two basic components: sovereignty (low import dependence) and robustness to risks (GEA, 2012). This leads to the construction of a simple country-level index of energy security:

$$I = -\sum_j \left(1 - m_j\right) \cdot s_j \cdot \ln(s_j)$$

Here m_j is the import share of fuel j, and s_j is the share of fuel j in total energy supply. The term in fuel shares is a Shannon-Wiener index of diversity, while the first term is a measure of the domestic share in energy supply. Since diversification is an effective way to reduce risk in a portfolio, this composite index captures some aspects of exposure to volatility in world energy markets and diversity of supply. The index is a useful ‘straw man’ measure of energy security, with increases in the index implying higher security. GEA (2012) then calculates this index for its portfolio of energy scenarios leading to different probabilities of limiting warming to 2°C above pre-industrial levels this century (a measure of climate policy stringency). The result is scattered in Figure 3.2.4.

Figure 3.2.4. Synergies between climate change mitigation and near-term improvement of energy security



Source: Global Energy Assessment (2012), reproduced with the permission of Springer

As can be seen, there is a general upward trend in the energy security index as climate policy becomes more stringent. This can be construed as a co-benefit of the stringency of climate policy. What results such as that shown in Figure 3.2.4 reveal, however, is the movement of an index of energy security, and not any measure of economic benefits associated with increases in the index.

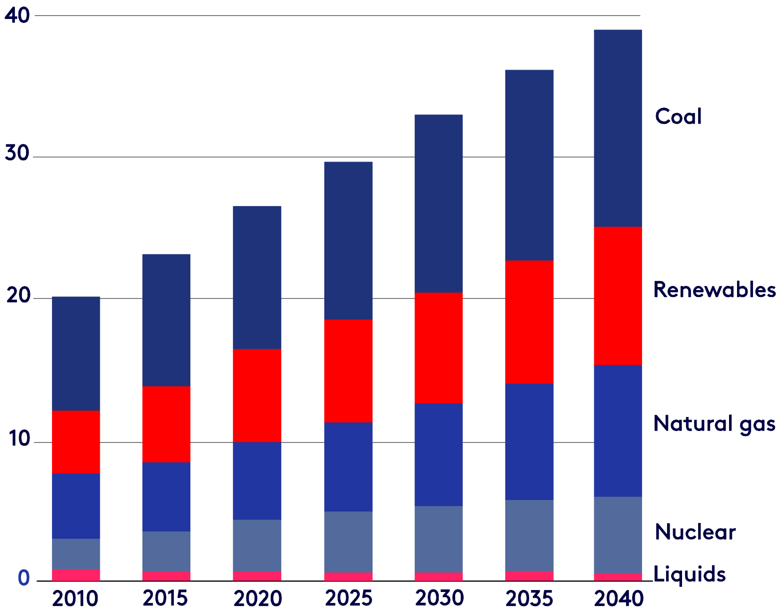
The question of the economic benefits of increasing energy security is obviously important, but largely unanswered because of the divergent definitions of energy security in the literature. That said, energy security is an issue of concern to many policymakers.

There is an obvious risk in using a simple energy security index such as the one developed in GEA (2012). Greener energy supplies, including sunlight, wind and energy efficiency, will generally be from indigenous sources, which highlights the sovereignty portion of the index. However, the investments required to tap these energy sources will require machinery and equipment and rights to technologies that may be imported. Governments may be tempted to favour domestic champions in supplying these investment goods, with harmful 'green protectionism' as the result. Moreover, until energy storage issues are solved, widespread deployment of renewable technologies may actually decrease the reliability of energy supply, reducing energy security.

3.3 Electric power sector

Meeting the world's demand for electricity requires the consumption of billions of tonnes of coal, trillions of cubic feet of natural gas, tens of thousands of large dams (WWF, 2013), hundreds of thousands of wind turbines (Global Wind Energy Council, 2013) and countless miles of transmission lines. Each of these forms of energy places a burden on local communities. As demand for electricity increases so too does the burden. The world produces nearly twice as much electricity today as it did in 1990. According to the US Energy Information Administration, demand is expected to grow another 50 per cent by 2030 as a result of increased economic growth, as seen in Figure 3.3.1. This is primarily driven by increased demand growth in developing countries whose electric consumption could double by 2030 (US Energy Information Administration, 2013). In absolute terms energy consumption is expected to increase in many countries as electricity access expands and citizens are lifted out of poverty, and as a result of vehicle electrification in some markets.

Figure 3.3.1. World net electricity generation by fuel, 2010–2040 (trillion kWh)



Source: US Energy Information Administration (2013)

As we document here, there is a strong intersection between reducing greenhouse gas emissions and reducing the broader set of public health and environmental impacts from electric generation. The evidence suggests that the largest quantifiable impacts from electric generation on a kWh basis, and in the aggregate, come from the combustion of coal. There are many cost-effective technologies that can help reduce this impact. However, it is also true that greenhouse gas reduction policies that shift generation away from coal can frequently bring public health and environmental co-benefits. Those policies include, but are not limited to: renewable standards and subsidies, carbon taxes, greenhouse gas cap-and-trade programmes, clean energy standards, and direct public investment in other forms of generation.

Better understanding the tradeoffs associated with various electric generation technologies choices allows decision-makers to account more fully for the wide range of impacts associated with various policy decisions, and may justify more ambitious climate policies in some cases. Therefore, below we unwrap the costs and benefits associated with various electric generation technologies.

Overview of the externalities associated with generation

Coal

Coal contains sulphur, nitrogen, mercury and other toxic substances, which are released through the combustion process along with some incompletely combusted hydrocarbons including particulate matter (PM) and volatile organic compounds (VOCs). The combustion process itself also forms nitrogen oxides (NOx) as the high temperatures generated cause nitrogen (N_2) and oxygen (O_2) in the atmosphere to react. Once emitted from the plant and given time to cool in the atmosphere, SO_2 and NOx form ultrafine particulate matter (PM 2.5), a dangerous air pollutant. They then rain down to the ground causing acid rain which can damage manmade structures and plants, and can make bodies of water inhabitable to fish. NOx also reacts with sunlight and VOCs from combustion and natural sources to form ground level ozone, another pollutant harmful to public health as well as to vegetation, including crops. Some toxic substances, such as arsenic, lead and cadmium, are enriched in particulate emissions. Mercury, a powerful neurotoxin, is emitted as a gas and falls to the earth through rain, concentrating in streams and ultimately bio-accumulating in fish.

A typical power plant (500 MW) produces enough heat to boil millions of litres of water each day (Bright Hub Engineering, 2011). Once used, this steam must be cooled so that it can recycle through the plant. The cooling process frequently employs once-through cooling systems or closed loop systems, which utilise nearby water sources such as rivers, lakes, aquifers or the ocean. According to the US Geological Survey, 41 per cent of all freshwater withdrawals in the United States in 2005 were for thermoelectric power operations, nearly all of which was used for cooling (Kenny et al., 2009). Depending on the systems employed, this can lead to stream impairment through heat effluent and by fish impingement on intake structures.

The combustion of coal also generates a considerable amount of unburned material, or ash, which then must be disposed of or put to beneficial use. This ash can contain high concentrations of toxic substances, such as naturally-occurring radioactive materials, mercury, arsenic and lead. From time to time these materials have been known to be discharged into nearby waterways when storage structures have been compromised.

Mining itself can impact local communities and the environment through acid mine draining, accidental discharge of slurry generated through coal processing, and through the destruction of landscapes when mountaintop mining is employed.

Natural gas

Natural gas is typically refined before its combustion in a large power plant. Therefore, natural gas-fired power plants are not major sources of SO_2 or toxic metals such as mercury. They also tend to burn cleaner than coal, leading to lower emissions of large particulate matter. They can, however, be a significant direct source of ultrafine particulate matter (PM2.5). While natural gas tends to contain very little nitrogen, like coal its combustion generates high temperatures that cause nitrogen in the atmosphere to react with oxygen to form NOx. Like coal, cooling infrastructure for natural gas generation facilities can impair streams through their heat effluent and by impinging fish on their intake structures.

The production, processing, transmission, storage and distribution infrastructure can be a large source of methane emissions and significant sources of emissions of SO_2, VOCs and toxic air pollutants such as benzene, ethylbenzene and n-hezane (US EPA, n.d.). In addition, the process of obtaining natural gas through fracking has come under increased scrutiny out of concerns for potential impacts on groundwater and for causing earthquakes.

Nuclear

Nuclear plants do not emit any SO_2, NOx, particulate matter, or toxic hydrocarbons or metals. They typically require considerable cooling for their steam condensers, and thus can impair streams through direct heat effluent and by impinging fish on their intake structures.

A typical plant generates 20 tonnes of nuclear waste each year (Nuclear Energy Institute, n.d.). This waste persists in the environment for tens of thousands of years (US Nuclear Regulatory Commission, 2015), raising important issues about how to value risks using the large discount rates (3 per cent or more) typically used in economic analysis. Reprocessing offers the possibility to recycle spent fuel but this process generates some amount of weapons-grade nuclear material, resulting in elevated security risks.

Accidents that lead to direct releases of radioactive material, like those at Fukushima and Chernobyl, are rare, but have significant impacts on human society, health and the environment.

Wind

Wind turbines do not result in any emissions of air pollutants, and do not impact streams through cooling needs. However, they can present a hazard for flying animals, such as birds and bats. It has been estimated that wind turbines in the United States have resulted in between 140,000 and 328,000 bird deaths per year (Loss *et al*., 2013a). By comparison, buildings are estimated to have resulted in 97–976 million deaths, cars 60 million, pesticides 72 million, and free-ranging domestic cats 1.3–4 billion (Loss *et al*., 2013b). Bat deaths have proven more difficult to estimate reliably (National Research Council of the National Academies, 2010).

Wind turbines also have a larger geographical footprint than fossil fuel power stations, leading to questions about the aesthetic impacts on landscapes. In some circumstances they can also present issues with regards to noise, flicker and radar interference.

Solar

Photovoltaic facilities do not have significant on-site impacts but do require considerable energy and material inputs. Thin-film PV technologies can require inputs of toxic metals such as arsenic and cadmium, whose extraction and processing can result in environmental releases. Concentrating solar facilities must cool and condense their steam. These facilities are located in regions with good solar resources, which can be water-scarce.

Hydropower

The creation of large dams typically involves the creation of large retention pools or lakes that can displace populations. These bodies of water are typically oxygen-poor environments that are inhospitable to fish. If their creation submerges significant amounts of vegetation, which is subsequently allowed to decay, then they can also lead to significant methane emissions. In addition, the creation of large hydro facilities can impact the flow of rivers, harming migratory fish populations and other wildlife. Smaller, run-of-river dams do not require the creation of large reservoirs, and thus have smaller impacts.

Biomass

Emissions from biomass power plants vary widely, depending on their inputs (National Research Council of the National Academies, 2010). Virgin wood inputs contain lower amounts of sulphur and nitrogen than coal. However, the wood combustion process can result in considerable emissions of NOx and particulate matter.

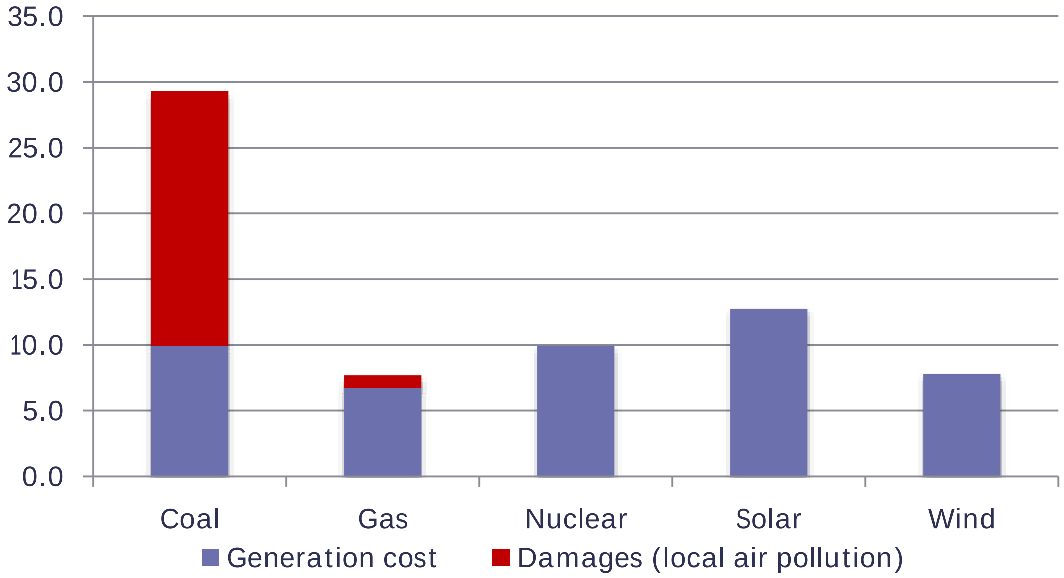
Modelling damages from electric generation in the US

While the preceding subsection lists the qualitative impacts of different electricity generation technologies, Shindell (2015) presents a model to comprehensively assess the emissions of air pollutants from a full range of fuel combustion technologies, including electricity generation. To give a sense of the potential co-benefits associated with mitigating greenhouse gas emissions from electricity generation in the US, we derive these results from Shindell (2015) and the supplementary material.

Shindell's measure of the 'social cost of atmospheric release' (SCAR) includes climate damages as estimated from integrated assessment models (IAMs), calibrated in Shindell (2015) to match the calculated social cost of carbon published by the US government (US Government Interagency Working Group, 2016). Shindell then adds recent estimates of damages to human health from climate change (over and above the esimates used by the IAMs), as well as health damages from exposure to local air pollutants (principally PM2.5 and ozone) and to products of incomplete combustion (black carbon, organic carbon and carbon monoxide). The climate damages in SCAR are also expanded to include the value of regional precipitation changes owing to aerosols.

Figure 3.3.2 below presents results on the cost of electricity generation by technology in the US in 2014 combined with the value of health damages (mortality only) from local air pollution associated with generation. All results are expressed in cents per kWh (in 2012 dollars) to permit comparisons between damages and generation costs.

