1. Distributional issues in climate policy: air quality co-benefits and carbon rent*

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1. INTRODUCTION

Climate change is often framed as posing a tradeoff between the welfare of the present and future generations. Policies that aim to mitigate climate change – most importantly, by reducing the use of fossil fuels – are assumed to require sacrifices on the part of those alive today for the sake of those who will follow. Invoking normative criteria of equity, efficiency or both, policy proponents maintain that the future gain from curtailing emissions will outweigh the present pain, while opponents make the opposite argument. Both sides agree, however, that the policies will require upfront costs. The public is left to weigh conflicting views on whether the benefits to future generations really justify the costs of taking action today.

The choice of an appropriate discount rate is a critical issue once this framework is accepted. For example, the UK government's Stern Review used a discount rate of about 1.4 per cent, whereas William Nordhaus of Yale University has used rates of 4 per cent or higher in his integrated assessment model.[1] The choice has major effects on what policies are deemed efficient: a lower discount rate justifies more aggressive policies to reduce emissions.

Inter-generational equity figures centrally in this debate. "Many would argue," the Stern Review noted, "that future generations have the right to enjoy a world whose climate has not been transformed in a way that makes human life much more difficult" (2007, p. 47). Citing a forecast that global per capita income will rise from $10 000 today to $130 000 (in today's dollars) in the next two centuries, Nordhaus (2008) countered: "While there are plausible reasons to act quickly on climate change, the need to redistribute income to a wealthy future does not seem to be one of them."[2]

Intra-generational equity has received less attention in climate policy debates. This reflects the prevalent assumption that climate policy necessarily will impose costs on the present generation. How these costs will be distributed has been seen as a secondary issue, overshadowed by the contentious tradeoff of present costs for future benefits.

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[1] Stern (2007); Nordhaus (2007). For a review of this debate, see Goulder and Williams (2012).

[2] This projected income may fail to account adequately for the potential adverse economic impacts of climate change itself (see Moore and Diaz, 2015; Stern, 2016).

This chapter challenges this conventional framing of the problem by examining the potential to design mitigation policies that yield substantial net benefits here and now.[3] These include benefits to the public at large, above and beyond the income and employment gains that would be generated by investments in energy efficiency and renewables.[4] They also are above and beyond the near-term benefits from mitigation itself, such as reduced risk from coastal flooding or extreme heat waves.

Two sorts of present-day benefits are considered here. The first are the improvements in air quality and public health that would come with reduced use of fossil fuels. The second is the net income gain that the majority of households would receive if the rent derived from a price on carbon emissions were to be recycled to the public as equal per capita dividends. Both sorts of benefits entail important issues of intra-generational equity. And both suggest that the assumed tradeoff between present and future generations can be overcome by climate policies that secure broad public support on the basis of present-day benefits.

2. AIR QUALITY CO-BENEFITS

In addition to reducing emissions of carbon dioxide, policies that curtail fossil fuel combustion reduce emissions of numerous air pollutants that damage human health, including particulate matter, sulfur dioxide, nitrogen oxides, and carbon monoxide. In fact, exercises that have assigned monetary values to the damages from carbon dioxide and these "co-pollutants" often put higher values on the latter. Co-pollutant intensity (the ratio of co-pollutant damages to carbon dioxide emissions) varies, however, across emission sources. For this reason, the spatial and sectoral composition of emissions reductions is important for efficiency and equity.

The World Health Organization (WHO) characterizes air pollution as "the world's largest single environmental health risk," calculating that outdoor and indoor air pollution together is responsible for one in eight premature deaths worldwide. Outdoor air pollution causes 3.7 million deaths annually. The WHO notes that more sustainable strategies in transport, energy and other sectors will be "more economical in the long term due to health-care cost savings as well as climate gains" (WHO, 2014).

Valuing the Health Costs of Air Pollution

Several studies have valued the costs of outdoor air pollution in monetary terms. A multi-country analysis by the Organisation for Economic Co-operation and Development (OECD) concluded that outdoor air pollution (specifically, particulate matter and ozone) was responsible for 2.45 million premature deaths annually in the OECD countries plus China and India in 2010 (see Table 1.1). China and India accounted for roughly 80 per cent of the total; among OECD countries the US ranked first with roughly 110 000 deaths.

[3] Here my focus is solely on mitigation. Climate change adaptation also poses deep distributional questions; for discussion, see Boyce (2014b).