Figure 3.3.2. Costs of electric generation and associated local air pollution damages in the US, 2014, cents/kWh (2012 dollars)



Source: Calculated by authors from Shindell (2015) supplementary material, including Fig S5 and Table S3

As the figure shows, local air pollution damages are significant for coal and to a lesser extent for gas generation, compared with zero local air pollution emissions for nuclear, solar and wind. While the social cost of gas generation in the US (here calculated as the sum of generation costs and local air pollution damages) is roughly equal to the the generation costs for wind power (7.7 cents versus 7.8 cents), it is lower than the generation costs for nuclear and solar generation. In contrast, the social costs of coal electric generation are three times the generation costs of nuclear, four times those of wind and two-and-a-half times those of solar. This analysis suggests that significant social benefits could be realised in the US if coal generation were replaced by nuclear, solar or wind in order to reduce greenhouse gas emissions. Of course, solar and wind power are intermittent sources of electricity and so are not perfect substitutes for baseline power from coal and gas, and so the cost of intermittency would need to be factored into any analysis of substituting greener electricity technologies.

Conclusions on the externalities from electric power generation

Coal-fired power plants can exert considerable impacts on public health and the environment. This creates a strong intersection between reducing greenhouse gas emissions and reducing the broader set of public health and environmental impacts from electricity generation, as any policy that shifts generation away from coal can frequently bring public health and environmental co-benefits. Accounting for the full range of these co-benefits will inherently lead to better policy outcomes that maximise public benefits. They may also justify more ambitious greenhouse gas reduction policies.

3.4 Transport sector

Transport makes up roughly one-third of total energy use in high-income countries, and large middle-income countries have similar ratios. From the co-benefits perspective the interesting question concerns the direction of causality – should climate change drive transport policy, yielding local co-benefits in terms of air pollution and congestion, or should local benefits drive transport reforms, yielding climate change co-benefits in the form of reduced greenhouse gas emissions?

Parry et al. (2007) review a large US literature on the social costs of automobile transport and arrive at consensus estimates of these costs expressed in dollars per gallon of transport fuel used or per vehicle mile travelled (VMT). While a US example might not appear to be representative of global trends, the US exhibits very high dependency on private cars and a relatively undeveloped public transport system, both of which are symptomatic of transportation issues in large middle-income countries. The key differences in developing countries compared with the US are a lower ratio of infrastructure to vehicles (China excepted), lower fuel efficiency, and much higher pollution emissions and accident rates.

Table 3.4.1 summarises the results from Parry et al. (2007). The first striking finding is that fuel-related externalities are a small proportion of the total – at roughly US$7/t, the social cost of CO_2 emissions from transport amounts to 6c on the gallon, while 'oil dependency' (largely the cost of oil price volatility) amounts to 12c. This compares with US$1.05 per gallon for the social cost of congestion and a further 63c for accidents and 42c for local pollution (see also OECD, 2014). At 2013 EPA estimates of the social cost of carbon, US$37/t CO_2, the contribution to climate change would rise to roughly 30c per gallon, but this is still dwarfed by other social costs. Excluding climate change, the external costs of automobile transport in the US amounted to 2.3 per cent of GDP in 2005.

Table 3.4.1. External costs of automobile transport in the United States, 2005		
	US cents/gallon	Cents/vehicle mile travelled (VMT)
Fuel-related costs		
Climate change	6	0.3
Oil dependency	12	0.6
Total fuel-related	18	0.9
Mileage-related costs		
Local pollution	42	2.0
Congestion	105	5.0
Accidents	63	3.0
Total mileage-related	210	10.0

Source: Parry et al. (2007)

This analysis strongly suggests that the primary drivers for transport sector reforms in high- and middle-income countries will be local benefits. For developing countries the benefits of reducing vehicle pollution emissions and reducing accident rates will be proportionately larger than in the US. Policy instruments for transport reforms include vehicle fuel efficiency and emission standards, investments in public transit infrastructure, and incentives to reduce private car use. On transport, therefore, reduced greenhouse gas emissions are likely to be a co-benefit of efforts to increase local wellbeing by investing in transport sector reforms. Transport reforms will yield climate co-benefits.

Forster et al. (2013) analyse a wide range of external costs and benefits associated with implementation of the 'carbon budget' which is a principal feature of UK climate policy. Table 3.4.2 shows baseline (2012) values of the costs of congestion and accidents per mile travelled, as well as the per mile travelled health benefits of active transport (walking and cycling). Compared with Table 3.4.1 for the US, the UK figures show a similar value for accidents and much higher costs of congestion per mile travelled. The striking figure is the value of health benefits associated with each mile of active transport – this echoes the large benefits reported in Woodcock et al. (2009).

Woodcock et al. (2009) and Forster et al. (2013) derive large benefits that can accrue to improved urban transport design that, in turn, facilitates an increase in active transport. Both note, however, that changes in household behaviour must accompany better design. Given that the majority of people in developing countries currently rely *by necessity* on active travel, it seems unlikely that any reversion to this mode of transport will occur in many developing countries for the foreseeable future.

Table 3.4.2. External costs (-) of vehicle transport and benefits (+) of active transport (walking and cycling) in the UK, 2012	
	US cents/mile travelled
Congestion	-32
Walking/cycling	+74
Accidents	-4

Source: Forster et al. (2013)

Work by the World Bank and ClimateWorks Foundation (2014) simulates a global scenario of increased vehicle efficiency, fuel switching and modal shifts for passengers and freight in 2030. The combined policies reduce CO_2-equivalent emissions by 2.4Gt per year. Compared to business as usual, they calculate the health co-benefits of the specified shifts to be US$36/t$CO_2$e.

3.5 Co-benefits of energy efficiency

While a report from the Intergovernmental Panel on Climate Change identifies energy efficiency as one of the key contributors to limiting global warming to 2°C (IPCC, 2013), investment in the energy efficiency market is lagging – estimated to be US$130 billion in 2012, it represents only 13 per cent of fossil fuel investment (IEA, 2014).

Increasing energy efficiency offers the prospect of multiple benefits to economies and human wellbeing. Assuming that energy efficiency investments are profitable (so their cost is less than the value of the energy saved), one of the major pathways for co-benefits is through cost reductions in the productive sectors of the economy as well as for households. But the list of co-benefits is potentially wider, as Ryan and Campbell (2012) suggest. Table 3.5.1 below lists the range of benefits at different levels of the economy.

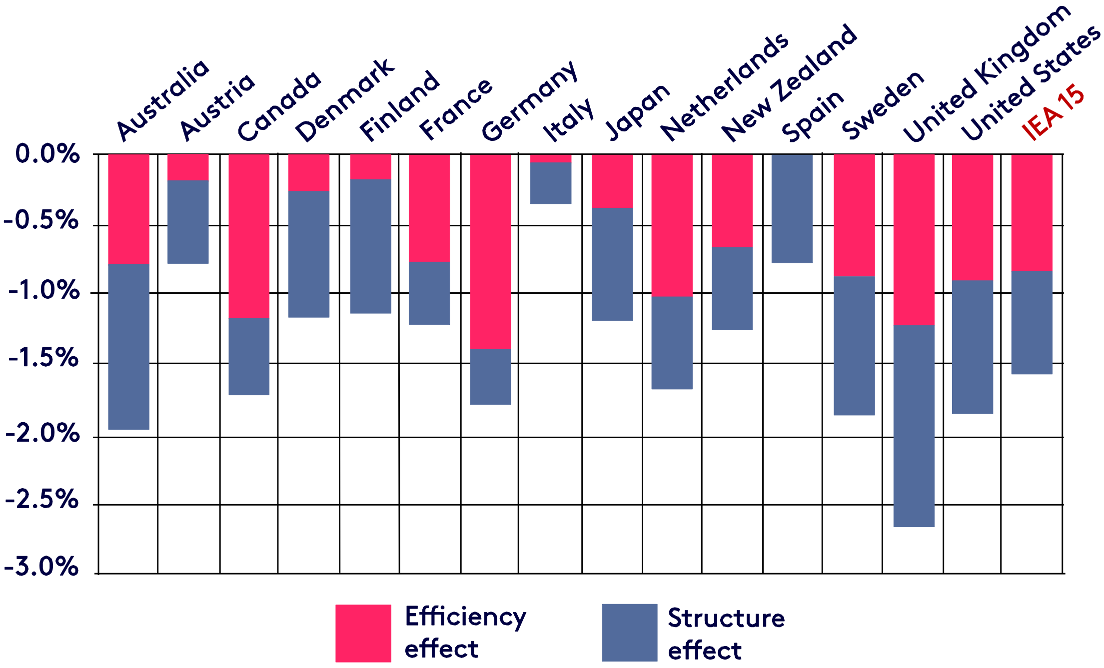
Table 3.5.1. Potential multiple benefits from energy efficiency increases

Individual level	Health and wellbeing impacts Poverty alleviation; energy affordability and access Increased disposable income
Sectoral level	Industrial productivity and competitiveness Energy provider/infrastructure benefits
National level	Job creation (although jobs may be lost as well) Lower energy-related public expenditure Energy security Macroeconomic effects (e.g. increases in real income)
International level	Reduced greenhouse gas emissions Moderating energy prices Natural resources management Meeting development goals for affordable energy

Source: Ryan and Campbell (2012)

Analysis by the International Energy Agency (IEA, 2013b) shows that there was steady improvement in energy efficiency in developed economies from 1990 to 2010. Figure 3.5.1 decomposes the annual change in aggregate energy intensity (primary energy used per dollar of real GDP on a purchasing power parity basis) into the effects of structural change in the economy, as well as pure energy efficiency. As the graph shows, average energy intensity decreased by close to 2 per cent per year in many countries from 1990 to 2010; roughly half of this was due to structural change in the economy, the rest to pure energy efficiency.

Figure 3.5.1. Decomposition of average annual change in energy intensity, 1990–2010

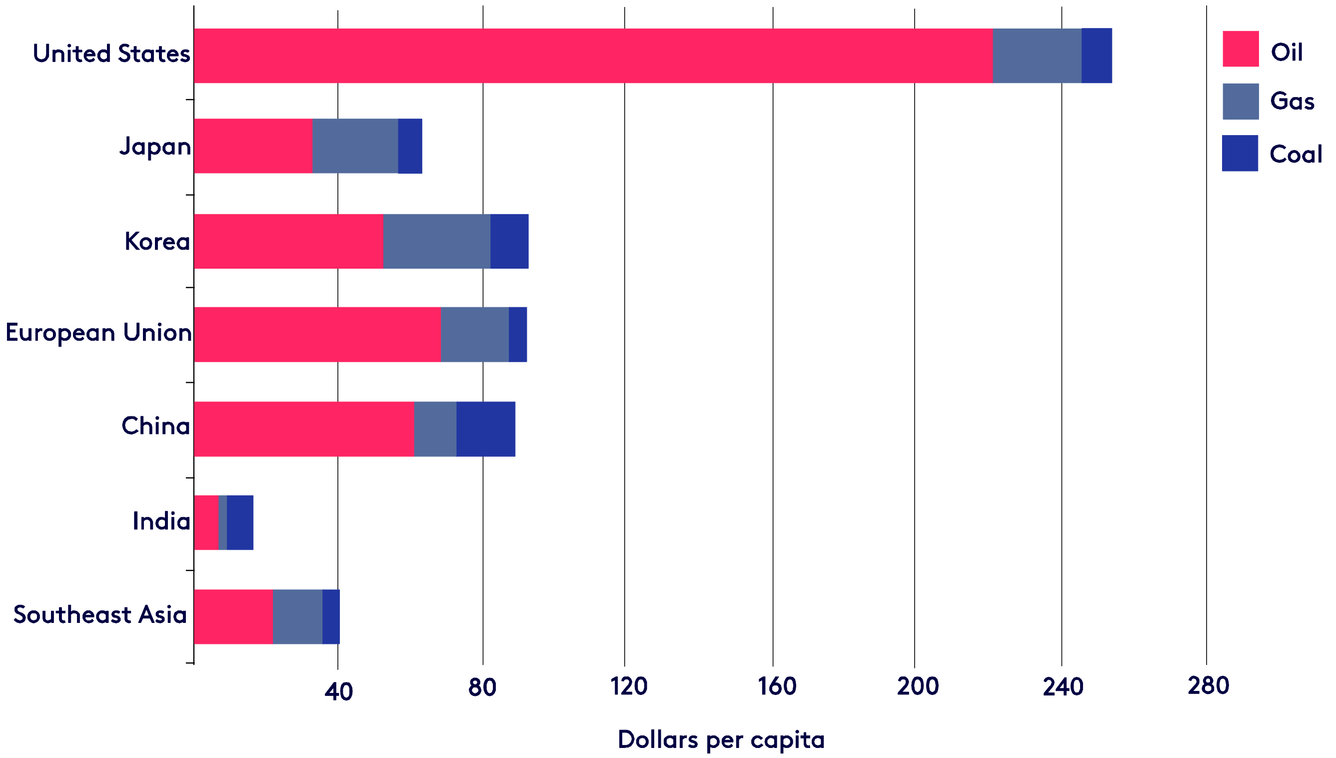


Notes: Efficiency effect represents the composite economy-wide adjusted energy intensity metric. IEA 15 member countries are those for which sufficient data is available to undertake analysis.
Source: Reproduced from OECD/IEA (2013). © OECD/IEA

Energy security benefits

Since increasing energy efficiency can substitute for energy imports, it can have a tangible effect on the security of energy supply. IEA (2013b) simulates three basic scenarios: the Current Policy scenario (BAU), the New Policies scenario (multiple reforms across the energy sector, based on recent government announcements), and the Efficient World scenario (accelerated investments in energy efficiency worldwide). Figure 3.5.2 shows the reductions in the value of energy imports per capita in major countries and regions as a result of the energy efficiency provisions of the New Policies scenario. While energy efficiency can be an effective substitute for imported energy, governments need to be wary of 'green protectionism', as discussed in Section 3.2. In addition, reductions in imports can have general equilibrium effects since, other things being equal, this will cause the exchange rate to rise, which in turn makes imports cheaper while penalising export sectors.

Figure 3.5.2. Reduced import bills per capita from energy efficiency actions under the New Policies scenario relative to the Current Policy scenario, 2035

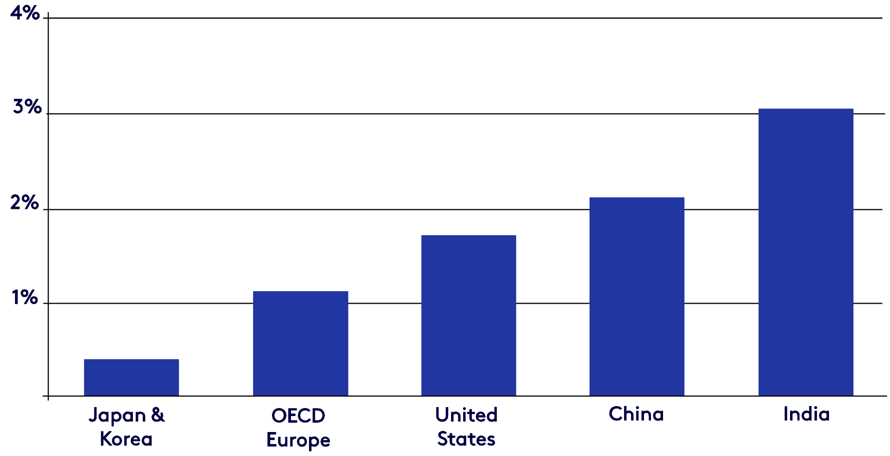


Source: Reproduced from IEA (2013b). © IEA

Macroeconomic benefits

Because energy efficiency investments decrease costs (assuming these investments meet cost–benefit criteria), this affects the efficiency of the economy across multiple sectors, resulting in potential increases in real GDP. IEA (2013b) compares real GDP in 2035 by country/region in the New Policies scenario and the Efficient World scenario, as shown in Figure 3.5.3. While the difference in the scenarios is small for Japan and Korea, it varies from 1–3 per cent across Europe, the US, China and India.

Figure 3.5.3. Change in real GDP in the Efficient World Scenario compared with the New Policies Scenario, 2035

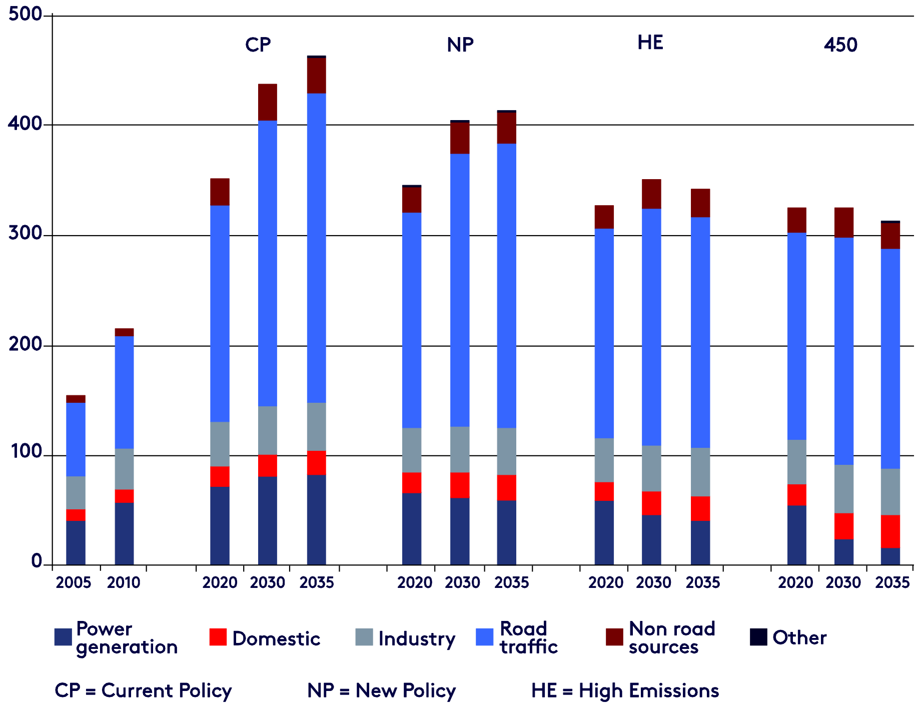


Note: Investment in energy efficiency of US$12tn is more than offset by fuel savings and triggers cumulative economic growth of $18tn. Source: Reproduced from Baroni (2013). © OECD/IEA

Air pollution control costs

Because energy efficiency is a zero-emission source of energy supply, this implies that some amount of pollution control expenditure can be foregone in scenarios featuring increased energy efficiency. The IEA Current Policy and New Policies scenarios assume a given degree of abatement of air pollution emissions by country over time. Figure 3.5.4 looks at world expenditures on pollution abatement by policy scenario and sector. (The 450 scenario 'sets out an energy pathway consistent with … limiting the global increase in temperature to 2°C by limiting concentration of greenhouse gases in the atmosphere to around 450 parts per million of CO_2' [IEA, 2017].)

Figure 3.5.4. Air pollution control costs for the World Energy Outlook 2012 scenarios by sector, billion €/year



Source: Cofala et al. (2012). Reproduced with permission from the International Institute for Applied Systems Analysis (IIASA)

Comparing the High Efficiency (HE) or Efficient World scenario with New Policies (NP) scenarios, the figure shows that the suite of policy differences between the two scenarios, mostly linked to energy efficiency, results in nearly €50 billion in savings on pollution control in 2030. The sectors most affected are transport and power generation.

Health impacts

As a zero-emission source of energy, energy efficiency not only reduces abatement costs, it also decreases pollution emissions and pollution exposure.[14] IEA (2012) models the effects of policy choices on excess mortality from air pollution exposure, and Table 3.5.5 compares years of life lost between the Current Policy and Efficient World scenarios.

[14] This link between pollution, pollution abatement and zero-emission energy is discussed in fuller detail in Section 5 below.

Table 3.5.5. Life years lost owing to exposure to anthropogenic emissions of PM2.5, millions									
	Current Policy scenario			Efficient World scenario			Co-benefits of energy efficiency		
Year	2015	2025	2035	2015	2025	2035	2015	2025	2035
China	1461	1483	1481	1422	1330	1247	39	153	234
India	761	1142	1594	730	977	1174	31	165	420
Russia	55	52	54	54	48	49	1	4	5
EU	146	115	112	141	105	98	5	10	14

Note: Russia includes only the European part. Source: IEA (2012)

As expected, co-benefits in terms of life years saved as a result of energy efficiency policies are very high in heavily polluted middle-income countries such as China and India. This reflects the high contribution that energy *inefficiency* makes to pollution emissions in these countries. The effect is much more muted in the EU, where energy efficiency is generally much higher and pollution emissions are already heavily regulated.[15]

The World Bank and ClimateWorks Foundation (2014) simulate a global scenario with fuel switching as well as a range of energy efficiency measures both in industry and buildings in 2030. The combined policies reduce CO_2e emissions by 6.1Gt per year. The calculated health co-benefits of these efforts amount to \$56/tCO_2e compared with business as usual.

Fossil fuel subsidies

While fossil fuel subsidies raise many issues – equity and fiscal impacts, in particular – they are of great interest to the energy efficiency discussion because they actually encourage the consumption of energy.

Globally, fossil fuel subsidies are large, widespread, and constitute the almost exact opposite of energy efficiency policies. They represent a fiscal drain, costing money rather than saving it. They reduce incentives to economise on the use of energy. By encouraging over-consumption, they lead to a range of co-costs in the form of increased health impacts, increased energy security concerns, and decreased economic efficiency. While theoretically aimed at assisting the poor, only 8 per cent of fuel subsidies reach the poorest 20 per cent of the population, by IEA's (2011) calculations.

According to IEA (2011), global fossil fuel subsidies amounted to US$409 billion in 2010 and will expand, in the absence of reforms, to US$660 billion by 2020, or 0.7 per cent of gross world product. By 2012 these subsidies totalled US$544 billion (IEA, 2012). The scale of subsidies is such that 15 per cent of global CO_2 emissions are now caused by over-consumption of energy driven by these policies, according to IMF research (Clements et al., 2013).

The logical starting point for energy efficiency policies is therefore the reform of fossil fuel subsidies.

Rebound effects

Increasing energy efficiency can have impacts on consumers and the economy because standard micro-economic theory suggests that consumers will consume more of a good if its price falls. For example, if cars become more energy efficient, then the cost of transport falls, other

[15] Note that the estimates of life years lost in IEA (2012) represent the cumulative benefits over the average remaining years of life for the different populations.

things being equal. Consumers could be expected to use their cars more because the cost of operation has fallen. This is termed the 'rebound effect' of increased efficiency.

Alternatively, consumers could choose to drive the same number of kilometres but would realise an increase in real income because the cost of these kilometres has fallen. In this instance the increased income could be spent on other goods and services, which in turn would require increased use of energy inputs into their production and use.

Finally, there could be a price effect of increased vehicle fuel efficiency. If the total demand for gasoline/petrol falls as a result of increased efficiency, then the price of energy would be expected to fall, stimulating higher use of energy across the economy.

All of these examples of the rebound effect imply that the *net* energy saving from increasing the fuel efficiency of cars will be less than would be calculated by simply multiplying energy savings per kilometre times the (pre-reform) total kilometres travelled in cars.

Gillingham et al. (2013) classify these different outcomes as *direct* rebound effects, *indirect* rebound effects, and *macroeconomic* price effects respectively. They then assess the empirical literature on rebound effects, focusing particularly on driving and residential electricity use, for which recent data is abundant. They conclude that direct rebound effects from increasing fuel efficiency could amount to 5–20 per cent of the expected gross reduction in energy used, while indirect rebound effects could be in the order of 5–15 per cent, and that rebound effects are real, but not sufficient to fully offset the effects of efficiency increases in the economy. Moreover, at some level these effects are welcome, because they are driven by increases in consumer wellbeing associated with efficiency increases.

Conclusions on the co-benefits of energy efficiency

As a zero-emission energy source, energy efficiency has clear co-benefits in the form of reduced health impacts, reduced air pollution control costs, potential increases in energy security, and macroeconomic growth benefits as a result of increased efficiency. However, this conclusion needs to be qualified in the following ways:

- Energy efficiency projects need to be profitable, so that financial costs are less than the value of energy saved.
- Fossil fuel subsidies are effectively policies fostering energy inefficiency. The prevalence of these subsidies can mute the impact of policies aimed at increasing energy efficiency.
- Rebound effects from energy efficiency increases are real and may be large. But these rebound effects generally reflect increases in consumer wellbeing as a result of energy efficiency policies.

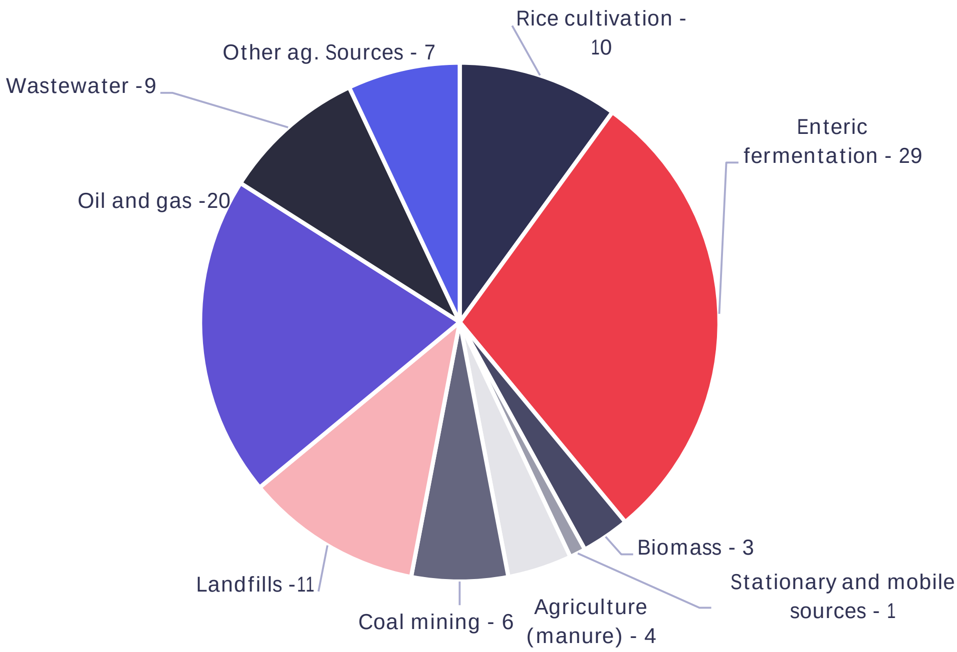
3.6 Methane and black carbon

Methane (CH_4) and black carbon (BC), along with tropospheric ozone and hydrofluorocarbons (HFCs), known as short-lived climate pollutants (SLCPs), are important contributors to anthropogenic climate change, responsible for about one-third of the current total greenhouse forcing (Shoemaker et al., 2013, 2010). Controlling SLCPs could roughly halve projected warming over the next few decades (CCAC/UNEP, 2014). Because black carbon and methane emissions are at historically high levels, action on mitigating them will save millions of lives and increase crop yields by tens of millions of tons annually via improved air quality.

UNEP and WMO (2011) estimates that reducing black carbon, through the widespread adoption of advanced cook stoves and clean fuels, has the potential to prevent over two million premature deaths each year, while methane recovery from landfill gas and coal mines could avoid the annual loss of more than 30 million tons of crops. Compared with CO_2 mitigation, the most substantial benefits of SLCP mitigation will be felt immediately in and around the regions where action is taken to reduce emissions. Asia is the greatest source of SLCPs and would reap the greatest health and crop benefits from mitigation (US EPA, 2012; IIASA, 2009; UNEP and WMO, 2011). While these reports speak to the potential benefits from controlling SLCPs, Shindell et al. (2012) simulate specific levels of control in 2030 and estimate the resulting benefits (see below).