[4] On the net employment gains of "green growth," see Pollin (2015).

Table 1.1 Costs of outdoor pollution in China, India and OECD countries, 2010

Country	Premature deaths (per year)	Value of a statistical life[1] (USD million)	Economic cost[2] (USD billion/year)
China	1 278 890	0.975	1371.4
India	692 425	0.602	458.4
US	110 292	4.498	545.8
Japan	65 776	3.068	222.0
Germany	42 578	3.480	163.0
Italy	34 143	2.995	112.5
Turkey	28 924	2.024	64.4
Poland	25 091	2.098	57.9
UK	24 064	3.554	94.1
Korea	23 161	3.027	77.1
Mexico	21 594	1.811	43.0
France	17 389	3.155	60.4
Other OECD	85 092	3.078	288.5
Total	2 449 419	1.321	3558.4

Notes:
[1] OECD calculation of the value of a statistical life (VSL) as a function of income per capita. For discussion, see text.
[2] Economic cost = Cost of mortality + morbidity.

Source: OECD, 2014, Tables 2.4, 2.7, 2.10, and 2.13–2.18.

To monetize these impacts, the OECD multiplied the number of deaths by the value of a statistical life (VSL, also sometimes termed the value of a prevented fatality), computed as a concave function of national income per capita, and then added 10 per cent for the costs of non-fatal illnesses. The total cost in these countries amounted to $3.5 trillion/year, with the OECD member countries accounting for about half of this and China and India for the rest (OECD, 2014).

The lower shares of China and India in monetary damages than total deaths are attributable to the OECD study's use of country-specific VSLs. Statistical lives in India and China were valued at $602 000 and $975 000, respectively, compared, for example, to $4.5 million in the US.[5] The OECD (2014, pp. 53–5) offers the following rationale for this procedure:

> A VSL value is meant to be an aggregation of individual valuations: an aggregation of individuals' WTP [willingness to pay], as communicated through WTP surveys, to secure a marginal reduction in the risk of premature death. In the world as we know it, individuals are differently endowed with the means with which to make such a trade-off; some work for their living for a dollar a day, some inherit a fortune yielding unearned income of a billion dollars a year. Human societies without exception have sought to socialise these risks to a greater or lesser extent in the

[5] Adopting a similar valuation procedure, a report of the World Bank and the Institute for Health Metrics and Evaluation (2016) used 2013 VSLs for India, China, and the US of $400 000, $978 000, and $5 million, respectively (purchasing power parity-adjusted 2011 dollars, calculated from data in Appendix B of the report).

> form of public goods . . . And it so happens that the level at which this socialisation of risks is executed today is the level of the nation-state. It is for this reason, and this reason alone, that it is appropriate to aggregate at the level of country-specific VSLs.

As discussed below, an alternative procedure would be to apply a uniform VSL to all countries, based on the ethical premise that all human lives are equally valuable regardless of individual wealth or per capita income in the country where the person happens to reside, or to use a poverty-weighted VSL that puts greater value on the lives of the most vulnerable. As Sunstein (2014, p. 89) remarks:

> If poor people are subject to a risk of 1/10 000, they do not have less of a claim to public attention than wealthy people who are subject to exactly the same risk. In fact they may have a greater claim, if only because they lack the resources to reduce that risk on their own.

Which of these valuation procedures is taken to be more appropriate depends on who foots the risk-reduction bill. When the poor must pay the cost of risk reduction themselves, a reasonable case can be made that they should not be compelled to spend as much as wealthier people would spend for protection against statistical risks. "Requiring poor people to buy Volvos," Sunstein (2014, p. 90) remarks, "is not the most sensible means of assisting them." On this basis, Sunstein contends that "for China or India, it would be disastrous to use a VSL equivalent to that of the United States or Canada." Whether the logic for individuals also applies to governments – with per capita income replacing individual income – is not obvious, however, since the distribution of risk-reduction costs and benefits is likely to vary across the national population. And as Sunstein himself notes, his argument "should not be taken to support the ludicrous proposition that donor institutions, both public and private, should value a risk reduction in a wealthy nation more than equivalent risk reduction in a poor nation."