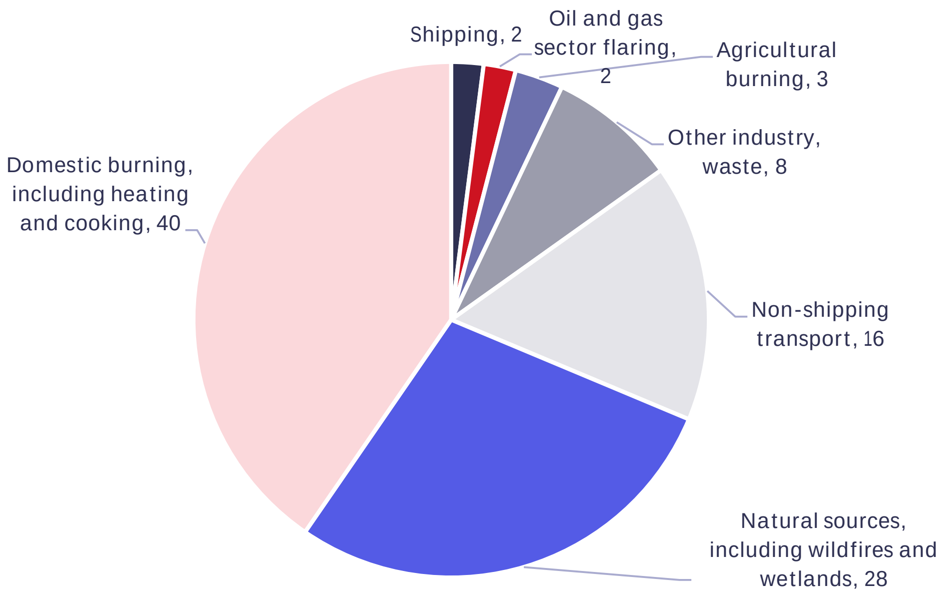
Figures 3.6.1 and 3.6.2 summarise the sources of global emissions of methane and black carbon.

Figure 3.6.1. Estimated global anthropogenic methane emissions by source (%), 2010



Source: Based on Global Methane Initiative (2010)

Figure 3.6.2. Global breakdown of black carbon emissions by source (%), 2010



Source: Based on Arctic Monitoring Assessment Programme (AMAP) (2015)

Modelling short-lived climate pollutants (SLCP) reductions

Shindell et al. (2012) simulate a package of measures aimed at reducing emissions of short-lived climate pollutants (SLCPs). The business as usual (BAU) baseline is an IEA projection of CO_2 emissions (IEA, 2009) which excludes SLCPs. Shindell et al. model emissions of SLCPs based upon the IEA's BAU projection and for their policy scenario, out of 400 potential mitigation measures, narrow this down to a set of 14 measures which both reduce emissions of SLCPs and provide local air pollution benefits: seven deal with methane, while the other seven reduce emissions of black and organic carbon. Measures are chosen based on their cost-effectiveness in reducing short-run climate forcing. These are detailed in Table 3.6.1.

Table 3.6.1. Measures adopted to reduce SLCP emissions cost-effectively

Measure	**Sector**
Methane (CH_4) measures	Extraction and transportation of fossil fuels
Extended pre-mine degasification and recovery and oxidation of CH_4 from ventilation air from coal mines	
Extended recovery and utilisation rather than venting of associated gas and improved control of unintended fugitive emissions from the production of oil and natural gas	
Reduced gas leakage from long-distance transmission pipelines	
Separation and treatment of biodegradable municipal waste through recycling, composting and anaerobic digestion as well as landfill gas collection with combustion/utilisation	Waste management
Upgrading primary wastewater treatment to secondary/tertiary treatment with gas recovery and overflow control	
Control of CH_4 emissions from livestock mainly through farm-scale anaerobic digestion of manure from cattle and pigs	Agriculture
Intermittent aeration of continuously flooded rice paddies	

Black carbon 'tech' (technology) measures	
Diesel particle filters for road and off-road vehicles as part of a move to worldwide adoption of Euro 6/VI standards	Transport
Introduction of clean-burning biomass stoves for cooking and heating in developing countries	Residential
Replacing traditional brick kilns with vertical shaft and Hoffman kilns	Industry
Replacing traditional coke ovens with modern recovery ovens including the improvement of end-of-pipe abatement measures in developing countries	
Black carbon 'reg' (regulatory) measures	
Elimination of high-emitting vehicles in road and off-road transport	Transport
Ban on open burning of agricultural waste	Agriculture
Substitution of clean-burning cookstoves using modern fuels (LPG or biogas) for traditional biomass cook stoves in developing countries	Residential

Sources: Shindell et al. (2012) and supplementary material

This suite of measures results in a simulated reduction in methane emissions of 139Tg (teragrams; i.e. 139 million tonnes) in 2030, a 40 per cent reduction from BAU. The combined black carbon measures lead to a simulated emission reduction in 2030 of 6.25Tg carbon, roughly a 75 per cent reduction from BAU. The short-term benefits modelled include reduced ozone exposure, both for crops and people. Excess mortality from local air pollution is calculated using a linear concentration-response function, building on the work of Krewski *et al*. (2009). Indoor air pollution mortality is modelled only for China and India. Excess deaths are valued using a 2030 estimate of value of a statistical life (VSL) for the US of US$9.5 million, which is then transferred to other countries using an assumed income elasticity of 0.4. Ozone damages to crops are based upon Van Dingenen et al. (2009).

The suite of SLCP control measures produces avoided warming of 0.47°C in 2050, valued at US$556 billion in 2030, as seen in Table 3.6.2. Across the world the co-benefits of reducing ozone exposure and exposure to PM2.5 from black carbon control measures in 2030 amount to avoided crop losses of US$8.2 billion and reduced mortality valued at US$5,290 billion, or roughly 5 per cent of gross world product. Country estimates of avoided deaths are substantial, as shown in Table 3.6.3, and are predominantly situated in Asia.

Table 3.6.2. Monetary benefits of short-lived climate pollutant (SLCP) control in 2030, US$bn

	Methane	Black carbon	Total SLCPs
Climate	331	225	556
Crops	4	4	8
Health	148	5,142	5,290
Total	**483**	**5,371**	**5,854**

Source: Shindell et al. (2012)

Table 3.6.3. Number of avoided deaths in selected countries from short-lived climate pollutant (SLCP) control in 2030	
India	814,000
China	684,000
Bangladesh	159,000
Pakistan	91,000
Indonesia	89,000

Sources: Shindell et al. (2012) and supplementary material

As Table 3.6.2 makes clear, local co-benefits consisting of decreased crop losses and improved health are roughly 10 times larger than climate benefits.[16] The health benefits from black carbon mitigation are over 30 times larger than the health benefits from methane mitigation. The *values* of increased crop production in Shindell et al. (2012) are relatively small, but the *increased quantities* produced and *yields* are substantial, as seen in Table 3.6.4.

Table 3.6.4. Increased crop production in 2030 as a result of reduced ozone exposure			
Increased crop production (wheat, rice, maize, soy), tonnes, top 10 countries		Increased crop yield (wheat, rice, maize, soy), %, top 10 countries	
China	15,744,000	Kuwait	8.0%
India	9,775,250	Iran	6.6%
United States	6,305,319	Jordan	6.5%
Pakistan	2,134,840	Israel	5.8%
Brazil	1,640,395	Pakistan	5.6%
Mexico	1,135,236	Armenia	4.8%
France	1,123,970	Lebanon	4.7%
Turkey	1,043,475	Malta	4.2%
Iran	1,022,335	Kyrgyzstan	4.2%
Egypt	948,971	Mexico	4.1%

Sources: Shindell et al. (2012) and supplementary material

Conclusions on co-benefits from SLCP reduction

Control of SLCPs can yield significant short-run warming benefits, but this does not reduce the need to reduce CO_2 emissions. While CO_2 has a weaker short-run impact on warming, it resides in the atmosphere for more than a century, leading to longer-run warming. Country efforts on SLCP mitigation will likely target impacts on health and crop production, but there will be large short-run climate co-benefits as a result of these efforts. The potential impacts of SLCP mitigation are large: over US$5 trillion of health benefits in 2030, combined with significant increases in crop production.

[16] The social cost of carbon used is roughly US$10 per ton of CO_2-equivalent.

4. Empirical estimates of co-benefits – model results

Climate mitigation actions, particularly those focused on changes in energy mix and energy efficiency, are widely acknowledged to produce co-benefits in the form of reductions in local air pollution emissions. This emission reduction occurs because of existing inefficiencies in policies, in particular the failure of environmental policies in most countries to optimally control emissions based upon the costs and benefits of controlling local air pollution. Emissions of fine particles (PM2.5) are particularly damaging to health, leading to excess cancers and cardiopulmonary disease, while ground level ozone exposure has acute health impacts as well as damaging crops.

This section reviews and assesses three sets of model results concerning local air pollution co-benefits (following the focus in Section 3.6 above on methane and black carbon).

4.1 Public health benefits of strategies to reduce greenhouse gas emissions: low-carbon electricity generation (Markandya et al., 2009)

Markandya et al. model the co-benefits of shifting electricity supply globally towards low-carbon technologies. They combine an emissions model for CO_2-equivalent and local pollutants, including CO_2e abatement costs, with an emission dispersion model to measure local air pollution changes. The simulations include a business as usual (BAU) baseline with no climate change mitigation, plus two mitigation scenarios where emission rights are either partially or fully tradable. The low-carbon technologies employed under mitigation policies include substitution of coal-electricity generation by nuclear and renewables, and CO_2 emissions mitigation via carbon capture and storage (CCS). The analysis focuses on the EU, China and India.

The BAU scenario includes an environmental Kuznets curve for air pollution which shows that increasing incomes eventually lead to mitigation of local pollution emissions. From 2010 to 2030, PM2.5 concentrations under BAU drop from 8.8ug/m^3 to 6.9ug/m^3 in the EU, from 88 to 81ug/m^3 in China, and increase from 55 to 108 ug/m^3 in India (reflecting the country's lower starting point in terms of income per capita). Under full carbon trading in 2030, CO_2 emissions in the EU drop from 1.7Gt to 0.7Gt, in China from 5.5Gt to 1.4Gt, and in India from 2.0Gt to 0.4Gt.

Damages from PM2.5 are assessed using the year 2000 World Health Organization comparative risk assessment, yielding an assumed log-linear concentration-response function (CRF) for excess mortality from cardio-vascular disease and lung cancer.

Damages are valued for all countries assuming the EU standard value of a statistical life (VSL) of roughly US$1.8 million;[17] there is no variation in VSL with income per capita, based in part on an assumed convergence of incomes across countries over the course of the century.

The results of the analysis are shown in Table 4.1. While Markandya et al. aim to establish the significant public health benefits derived from shifting electric power generation away from fossil fuels, the key message from the viewpoint of co-benefits is that the net costs of CO_2 abatement in India, after accounting for local pollution damages, are negative.

[17] Note that this value of a statistical life is low compared with the work presented in OECD (2014).

Table 4.1. Years of life lost per million population and net cost of CO_2 abatement in 2030		
	Years of life lost per million population	Net cost of CO_2 abatement relative to BAU (US$ per ton CO_2 abated), full trade
EU	104	135.0
China	542	61.2
India	1,492	-4.1

Source: Markandya et al. (2009)

4.2 Co-benefits of mitigating global greenhouse gas emissions for future air quality and human health (West et al., 2013)

West et al. (2013) model air pollution co-benefits in Representative Concentration Pathway (RCP) 4.5, a scenario which achieves a least-cost path to climate forcing of 4.5 Watts/m^2 in 2100. This corresponds to a 66 per cent probability of limiting warming to 2.4°C. The following analysis builds on the online supplementary material from West et al..

As the end of the century is reached in RCP 4.5, the electricity sector is carbon-negative – all fossil fuels have been phased out and CCS is used for any biomass sources of electric generation. This has the co-benefit of eliminating many sources of PM2.5, the major source of air pollution mortality. West et al. use coefficients to simulate emissions of PM2.5 in RCP 4.5 compared with their reference BAU scenario without climate policy; the BAU scenario does include an environmental Kuznets curve mechanism whereby countries start to control PM2.5 emissions as they get richer. Dispersion of PM is modelled spatially and overlaid with spatial population projections to arrive at human exposure measures. Concentration-response functions (CRFs) are used to estimate reductions in deaths relative to BAU, and projected values of a statistical life (VSL), varying by country over time, are used to value the economic benefits of reduced pollution mortality (see Appendix 1).

CRFs are built upon a large cohort study carried out by the American Cancer Society (ACS) in the 1970s and 80s. Given that concentrations of PM2.5 did not exceed roughly 30ug/m^3 in the ACS study, there is a question of how to extrapolate from this study to developing countries where concentrations may exceed 100ug/m^3. In addition to a linear extrapolation of the CRFs built on the ACS data, West et al. also use CRFs with an assumed lower threshold of 5.8ug/m^3, an upper threshold of 30ug/m^3, and a log-linear specification of the CRF, which flattens the dose-response relationship at higher concentrations but also boosts the sensitivity to lower concentrations compared with the linear CRF.

The log-linear specification is arguably closest to research on PM mortality by Pope et al. (2011), which builds on the literature on second-hand cigarette smoke to understand the effects of high concentrations of particulate matter. Pope et al. find that cardiovascular deaths level off quickly as PM2.5 concentrations rise, but cancer deaths rise linearly and reach very large relative risks of 30 or more at higher concentration levels (i.e. risks of cancer mortality are 30 times higher in the exposed population compared with an unexposed population).

As shown in Table 4.2, avoided deaths under the high concentration threshold yield the lowest number of avoided deaths, but the evidence in Pope et al. (2011) suggests that this is probably an overly-conservative CRF.

Table 4.2. Avoided deaths in 2050 under alternative concentration response functions (RCP 4.5 versus business as usual)				
	Linear	Lower concentration threshold	Log-linear	Higher concentration threshold
Africa	81,600	79,600	48,600	37,600
Australia and NZ	222	205	402	220
Canada	5,750	2,220	14,100	5,750
China	430,000	430,000	337,000	351,000
Eastern Europe	8,220	8,220	12,600	8,220
Former USSR	54,100	51,900	65,600	44,400
India	235,000	235,000	77,000	31,700
Japan	9,190	9,190	12,700	9,190
Latin America	25,100	18,100	48,300	25,100
Middle East	8,260	8,260	2,270	0
South Korea	3,180	3,180	3,320	3,180
Southeast Asia	204,000	202,000	99,900	46,300
United States	29,500	14,200	70,000	29,500
Western Europe	32,200	31,700	41,800	31,000
Global	1,130,000	1,090,000	834,000	623,000

Source: West et al. (2013) and supplementary online material

The value of a statistical life (VSL) used in West et al. is obtained for different regions and different periods using US estimates from the 1990s. The US EPA uses a VSL of US$4.8 million, measured in 1990 $ for 1990 levels of income. This figure is based on a large number of hedonic wage studies. The meta-analysis in Viscusi and Aldy (2003) suggests an income elasticity of willingness to pay of 0.5.

West et al. 'transfer' the US VSL to different regions and different time periods by using the baseline EPA figure combined with modelled estimates of GDP per capita and the assumed 0.5 income elasticity. This is standard procedure in the literature; it can lead to large VSLs relative to income per capita in very poor countries (Hammitt and Robinson, 2011), but this is less of an issue for projections decades into the future because of the convergence of incomes per capita. (See Hamilton [2014] for further discussion of the income elasticity of the VSL.)

Table 4.3 calculates the value of co-benefits of climate mitigation in 2050 using this analytical apparatus and assuming a CRF that is log-linear in form. This is therefore an extension of the values reported in West et al., who use a linear CRF and report correspondingly larger co-benefits.

Table 4.3. Regional co-benefits of climate mitigation versus CO_2 price in 2050, RCP 4.5					
	Deaths avoided (log-linear CRF)	CO_2 abated (million t)	Value of a statistical life (US$ m)	Co-benefit (US$/t$CO_2$)	CO_2 price (US$/t CO_2)
Africa	48,600	1,920	1.9	47	22
Australia and NZ	402	151	14.3	38	22
Canada	14,100	439	9.5	307	22
China	337,000	5,210	5.3	342	22
Eastern Europe	12,600	356	5.9	210	22
Former USSR	65,600	934	3.6	250	22
India	77,000	2,290	3.0	102	22
Japan	12,700	285	12.3	547	22
Latin America	48,300	1,220	4.7	185	22
Middle East	2,270	678	4.6	15	22
South Korea	3,320	121	7.4	203	22
Southeast Asia	99,900	2,390	4.2	175	22
United States	70,000	1,720	12.4	503	22
Western Europe	41,800	1,160	9.1	327	22
Global	834,000	18,900	4.6	205	22

Sources: Authors' calculations based on West et al. (2013) and supplementary online material

The value of the co-benefit per ton of CO_2 abated is calculated as deaths avoided as a result of CO_2 abatement, measured relative to business as usual, multiplied by the VSL and divided by tons of CO_2 abated. This is compared with the price of CO_2 in 2050 calculated in RCP 4.5. With the exception of Africa, Australia and New Zealand, and the Middle East, co-benefits exceed the marginal cost of abatement by very wide margins.

Figures 4.1 to 4.3 below compare co-benefits with the price of CO_2 in 2030, 2050 and 2100. In each year the bar representing the CO_2 price is second or third from the bottom, indicating that for the great majority of regions/countries the value of co-benefits from climate mitigation exceed marginal CO_2 abatement costs – and generally by a very wide margin.

RCP 4.5 exhibits relatively low carbon prices (US$9.60, $22.00, and $85.20 in 2030, 2050 and 2100 respectively) owing to the fact that 4.5W/m^2 is not a particularly stringent abatement target, and to a heavy reliance on land and forestry interventions in order to keep costs low. Nevertheless, Figures 4.1 to 4.3 show that carbon prices well in excess of US$100 would be required before they would exceed the majority of regional co-benefits.

Figure 4.1. Co-benefits per tonne CO_2 vs. CO_2 price, 2030

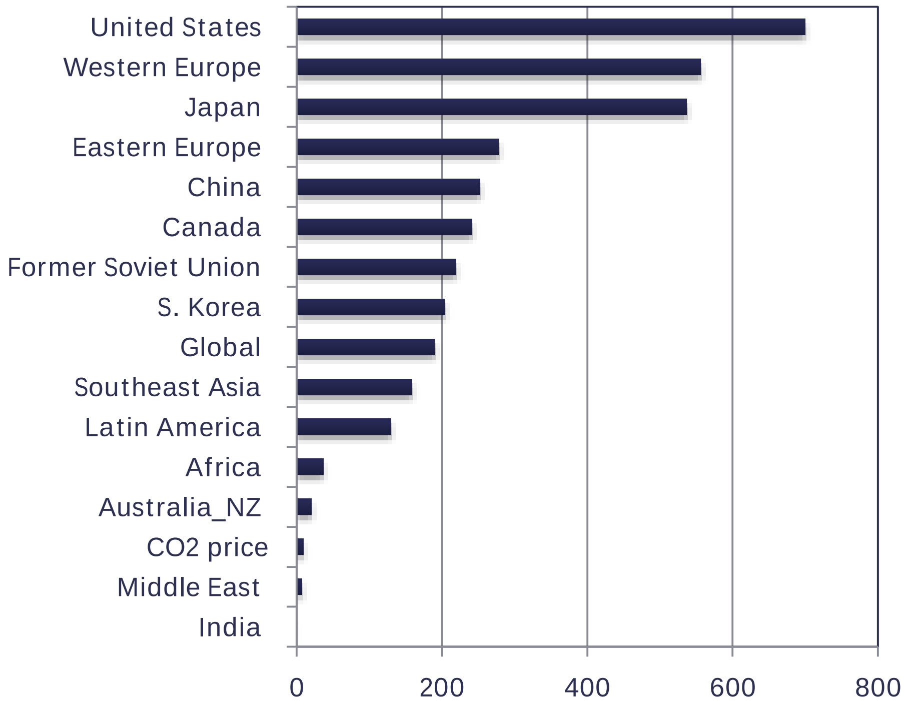


Figure 4.2. Co-benefits per tonne CO_2 vs. CO_2 price, 2050

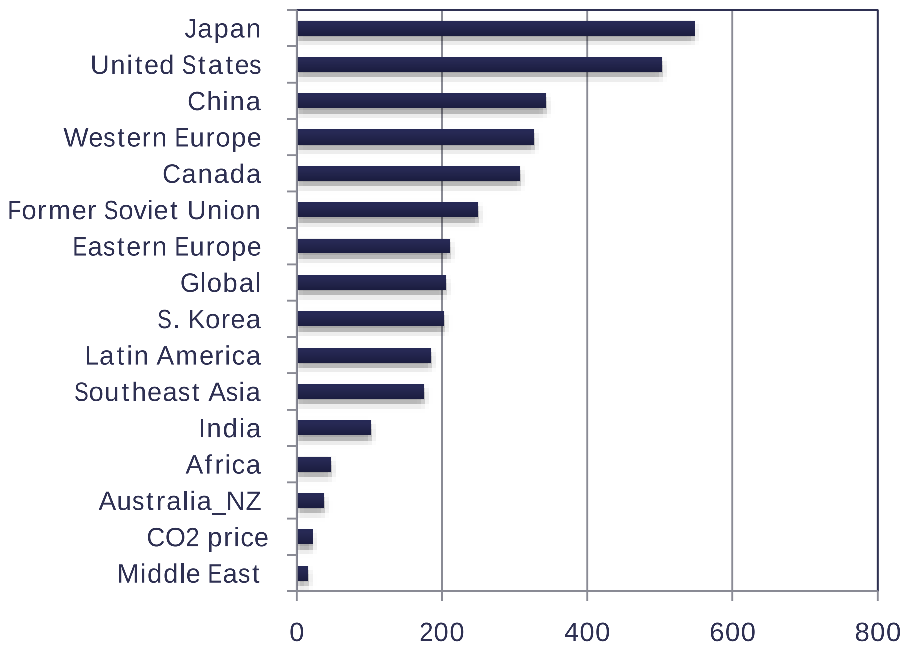


Figure 4.3. Co-benefits per tonne CO_2 vs. CO_2 price, 2100

Source for Figures 4.1–4.3: Authors' calculations

4.3 The reduction in air quality impacts and associated economic benefits of mitigation policy: summary of results from the EC RTD Climate Cost Project (Holland et al., 2011)

The analysis of Holland et al. is narrower in its geographic coverage than West et al. (2013), covering the European Union (27 countries at the time of the study) plus China and India, but its coverage of the outcomes valued is broader, spanning damages to health, materials, crops and avoided local pollution abatement costs. The health benefits include morbidity as well as mortality. Outcomes measured in physical terms include acidification of forests and other ecosystems and nutrient nitrogen loadings.

The policy scenario is the Ensembles E1 scenario, a least cost path to 2°C warming by the end of the century, with CO_2 emissions reduced by 60 per cent globally in 2050. It is updated with recent data on air pollution emissions from POLES and updated trajectories for EU legislation on air pollution up to 2050. The BAU scenario is SRES A1B (the IPCC's Special Report on Emissions Scenarios A1B – a balanced emphasis on all energy sources).

Focusing on the EU creates two distinct advantages in the analysis of co-benefits. Because the trajectory of EU regulations on air pollution emissions is known, this can be built directly into the BAU scenario (assuming full compliance, of course). This avoids the need to model an environmental Kuznets curve, reducing one level of uncertainty in the model results. The second advantage is that it is possible to measure avoided pollution abatement costs, because these costs can be measured in the BAU scenario, based on projected EU regulations; as a result of the increased use of zero-emission energy in the policy scenario, some of these abatement costs can be avoided. As Holland et al. (2011) note, as long as care is taken to avoid double counting (the change in pollution abatement costs would be a part of modelled energy system costs), this is a useful result from a policy perspective.

Table 4.4 summarises the value of co-benefits from Holland et al., with mortality valued using a 'mid-range' value of a statistical life of €3.8 million in 2005.[18] It shows that health co-benefits are heavily dominant, but the foregone cost of air pollution abatement is also significant at 13 per cent of the total. Although values of health damages are not reported for India and China (see instead Rafaj et al. [2013]), Holland et al. report that mean life expectancy in India in 2050 increases by 30 months and China by 20 months as a result of the co-benefits of green energy. These changes represent the cumulative gains in life-months over the mean remaining years of life in each country.

Table 4.4. Value of co-benefits in EU countries in 2030, €m 2005, mid-range value of a statistical life (VSL)		
Co-benefit category	**Value (€m)**	**Per cent**
Health	110,305	86.8%
Materials	131	0.1%
Crops	254	0.2%
Pollution abatement	16,400	12.9%
Total	127,090	

Source: Holland et al. (2011)

It is worth comparing the Holland et al. and West et al. results of EU countries, since the analytical framework is very similar in each study. As Table 4.2 shows, West *et al*. estimate a reduction of roughly 30,000 excess deaths in Western Europe in 2050 as a result of their policy scenario. Holland et al. report reductions of 1,374 excess deaths from ozone exposure, and 482,446 life-years saved as a result of reduced deaths from PM2.5 exposure; assuming an average of 10 life-years lost per death from PM2.5 (Public Health England, 2014), this translates into a total reduction of nearly 50,000 excess deaths in 2050. It makes sense that the Holland et al. figure for PM2.5 mortality would be higher because their policy scenario is much more stringent than that of West et al..

Holland et al. estimate a co-benefit of €21/tCO_2 abated in 2030, based on using a value of a statistical life year of €60,000. Re-calculating this figure based on a VSL of €3.8 million, this comes to €115/tCO_2 abated. If the VSL is updated to 2030 using a 2 per cent growth rate in per capita income and an income elasticity of 0.8 (OECD, 2012), then converted into dollars, the result is roughly US$230/t$CO_2$ abated. This compares with over US$500/t$CO_2$ in West et al. for Western Europe – the discrepancy probably reflects the influence of the stricter climate target, 2°C, in Holland et al. (2011).[19]

Discussion of the modelling studies

Table 4.5 provides a useful synthesis of the co-benefits figures for 2030 that can be derived from the most recent model analyses. The global and country-regional figures on air pollution are derived from West et al. (2013). The ranges combine different assumptions on the VSL (low versus high) with different concentration response functions (a high-concentration cap at the low end, a log-linear specification at the high end). Damages to materials and crops, as well as local pollution abatement costs avoided, are derived from Holland et al. (2011). The global figure for

[18] Unlike the other studies considered here, Holland et al. (2011) express preference for an alternative valuation metric, the value of a life year, which leads to more modest estimates of co-benefits (€20bn, rather than the €110bn shown in the table). The VSL-derived estimate shown here is included in the report as part of the uncertainty analysis.

[19] In a more stringent climate scenario (compared with one that is less stringent), less fossil fuel is being burned relative to business as usual, leading to lower health damages, also relative to BAU, while CO_2 abatement relative to BAU is higher.

black carbon co-benefits is imputed from the figures in Shindell et al. (2012) and should be treated with caution.
Roughly speaking, a conservative estimate of air pollution co-benefits in 2030 per tonne CO_2 in high-income countries (Western Europe, US, Japan) would be over US$100, while for middle-income countries (China, Eastern Europe, Former Soviet Union, Southeast Asia) it would be US$50. As seen in Table 1.2, this discrepancy between high- and middle-income countries is largely explained by differences in carbon efficiency (dollars of GDP generated per tonne of CO_2 emitted).

Table 4.5. Summary co-benefits of climate mitigation in 2030, US$/tonne CO_2e

	PM2.5 damages	Ozone damages	Total air pollution	Black carbon	Damage to materials	Damage to crops	Abatement cost avoided
Africa	8 – 31	2 – 6	10 – 36				
Australia & NZ	2 – 9	2 – 6	4 – 15				
Canada	32 – 222	3 – 8	35 – 230				
China	30 – 226	22 – 67	52 – 293				
Eastern Europe	56 – 245	7 – 21	63 – 265				
Former USSR	48 – 206	3 – 9	51 – 215				
India	-8 – -1	20 – 59	12 – 58				
Japan	114 – 390	43 – 128	157 – 517				
Latin America	18 – 103	5 – 13	23 – 117				
Middle East	0 – 6	1 – 5	1 – 11				
South Korea	86 – 252	10 – 31	96 – 282				
Southeast Asia	29 – 94	19 – 54	48 – 149				
United States	103 – 622	13 – 39	116 – 662				
Western Europe	122 – 473	22 – 65	144 – 538				
EU					0.13	0.25	16.40
Global	33 – 167	16 – 47	49 – 214	495 – 3,116			

Notes : 'Abatement cost avoided' refers to local air pollution. Blank cells indicate lack of data.
Sources: Authors' calculations based on West et al. (2013), Holland et al. (2011), Shindell et al. (2012)

The results seen in the above tables and figures obviously build upon a sequence of modelled relationships, each of which are potential sources of uncertainty. Points to consider include the following.