Calculating the Co-Pollutant Cost of Carbon

The OECD's mortality data refer to outdoor air pollution from all sources, including not only fossil fuel use but also other sources such as wildfires, the burning of biomass, and construction dust. Reliable data on source-wise apportionment of air pollution are sparse, but for many pollutants in many countries fossil fuels are the most important source. Road transportation alone accounts for approximately half the outdoor air pollution in the EU24, according to the OECD (2014, p. 63), and for roughly one-third in the US where electric power generation (also from fossil fuel combustion) accounts for a higher share of the total than in Europe.

A study by MIT researchers estimates that 211 875 premature deaths (90 per cent confidence interval: 91 000–383 300) in the US in 2005 were attributable to particulate matter and ozone as a result of combustion emissions (Caiazzo et al., 2013).[6] Transportation (road, marine, rail and aviation) and electric power generation accounted for 60 per cent

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[6] The difference between the total US deaths from outdoor air pollution as estimated by Caiazzo et al. (2013) and the OECD (2014) is likely attributable, in part, to emissions reductions between 2005 and 2010. See Fann et al. (2013).

Table 1.2 Premature deaths from outdoor air pollution in the US associated with combustion emissions from different sectors, 2005

Sector	Premature deaths	
	Number	%
Road transportation	58 050	27.4
Electric power generation	53 900	25.4
Industry	42 550	20.1
Commercial/residential	42 150	19.9
Marine transportation	8830	4.2
Rail transportation	5040	2.4
Aviation	1355	0.6
Total	211 875	100

Source: Calculated from Caiazzo et al. (2013), Table 4.

of these, with the remainder due to other industrial, commercial and residential activities (Table 1.2).

An international analysis of premature mortality from outdoor air pollution reaches similar conclusions for the US, attributing 52 per cent to electric power generation and land traffic (Lelieveld et al., 2015). In other OECD countries included in this analysis, the authors estimate that the joint share of these two sectors ranges from 27.3 per cent (in Korea) to 35.5 per cent (in the UK). The joint shares in China and India are 20.8 per cent and 18.5 per cent, respectively, with residential and commercial energy use accounting for larger shares in those two countries. If we attribute all air pollution from transportation and the power sector to fossil fuel combustion, plus one-quarter of the air pollution from other sectors, these figures would imply that fossil fuel use accounts for roughly 65 per cent of premature mortality from outdoor air pollution in the US, 50 per cent in other OECD countries, and 40 per cent in China and India.

Applying these percentages to the OECD data in Table 1.1, we can calculate health impacts of co-pollutants per ton carbon dioxide emissions. I term this ratio the Co-Pollutant Cost of Carbon (CPCC). Three measures of the CPCC are reported in Table 1.3. The first, the number of premature deaths/ton CO_2, ranges from fewer than 13 in the US to more than 160 in India. The second, US dollars/ton using the OECD's valuation procedure (in which VSL varies with per capita income), ranges from \$50/ton in Mexico to \$134/ton in Italy. The final measure applies a uniform VSL to all countries, while holding unchanged the sum total of monetary damages according to the OECD study. By this measure, which is directly proportional to deaths/ton, India's CPCC exceeds \$200/ton.

The CPCC for the US in 2010 based on the OECD valuation procedure was \$64/ton. If, rather than the \$4.5 million VSL used for the US in the OECD study, we were to apply the higher VSL used by the US Environmental Protection Agency (USEPA), the CPCC would increase correspondingly.[7]

[7] The official VSL used by the USEPA in 2013 was \$9.7 million (USEPA, 2016, p. 2). For comparisons of the VSL used by USEPA and other US government agencies, see Robinson (2007).

Table 1.3 Co-pollutant cost of carbon, 2010

Country	Premature deaths from fossil fuel emissions	CO_2 emissions (million mt)	Co-pollutant cost of carbon (per mt CO_2)		
				US dollars	
			Deaths	OECD VSL	Equal VSL
China	511 556	7388.5	69.2	74.2	100.6
India	276 970	1714.9	161.5	106.9	234.6
US	71 690	5580.0	12.8	63.6	18.7
Japan	32 888	1177.3	27.9	94.3	40.6
Germany	21 289	797.0	26.7	102.3	38.8
Italy	17 072	419.8	40.7	134.0	59.1
Turkey	14 462	268.5	53.9	119.9	78.2
Poland	12 546	304.6	41.2	95.0	59.0
UK	12 032	529.5	22.7	88.8	33.0
Korea	11 580	584.0	19.8	66.0	28.4
Mexico	10 797	434.0	24.9	49.6	35.6
France	8 694	385.6	22.5	78.3	32.3
Other OECD	42 546	2588.3	16.4	55.7	23.6
Total	1 044 122	22 172.1	47.1	68.4	68.4

Sources: Premature deaths from fossil fuel emissions and co-pollutant cost of carbon: author's calculations (see text). CO_2 emissions (from consumption of coal, petroleum + natural gas): US Energy Information Agency, https://www.eia.gov/cfapps/ipdbproject/IEDIndex3.cfm?tid=90&pid=44&aid=8, accessed 11 February 2016.