- In the model analyses, the underlying BAU scenario employs an environmental Kuznets curve to ensure that PM2.5 emissions eventually fall as incomes rise. The particulars of the Kuznets curve are not given in West et al., but its use certainly reduces what could be an important source of bias in the co-benefits analysis. Holland et al. rely on prospective EU legislation on air pollution, which provides a sound basis for measuring the true co-benefits of climate action in the EU.

- The calculation of the VSL across countries and across time in the modelling studies is consistent with good practice. This builds on an extensive literature, based on revealed preferences, but there are relatively few developing country primary estimates (Viscusi and Aldy, 2003). Note, however, that there is a new meta-analysis of the stated preference literature on VSL in OECD (2014), and future modelling work would need to take this into account. Note as well the suggestion in Hammitt and Robinson (2011) and World Bank (2016) that an income elasticity of at least 1.0 should be used when transferring VSL from high-income countries to developing countries – this would reduce the West et al. (2013) figures in particular for developing countries.

- There is considerable uncertainty about extrapolating a US-based concentration response function to developing countries with very high ambient concentrations of PM2.5. In the extreme case, the high concentration threshold (HCT) of 30ug/m^3, global deaths are 40 per cent lower in 2050 compared with the linear concentration response function (CRF), as seen in Table 4.2. But both the linear and HCT CRFs are likely the extreme cases of the estimate. Perhaps the most important step towards deepening the analysis of West et al. would be to incorporate Burnett et al.'s (2014) CRF, which deals explicitly with high concentrations of PM2.5, into the analysis. This would bring the analysis into line with the WHO's Global Burden of Disease 2010 report, as well as more recent Global Burden of Disease reports.

- Baseline demography has a potentially large impact on the estimates. West et al. use the International Futures projection (Hughes et al., 2011). As the century progresses towards seeing 9 billion people on the planet, the population growth rate slows and the average age of countries increases – this results in higher all-cause mortality rates, which then boosts the estimate of deaths related to PM2.5 exposure. The other issue is increasing urbanisation, which will substantially increase the proportion of the population that is exposed to elevated levels of PM2.5.

- What of more stringent climate policy compared with RCP 4.5? Table 4.2 gives some sense of how this would play out. The reduction of CO_2 emissions compared with BAU would be higher than in RCP 4.5, which increases the denominator in the co-benefits calculation. But exposure to air pollution would also fall compared with what is seen in Table 4.2, which increases the numerator in the co-benefits calculation. So the effects of more stringent climate policy would to some extent have offsetting impacts on the value of co-benefits. Recent modelling work for the US, however, suggests that while co-benefits of near-term climate policies are commensurate with the costs of these policies, the ratio of co-benefits to costs drops rapidly as climate policies become more stringent (Thompson et al., 2014).

- If the offsetting effects of lower CO_2 emissions combined with lower PM2.5 exposure were perfect, we could assume the country/region damage bars in Figures 4.1–4.3 to be unchanged, while the CO_2 price bar would increase. A CO_2 price somewhere around US$200 in 2100 would lead to large net global co-benefits because the largest emitters (US, Japan, Western and Eastern Europe, China and Southeast Asia) would still have net benefits at this price. India, however, would face a net cost of abatement at this carbon price.

In summary, based on the model analyses reviewed, there is strong evidence that there are net benefits to the world of moderately stringent CO_2 emission reductions. In particular, the largest economies in the world, with further analysis needed for India, could all enjoy net benefits from emission reductions.

Markandya et al. (2009) model a narrow range of countries and limit interventions to the electric power sector. Their results for India are apparently in conflict with West et al. and a closer analysis of the treatment of India and South Asia in the two models is needed. Markandya et al. assume a large increase in PM2.5 emissions in India to 2030 under BAU, which are reduced as a result of power sector interventions. The West et al. model produces a net increase in deaths in India in

2030 under the policy scenario RCP 4.5 – this is due to an increase in the use of biofuels in households as a result of the CO_2 price required to minimise global costs of abatement. These factors are part of the discrepancy between the studies, but not necessarily the whole story.

A key point in the model analysis is that the policy scenario in most papers modelling climate outcomes – for example, 4.5W/m^2 climate forcing in the case of West et al. (2013), and 2°C warming in Holland et al. (2011) – is derived by minimising the *financial* costs of reaching the given endpoint. From a co-benefits perspective this is unsatisfactory because *a study focused on co-benefits of climate action should proceed by minimising the total economic costs of achieving the endpoint, including externalities*. This could potentially have a large impact on specific findings, such as the India results of West et al., and would simply be a more self-consistent way of analysing the co-benefits of climate actions.

5. Local pollution: abate or go green?

The model results of West et al. (2013) and Holland et al. (2011) suggest that there are major co-benefits from switching to zero-emissions technologies such as wind generation, which eliminate to a substantial degree (recognising that there will be some emissions linked to manufacture and installation of turbines and ancillary equipment) both local pollutants and CO_2 emissions. The co-benefits derive primarily from the elimination of local pollution. These results raise an important policy question: if countries are not primarily interested in reducing greenhouse gas emissions, should they aim to increase welfare by abating local pollution through classic ‘end-of-pipe’ measures, which yield no CO_2 abatement or could it be cheaper to move to zero-emission green technologies, which yield both local and global climate benefits?

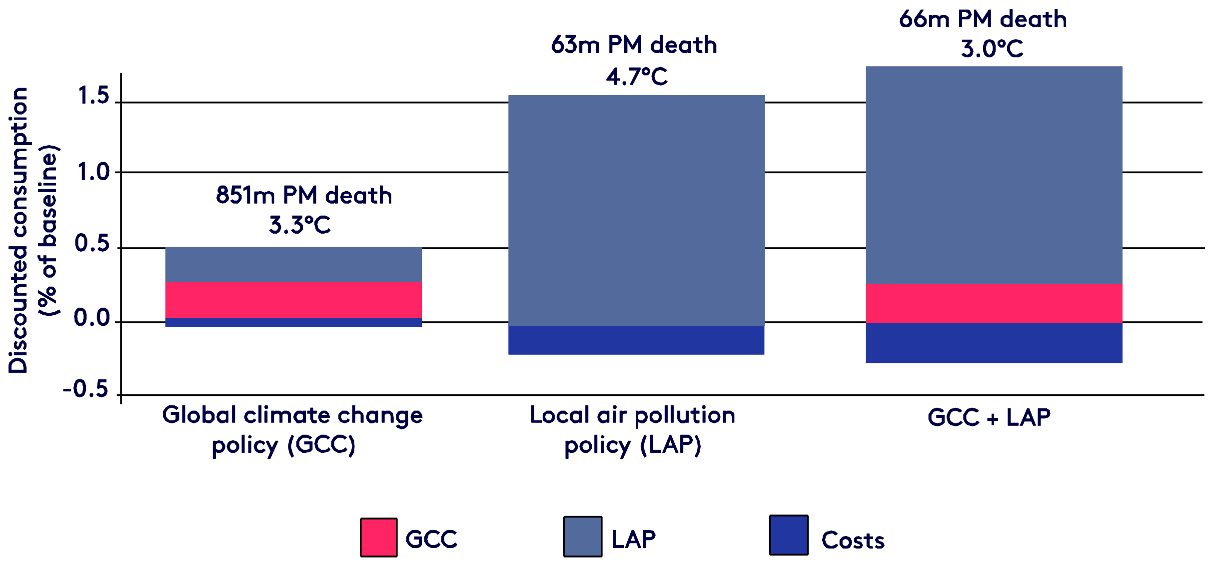
As presented in Appendix II, answering this question builds upon the analytical framework for optimal pollution. Both theory and measurement suggest that marginal damages from pollution rise with increasing emissions, while abatement costs rise with increasing levels of abatement. Efficient pollution policies will therefore decrease emissions up to the point at which marginal costs and damages are equated. The analysis in Appendix II shows that zero-emission technologies have two cost advantages over fossil fuels – they both reduce the need for costly pollution abatement and reduce the damages from any pollution emissions which remain after marginal costs and benefits are equalised.

5.1 Modelling decisions to abate both local pollution and carbon dioxide

In the context of the ‘abate or go green’ decision it is worth returning to a point made in the Introduction. Since fossil fuel-dependent countries face two externalities – local air pollution and CO_2 – the efficient way to reduce these externalities is to employ two separate policy instruments, for example an emissions tax on local pollution and a separate emissions tax on CO_2.

Bollen et al. (2009) analyse this question of two externalities using a global optimising integrated assessment model. They run three policy scenarios running over this century: (i) a policy to internalise the carbon externality (labelled GCC for ‘global climate change’), (ii) a policy to internalise the local pollution externality (labelled LAP for ‘local air pollution’), and (iii) a policy to internalise both externalities (labelled GCC+LAP). The data they use to model the abatement of local air pollution suggests that end-of-pipe treatment of local pollution is relatively low cost.

Figure 5.1. Benefits of three policy scenarios on climate change and local pollution



Note: GCC+LAP = policy to internalise both externalities
Source: Reproduced from Bollen et al. (2009) with permission

The welfare measure in Bollen et al. is discounted consumption, so Figure 5.1 shows the present value of benefits of the policies as a percentage of total welfare. Policy GCC produces a climate benefit of roughly 0.2 per cent of welfare, with an associated local air pollution co-benefit of 0.3 per cent of welfare. Policy LAP produces an air pollution benefit of 1.5 per cent of welfare, but virtually no climate benefit. The combined policy produces a climate benefit of 0.3 per cent of welfare and local air pollution of nearly 1.5 per cent of welfare. Overall benefits are highest with the combined policy, as theory would suggest.

5.2 Conclusions on the 'abate or go green' decision

The marginal analysis in Appendix II highlights a constant economic cost advantage of zero-emissions energy over fossil fuel that is independent of the degree of stringency of air pollution policy. This advantage may be large if pollution abatement is sufficiently costly and/or pollution damages are sufficiently large at low emission levels. These can be considered structural cost advantages enjoyed by zero-emissions energy.

There is also a *policy-sensitive* economic cost advantage of clean energy that depends on the extent to which local pollution is currently being abated. This cost advantage will be large in highly polluted countries such as India and China where pollution emissions, at least until recently, go largely unabated. For these countries zero-emissions energy may be the preferred way to abate local pollution at the margin.

While the economic costs of zero-emission energy may be less than those for fossil fuel-based electricity, the financial costs may well be higher. In this instance financial profitability at the prevailing price of electricity may be lower than for coal-electric generation. This cost may be borne by the electric power sector, or by the public sector via subsidies. Raising prices could restore profitability, but at the cost of a fall in consumer surplus.

Finally, the analysis of combined policy interventions for climate change and local air pollution in Bollen et al. (2009) points a useful way forward for policymakers grappling with these two problems. Rather than casting the question as purely 'abate or go green', the policy intervention aims to internalise both externalities in an optimising framework. The combined policy produces the highest increment to welfare, producing more climate mitigation benefits than the optimal climate policy, while yielding very nearly as much benefit in terms of local pollution as the optimal air pollution policy.

6. Creating co-benefits: public investment management reforms

If development policies and climate change policies, taken jointly, are to have the maximum impact on wellbeing, there needs to be a systematic framework for assessing, selecting and implementing investments by the public sector. Whether the issue is climate-smart agriculture, energy R&D, or investments in electric power, the public sector will play a major role and, in some sectors, it will make major public investments.

From a co-benefits perspective, public investments in electric power generation will be the dominant contributors to emissions of local air pollution and CO_2, with impacts on both public health and future climate risks. At the same time, investments in modern energy are a key driver of economic development. The decisions made in this sector not only contribute to growth, they also lock in environmental footprints and the associated impacts on wellbeing. While there is significant private sector investment in the sector, in many high-income countries and most developing countries the great majority of investments are made in the public sector.

6.1 Assessing the costs and benefits

When choosing between high-carbon and low-carbon energy sources it is vital that the full range of costs and benefits are assessed and valued. This is the subject of social cost–benefit analysis (CBA). Building and strengthening CBA provides the information needed to choose between highly polluting coal-electric generation, for example, and low- or zero-emission alternative energy sources. As many of the model results on low-carbon futures show, the health benefits of clean energy can heavily outweigh the costs of abating carbon emissions.

However, CBA is a tool, not a system for ensuring that public investments are effective engines of growth. It is a key tool, but it needs to be situated in a wider Public Investment Management (PIM) framework. Work on PIM diagnostics at the World Bank (Rajaram et al., 2010) provides useful guidance on the components of an effective public investment management system:

- ***Investment guidance, project development and screening.*** Project choice should be guided by an overarching development plan or strategy. Box 6.1 below highlights PIM issues in the new member states of the European Union, and notes that externally driven priorities can create difficulties for public investment management.

Box 6.1. Public investment management challenges in new members states of the EU

As a result of EU accession, new member states have access to significant amounts of external finance for public sector infrastructure, often amounting to 3–4% of GDP. However, public investment in many Eastern European countries has tended to be short term and politicised. National strategic plans tend to be broad and lack detail, which creates particular difficulties when many of the investment priorities are driven by EU accession – this reduces ownership and, ultimately, effectiveness.

In addition, project appraisal is weak and unlinked to the budget process. Accountability is often lacking, which creates problems for ex-post assessment of project outcomes. Comparing the new member states with countries with advanced PIM systems such as the UK leads to a number of priorities for PIM reforms:

- Sector strategies closely linked to and consistent with projected budgetary commitments
- Significant investments in cost–benefit analysis methodologies, supplemented by business cases analysis and risk management strategies
- Procedures to evaluate projects against value-for-money criteria, both ex-ante and ex-post
- Systematic procedures to involve external experts in the review of sector strategies and project business cases
- Multi-year budget commitments to facilitate efficient management of project planning
- Formal and informal checks and balances to assure that procedures are being complied with in terms of project appraisal and project management
- Public procurement strategies designed to manage risks between the government and the contractor
- Investment in staff training and the employment of specialist experts
- Effective audit and reporting processes that facilitate transparency and encourage feedback to improve the quality of the decision-making and management process

Source: Laursen and Myers (2009)

- ***Formal project appraisal***

- ***Independent review of appraisal***

- ***Project selection and budgeting***. Once project selection is complete, project finance needs to be integrated into a multi-year budget framework that accounts for both investment and operational outlays.

- ***Project implementation***

- ***Project adjustment***. External circumstances change and not every contingency can be foreseen. Managing project implementation therefore has to allow for decisions to be made at distinct project phases concerning the continued viability of the project, leading to changes in design or even halting of disbursement in the most extreme cases.

- ***Facility operation***. Processes are required for the handover of facilities upon project completion. This in turn requires assessment of the state of the facility at handover and the need for adaptation or further investment in order to meet design objectives.

- ***Completion review and evaluation***. Completion reviews should document whether delivery of the project was within the agreed budget and timeline, and whether outputs were delivered as specified. Assessing lessons learned from project implementation should be carried out routinely to ensure that there is ongoing improvement in investment project design and implementation.

6.2 Assessing the performance of country PIM systems

Rajaram et al. (2010) provide a fairly demanding specification of the characteristics of an effective public investment management system, and it would be surprising if country performance against these criteria were not highly variable. Joint work by the World Bank and International Monetary Fund on a PIM assessment tool (Dabla-Norris et al., 2010) shows this to be the case. The assessment tool scores country PIM systems against 17 indicators on a scale of 0–4. Individual indicators are grouped into four categories: strategic guidance and project appraisal; project selection; project management and implementation; and project evaluation and audit. The individual indicator scores are combined into an unweighted average, the Public Investment Management Index (PIMI).

As expected, middle-income countries generally outperform low-income countries on the PIMI, but there is considerable variability within each income class as well. While countries generally dislike being ranked this way by donors, the real value of the index is the provision of a toolkit which can be used to identify weak points in PIM systems in a given country, leading to capacity-building and technical assistance to strengthen country systems.

Conclusions on PIM reforms

As multiple sections of this study have suggested, the welfare benefits of reducing mortality linked to local air pollution exposure are significant. Even countries with long traditions of pollution management such as the US still suffer excess damages in the order of several per cent of GDP (see Table 1.1). Because greener energy can reduce these damages substantially, the choice of electricity generation technologies in particular can have large impacts on wellbeing. The best tool to make the case for greener energy, from a co-benefits perspective, is social cost–benefit analysis.

However, cost–benefit analysis alone will not produce effective outcomes if it is not placed at the heart of a robust public investment management system. Building on longstanding work at the World Bank, there is a growing understanding of what makes an effective PIM and what actions are required to improve PIM systems. Furthermore, PIM reforms aimed at one particular outcome, such as energy sector infrastructure investments, create their own co-benefits in the form of a generally upgraded capacity to invest effectively across multiple sectors.

7. Overall conclusions on co-benefits, and implications for policy

This analysis has emphasised at different junctures that the causal flow in co-benefits analysis can go in different directions – from local actions to global benefits, or vice versa. While this complicates the story, it should not obscure the fact that either causal flow is likely to be good news. Assuming efficient interventions, if the primary goal of a project or policy is reductions in greenhouse gas emissions, then the associated co-benefits in terms of air pollution damages help to reduce the social cost of mitigation; if the primary goal of the project is local benefits, such as a more efficient urban transport system, then the associated climate benefits are a bonus.

7.1 Economy-wide costs and benefits

Because energy use pervades the economy, one question naturally arises: are there economy-wide co-benefits of taking action on climate mitigation? The conclusion from Section 3.2 above is that the evidence for 'macro' co-benefits is not strong.

The impacts of the cost of clean energy on competitiveness appear to be small on average, but trade-exposed energy-intensive sectors can be hard hit. However, the evidence for pollution havens and the offshoring of energy-intensive industry suggest that the effect is modest. The literature on the Porter Hypothesis suggests that environmental policies do induce innovation aimed at reducing costs, and that flexible, market-based policy instruments favour this innovative effort. But the evidence for the 'strong' version of the Porter Hypothesis, that environmental policy can actually result in increased economic efficiency, is decidedly mixed, with the most recent analysis of a large-scale data set for OECD countries (Johnstone et al., 2010) coming out against this version of the hypothesis.

Can climate action produce net employment gains through 'green jobs'? Here the evidence suggests that any gains or losses of jobs are likely to be small. An important consideration is that green jobs are going to replace 'brown' jobs, so that there are both gains and losses from environmental policy; general equilibrium analysis is required. Since jobs are a primary political concern, it is likely that growth-friendly macro policies are a necessary component of any transition from fossil fuels to cleaner energy, and that maintaining flexibility in the labour market and dealing with adjustment costs through retraining (for example) are useful complementary policies.

Energy security is another widespread political concern, but precise definitions of the concept tend to vary. For large energy importers the exposure to oil price volatility is a concern, given the evidence that volatility is damaging to growth. But there is some research indicating that increased flexibility in job markets, combined with credible central bank policies, has tended to decrease the sensitivity of growth to oil shocks. Renewable energy is often from indigenous sources – think of wind or sunlight or increased energy efficiency – and there are model results from GEA (2012) which suggest that both domestic market shares and diversity of energy supplies increase with more stringent climate policies. But governments will need to avoid 'green protectionism' over the course of the energy transition.

7.2 The evidence on environmental co-benefits

There is generally strong evidence for environmental co-benefits from climate mitigation. Shindell et al. (2012) provide a comprehensive analysis of the joint benefits of reducing methane and black carbon emissions, both being short-lived climate pollutants (SLCPs). The 14 different interventions considered produce a reduction in short-term warming of 0.47°C by 2050 compared with a business as usual scenario from the IEA. Reduced mortality in 2030 from black carbon exposure (focused on the PM2.5 subcomponent) amounts to 1.8 million lives saved, valued at US$5.3 trillion, or roughly 5 per cent of gross world product. The largest health benefits are in India, China, Bangladesh, Pakistan and Indonesia. Reducing methane emissions also reduces ground level ozone, and the resultant increases in crop yields are significant in some countries. SLCP emission reductions are an attractive target for climate policy, owing to the relatively short-run benefits of reduced warming. Shindell (2015) expands this analysis to include the latest estimates of the broader costs of climate change, including the effects of aerosols.

Turning to a moderately stringent climate policy scenario, RCP 4.5, West et al. (2013) measure public health co-benefits from reduced air pollution relative to BAU. They find net benefits of CO_2 abatement for most countries (co-benefits minus marginal costs of abatement) measured in hundreds of dollars per ton of CO_2 abated over the course of this century. Of the roughly 19 billion tCO_2 abated in 2050, only 3 billion were in high-income countries, the rest being in middle-income countries. The analysis in West et al. (2013) suggests that careful consideration of economic growth rates and demographic change are important inputs into modelling co-benefits, as is the treatment of very high concentrations of PM2.5 in assessing excess deaths from exposure. New work on concentration-response functions underpinning the Global Burden of Disease 2010 will likely be an important input into future attempts to model air pollution co-benefits of climate mitigation.

Holland et al. (2011) and Rafaj et al. (2013) model a more stringent climate target, 2°C warming by the end of the century, and focus on co-benefits generated in the EU and, more briefly, China and India. One strength of the BAU scenario in this work is that it incorporates projected EU air quality regulations, giving a realistic counterfactual and permitting an explicit measure of air pollution abatement costs avoided in the policy scenario. These avoided costs are large, roughly 13 per cent of total co-benefits, while health benefits constitute 86 per cent of the total. Rafaj et al. (2013) also generate substantial life expectancy gains in India, 30 months on average in 2050, and 20 months in China.

These model results on co-benefits are striking and could be more so if cost minimisation techniques were taken to their logical conclusion. As noted above, rather than minimising the *financial* costs of achieving a given climate target, a more general approach would minimise the *economic* costs of reaching the target – this would effectively endogenise co-benefits in seeking the least cost path.

Energy efficiency increases are effectively a source of zero-emissions energy supply. The full range of co-benefits from clean energy supply therefore accrue to investments in energy efficiency, including health benefits, energy security, macroeconomic benefits from lowered production costs, and reduced air pollution control costs. Looking just at health benefits, the IEA 'efficient world' scenario yields over 200 million life years saved in China in 2035, and over 400 million in India, compared with BAU (the IEA 'current policies' scenario).[20] Fossil fuel subsidies are in effect policies for *energy inefficiency*, and should be a prime target for governments seeking to foster more efficient energy use.

The analysis of the 'abate or go green' decision should be of interest to governments aiming to reduce pollution damages. Countries need to decide whether to apply traditional pollution abatement or to jump to zero-emission technologies such as energy efficiency, wind or solar.

[20] As before, these figures on life years saved need to be reconciled with the Global Burden of Disease 2010 estimates.

While the choice of abating or going green at the margin rests on empirical considerations, the analysis suggests that zero-emission technologies have a structural cost advantage when economic costs are considered, as well as a policy-sensitive advantage which is largest in countries where very little abatement activity has taken place to date. The best solution, however, is to do what theory suggests – internalise both externalities. Bollen et al. (2009) suggest that the returns to internalising both externalities are substantial.

Even in a country such as the US, with reasonably stringent environmental standards, the social cost of local air pollution emissions from conventional coal generation averages over 19 cents/kWh (in 2012 dollars), as seen in Figure 3.3.2. By including environmental damages as part of the cost of generation, it is clear that many of the low-carbon alternatives – geothermal, onshore wind, hydroelectric, advanced cycle natural gas with carbon capture and storage, and advanced nuclear – are cheaper or competitive with conventional coal.

7.3 Policy implications

By focusing only on the cost of greenhouse gas abatement, or the cost of adapting to climate change, countries arguably have taken a blinkered approach to climate policy. Incorporating the analysis of co-benefits and co-costs of climate action can remove the blinkers and lead to better economic decisions with regard to climate change, decisions that are in the country's best interest. It can also widen the scope of potential interventions that governments can make, by considering the different primary and co-benefits available.

For the public sector, two broad reforms to the economic analysis of climate policy actions are needed:

(i) Upgrading cost–benefit analysis capacity, and mandating its application to major public investments

(ii) Reform of the public investment management (PIM) system

With respect to cost–benefit analysis, many countries and international institutions (including the World Bank) in fact do mandate its use for policy and project analysis. The reform issues therefore surround the enforcement of already existing policies, and upgrading capacity and quality. The capacity to value damages to public health from environmental degradation is particularly important in the co-benefits context. The capacity and quality issues are particularly pertinent because these costs and benefits *should already be part of any reasonably comprehensive and competent project appraisal.*

Cost–benefit analysis is the lynchpin of a quality public investment management (PIM) system. Reforms to public investment management systems can permanently increase a country's rate of growth (Harberger, 2005).

From the perspective of climate change, public investments in the electric power sector are likely to be particularly important. Government should establish a social cost of carbon (SCC) for project analysis. Green power investments should be evaluated against the alternative of conventional coal, where local air pollution damages form part of the cost of coal-electric generation. Governments can then measure the carbon switching price for the green project – if this is negative then the green project is socially profitable even at a carbon price of $0; if it is positive but less than the SCC, then the project is profitable for the world but costly for the country. If the decision is to make the green investment, the carbon switching price is a useful measure of how altruistic the country is being.

More comprehensive project analysis will also contribute to actions where climate mitigation is the co-benefit rather than the primary benefit. As noted, this is likely to be the case for transport sector interventions where local benefits are the dominant driver for action. But unless the full range of benefits (e.g. improved public health, less congestion, fewer traffic deaths and lower CO_2 emissions) are valued, the impetus to invest in transport infrastructure may be lacking. Similar considerations may apply to investments in energy efficiency or building soil carbon in agriculture.

Agency issues are an important characteristic of sectors such as transport and buildings, which has consequences for greenhouse gas emissions as well as co-benefits. The person or institution designing the building or transport system is not the person paying the heating/cooling bills or suffering from the effects of vehicle air pollution. Government may need to use instruments such as building standards and vehicle efficiency mandates to deal with these issues. Incentives for landlords also need to be considered since, again, they are not the ones paying the utility bills.

The role of the public sector in energy efficiency or soil carbon may be a matter of overcoming barriers, by increasing access to finance for example, or providing information and training.

With respect to the private sector, governments taking climate actions will need to change private incentives through policy reforms. In a sense, this is just textbook environmental economics – taxes on air pollution need to be combined with taxes on CO_2 emissions. By using efficient economic instruments governments can reduce the cost of abatement and create incentives to innovate; the key message from the Porter Hypothesis literature is that flexible and efficient instruments create the strongest incentive for innovation.

Finally, an important distinction needs to be made between low-income countries and the rest. Low-income countries contribute an extremely small share of global emissions of greenhouse gases. Their priorities must continue to be growth and poverty reduction. Where low-cost mitigation opportunities exist in low-income countries, flows of carbon finance will be needed to support these investments in order to ensure that poor countries and poor people do not end up financing a global public good.

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Appendix I. Estimating and valuing air pollution impacts

Estimating impacts of air pollution exposure

The *impacts* of human exposure to air pollutants are the subject of a large epidemiological literature, in particular a major cohort study carried out by the American Cancer Society (ACS) over the years 1979–83 and 1999–2000. Pope et al. (2002) provide careful statistical analysis of this data, controlling for potential confounding factors, and estimate the relative risks of mortality from cancer, cardiovascular disease, and all cause mortality, linked to exposure to particulate matter less than 2.5 microns in size (PM2.5). The cohort study considers risks to people aged over 30 years.

The US Environmental Protection Agency (EPA) relies upon the three studies shown in Table A1.1 as a source for *concentration response functions* (CRFs) to estimate excess mortality linked to PM2.5 exposure. These CRFs are used in the regulatory impact analysis underpinning air quality standards in the US. The output of the CRFs are relative risks (RR), which measure the risk of mortality in the exposed population relative to the unexposed population.

Table A1.1. Summary of effects of long-term exposure to PM2.5 on all-cause mortality, based on US cohort studies

Study	Data source	Relative risk of all-cause mortality for 10ug/m^3 change in PM2.5 exposure (95th percentile confidence intervals)
Pope et al. (2002)	American Cancer Society (age >30)	1.06 (1.02–1.11)
Krewski et al. (2009)	American Cancer Society (age >30)	1.06 (1.04–1.08)
Lepeule et al. (2012)	Six Cities (age > 25)	1.14 (1.07–1.22)

Source: US EPA (2012)

For relatively unpolluted high-income countries (with maximum PM2.5 concentrations of 30ug/m^3) the choice has been to construct a roughly linear CRF for relative risks (RR) as follows:

$$RR = e^{\beta(C_2 - C_1)}$$

Here $(C_2 - C_1)$ is the change in pollution concentration, while β is calibrated to the relative risks shown in Table A1. The computed relative risk is multiplied by the rate of all-cause mortality for the population of interest, and the revised rate of mortality is then multiplied by the exposed population in order to arrive at estimated excess deaths associated with the change in pollution concentration.

The problem with this linear CRF is that there have been no large cohort studies of developing countries where populations can be exposed to PM2.5 concentrations of 80ug/m^3 or more: assuming linearity, combined with generally high rates of all-cause mortality in developing countries, could lead to unrealistically large estimates of excess deaths from air pollution exposure. The approach taken in early versions of the Global Burden of Disease (WHO, 2010) was to simply cap relative risks at the level of 30ug/m^3, which potentially underestimates excess deaths in highly polluted developing countries.

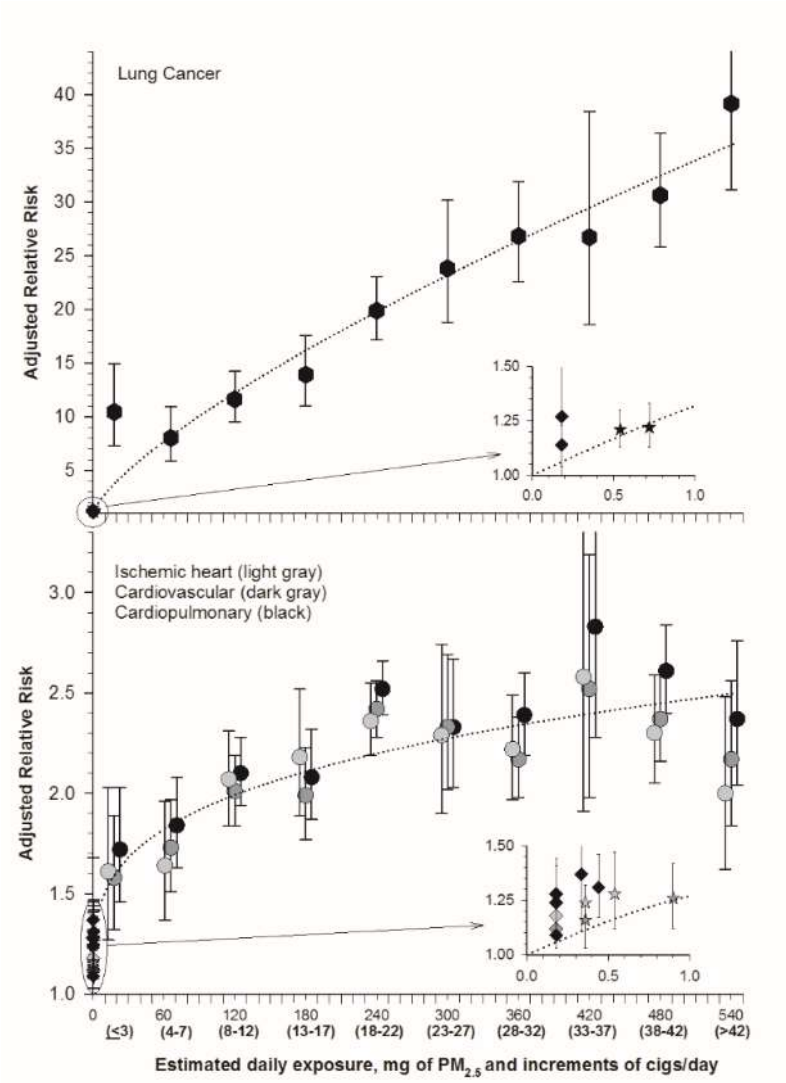
Cohen et al. (2004) suggest a log-linear CRF to deal with the problem of high concentrations,

$$RR = \left(\frac{C_2}{C_1}\right)^{\gamma}$$

Here parameter γ is again calibrated to the relative risks in Table A1. This formulation assumes that there are damages at pollution concentrations above 30ug/m³, but the rates of damage tail off rather than rising linearly.

Pope et al. (2011) have exploited data on deaths from exposure to second-hand cigarette smoke in order to estimate the relative risks of exposure to high concentrations of PM2.5. Figures A1.2 and A1.3 show the fitted CRFs for lung cancer and heart and lung disease. The relationship for lung cancer is roughly linear, with a relative risk of mortality of 10 at a PM2.5 concentration of 80ug/m³. For heart and lung diseases the CRF quickly tails off, but there is still a large relative risk of 1.9 for a PM2.5 concentration of 80ug/m³.

Figures A1.2 and A1.3. Relative risks of lung cancer mortality and heart and lung disease mortality as a function of daily exposure to PM2.5



Source: Pope et al. (2011)

When applied in developing countries the CRFs from Pope et al. (2011) are likely to imply high levels of excess mortality from air pollution exposure. The Global Burden of Disease 2010 report (Lim et al., 2012) incorporates the findings of Pope et al. (2011) and constructs a suite of CRFs based on cohort data on PM2.5 exposure from active smoking, second-hand tobacco smoke, and indoor air pollution from biomass stoves. The technical paper is published as Burnett et al. (2014).

One general consideration in applying CRFs to estimate impacts of air pollution exposure is the baseline rate of mortality, whether cause-specific or all-cause. Elevated baseline mortalities will tend to increase estimated deaths linked to air pollution exposure. As populations age over the course of this century, this will be an increasingly important factor in estimating air pollution mortality.

Estimating the value of damages

For each endpoint in exposure to air pollution there is an associated physical effect (loss of human life, or acidification of a lake, for example) and economic value of this effect. While damages to many assets can be valued using a loss of productivity approach, the question is more complex when human lives are at risk. Although there is some tradition of valuing premature mortality as the present value of lost wages, this runs counter to the usual assumption in economics that it is the change in wellbeing which is the correct way to value a given outcome.

The economic approach to the question of valuing premature mortality is based on 'valuing a statistical life' (VSL). Asking someone how much they would be willing to pay to avoid dying tomorrow is unlikely to produce a sound answer to the question of the value of a life lost. Stronger methods are required to value mortality. But the VSL terminology is not entirely satisfactory as a name. In fact the VSL is based on measuring the willingness to pay to reduce the risk of death – if people are willing to pay $6,000 to reduce the risk of death by 1/1,000, then the implicit VSL is $6 million.

Viscusi and Aldy (2003) reviewed a large literature on measuring VSL using revealed preference techniques – in particular hedonic wage studies where remuneration can be assumed to reflect the risk of death on the job. They carried out a meta-analysis of over 40 studies (mostly in the US) and found that, depending on the econometric specification, (i) the VSL for all studies varied from US$5.0 to $6.2 million (in year 2000 dollars), and that (ii) for the US studies the value varied from US$5.5 to $7.6 million.

It is reasonable to assume that willingness to pay measures will vary with income, and the meta-analysis of Viscusi and Aldy finds an income elasticity of the VSL that varies from 0.46 to 0.60.

Because there are very few estimates of VSL in developing countries, the standard approach to estimating VSL in the literature is to 'transfer' the US figure for VSL by accounting for differences in income per capita and the income elasticity of VSL. If the assumed income elasticity is 0.5, Y_C is the per capita income in country 'C', and Y_{US} is the US per capita income, then the value of a statistical life in country C is given by:

$$VSL_C = VSL_{US}\left(\frac{Y_C}{Y_{US}}\right)^{0.5}$$

This approach is reasonable when the differences in income per capita between countries are not too large (as is the case for most of the studies appearing in the meta-analysis of Viscusi and Aldy), but can lead to extreme results for countries with very large income differences. If the US income per capita is US$40,000 and the income per capita of country C is US$400, then the VSL calculated for country C is fully 10 per cent of the US figure for VSL, while the ratio of incomes is a factor of 100.

From the perspective of climate modelling over the century, this concern about the assumed income elasticity of the VSL becomes less of an issue, however, as incomes converge across countries.

It is possible to value serious illness associated with exposure to air pollution in a similar manner by using the loss of quality-adjusted life years (QALYs) associated with the illness. The measurement of QALYs suggests that living a year with severe chronic bronchitis, for example, equates to 0.4 of a life year lost (Miller et al., 2006), and incidence of severe chronic bronchitis could therefore be valued at 0.4 of the VSL (less severe incidence would of course imply a lower proportion of the VSL).

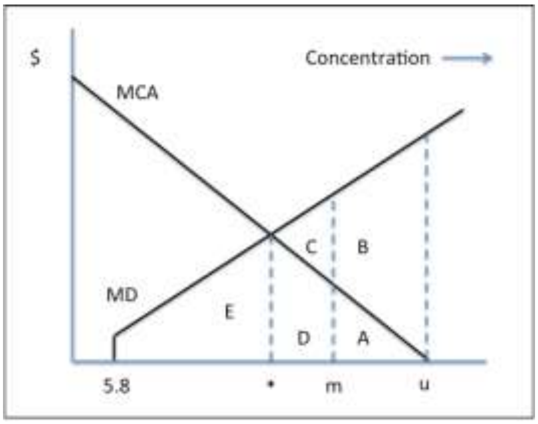
Finally, it is important to note new analysis of the VSL literature published in OECD (2012). This work focuses on stated preference studies, as opposed to the revealed preference (hedonic wage) studies analysed by Viscusi and Aldy (2003). The new meta-analysis suggests an income elasticity of the VSL of 0.8, substantially larger than the 0.4 figure used by the US EPA. The higher income elasticity will have significant impacts on transferring VSL values between high-income countries, and especially so between high-income and low-income countries. In fact, Hammitt and Robinson (2011) suggest that the income elasticity for transferring VSL between high- and low-income countries should be at least 1.0.

Hamilton (2014) calculates the values of PM2.5 mortality appearing in Tables 1.1 and 1.2 using the OECD income elasticity of 0.8 when transferring values from the EU to countries richer than the mean EU country. For countries poorer than the EU mean, an income elasticity of 1.0 is used.

Appendix II. Analysing the decision to abate local pollution, or to shift to clean energy

Figure A2.1 presents the classic marginal cost/marginal benefit diagram for analysing pollution abatement policies, in this case abating fine particulate matter (PM2.5). Assuming fixed dispersion and population distribution, concentrations can be assumed to be proportional to emissions. 'MCA' is the 'marginal cost of abatement' curve. 'MD' is the 'marginal damage' curve – this is assumed to be zero for concentrations of PM2.5 less than 5.8ug/m^3.

Figure A2.1 Marginal cost of abatement and marginal damages as a function of PM2.5 concentrations



Notes: MCA = marginal cost of abatement. MD = marginal damage. Source: Authors

Point **u** on the horizontal axis is the uncontrolled level of pollution, leading to maximum pollution concentration and damages. Point **m** is a moderate level of abatement, while point ***** is the optimum level of abatement. Letters **A** to **E** represent areas measuring costs and benefits.

To simplify the analysis, assume that there is a uniform coal-electric technology consisting of identical plants which in total produce *X* kWh of electricity per year.

Marginal analysis

If total PM2.5 emission is equal to **m** tons as a result of a moderate level of pollution abatement expenditure and one of the existing coal plants is about to be retired, then government has the choice of building a new coal plant, with moderate pollution controls in order to maintain the existing air quality, or switching to wind (standing in for other zero-emission technologies) with a baseline generation cost of G^W. The baseline generation cost per kWh for coal *with no pollution abatement* is G^C. Referring to Figure A2.1, the financial and economic costs of generating 1kWh with the alternative technologies are therefore as follows:

Coal: unit financial cost per kWh	$G^C + \frac{A}{X}$
Coal: unit economic cost per kWh	$G^C + \frac{A}{X} - \frac{A+B}{X} = G^C - \frac{B}{X}$
Wind: unit financial cost per kWh	G^W
Wind: unit economic cost per kWh	$G^W - \frac{A+B+C+D+E}{X}$

Economic costs in this instance subtract the external benefits of avoided pollution damages. For coal generation, therefore, $\frac{A}{X}$ is the financial cost of abating pollution up to level **m** (in per-kWh terms), while $\frac{A+B}{X}$ is the gross external benefit of abating pollution to this level. By this logic, it follows that when making the investment decision to replace the retired coal plant:

Wind is preferred to coal if $G^W < G^C + \frac{A+C+D+E}{X}$.

There are two points to note about this result:

(i) Quantity $\frac{A+D+E}{X}$ is constant, unaffected by the stringency of pollution policy – it represents economic cost savings irrespective of the stringency of pollution policy. This can be seen by inspection in Figure A2.1.

(ii) Quantity $\frac{C}{X}$ is sensitive to pollution policy, and shrinks to 0 as policy approaches the optimal level.

A hypothetical example gives a feel for the numbers. Suppose that:

G^C = 7c/kWh, $\frac{A}{X}$ = 1c/kWh, $\frac{D}{X}$ = 1c/kWh, $\frac{E}{X}$ = 1.5c/kWh, $\frac{C}{X}$ = 1c/kWh.

Seven cents per kWh is a realistic estimate of conventional coal generation costs in the US, excluding pollution mitigation (see the main text). If wind generation costs were less than 11.5c/kWh in this example, wind would be the preferred investment to replace the retired coal plant. If the country aimed to reduce the pollution emissions from the replacement plant, then $\frac{C}{X}$ might drop by half a cent, but wind would still be preferred at any price up to 11c/kWh. The constant economic cost advantage enjoyed by wind, irrespective of the stringency of pollution policy for coal, is $\frac{A+D+E}{X}$ = 3.5c/kWh.

Because substituting zero-emission energy for fossil fuels produces savings in terms of both improved health and foregone pollution abatement, the 'abate or go green' decision may favour choosing zero-emission energy as the optimal pollution abatement strategy. In the end this is an empirical question for policymakers to consider. If governments wish to control both greenhouse gas emissions and local air pollution, then the optimal strategy will be to internalise both externalities, using separate instruments, as discussed in the main text.