It is instructive to compare the CPCC to the Social Cost of Carbon (SCC) that is used by the US government in regulatory analyses as a measure of climate damage. The SCC does not include the damages from co-pollutants that are released along with carbon dioxide. The average SCC in 2015 ranged from \$11 to \$56/ton CO_2 depending on the choice of the discount rate, with \$105/ton used to test the sensitivity of cost–benefit analysis results to "the potential for higher-than-average damages" (USEPA, 2015).[8] The CPCC in the US thus is comparable to, and possibly even larger than, the government's average SCC.

Other studies have come to similar conclusions. An analysis of prospective air quality co-benefits from "deep decarbonization" policies in the US found that they would prevent approximately 36 000 premature deaths/year from 2016 to 2030 and concluded that the co-benefits would exceed the climate benefits valued on the basis of the official SCC (Shindell et al., 2016). For the European Union, a study by the Netherlands Environmental Assessment Agency concluded that the air quality co-benefits from a stringent climate policy would be large enough to offset the policy's costs "even when the long-term benefits of avoided climate impacts are not taken into account" (Berk et

[8] As of 2014 the SCC had been used in more than 40 regulatory impact analyses by US government agencies (GAO, 2014). For details on how it was derived, see US Interagency Working Group on the Social Cost of Carbon (2013). For critiques, see Ackerman and Stanton (2012) and Foley et al. (2013).

al., 2006). Summarizing 37 studies from around the world, Nemet et al. (2010) found a mean value for air quality co-benefits of \$49/ton of CO_2. An IMF study (Parry et al., 2014) concluded that in the top twenty CO_2-emitting countries the average nationally efficient carbon price based on domestic co-benefits alone would be \$57.5/ton, without counting global climate benefits.

For climate policy the salience of air quality co-benefits may be even greater than these monetary valuations suggest. Air quality benefits are predominantly near-term and national, whereas climate benefits are predominantly long-term and global.[9] Greater emphasis on the magnitude of air quality co-benefits may therefore help to overcome the political impediments to climate policy that arise from myopia and concerns about international free riding.

Efficiency Implications

From an efficiency standpoint two conclusions follow:

First, inclusion of the air quality co-benefits justifies more stringent regulatory measures than if policy were based solely on damages from CO_2 emissions. Let us define the Full Social Cost of Carbon (FSCC) – total damages per ton of fossil CO_2 emissions – as the sum of the climate change cost of carbon (CCCC) and the co-pollutant cost of carbon (CPCC):

$$\text{FSCC} = \text{CCCC} + \text{CPCC}$$

Compared to the conventional SCC, which is based on the CCCC alone, the FSCC strengthens the efficiency case for curtailing use of fossil fuels, providing a yardstick for higher carbon prices and more ambitious emission reduction targets.[10]

Second, insofar as air quality co-benefits per ton of CO_2 vary across pollution sources and locations, efficiency can be enhanced by designing policies so as to achieve deeper emissions reductions where co-benefits are greater. The rationale for doing so can be illustrated by an example. Consider two facilities in California: a power plant located outside Bakersfield and a petroleum refinery located in metropolitan Los Angeles, each of which emits the same amount of CO_2, roughly 3 million tons per year (t/yr). The power plant also emits about 50 t/yr of particulate matter (PM) and has fewer than 600 residents living in a 6-mile radius, while the refinery emits about 350 t/yr of PM and has about 800 000 residents living within a 6-mile radius (Pastor et al., 2013). Clearly, the health co-benefits associated with a ton of carbon emission reductions will be greater at the refinery than at the power plant. Though this example is particularly dramatic, substantial variations in co-pollutant intensity are found across industrial sectors in the US (see, for example, Table 1.4).

9 As Shindell (2015) observes, "near-term health impacts seem to typically be considered more important to citizens than longer-term impacts of any sort, consistent with the vastly greater sums spent on medical care and research than on long-term environmental protection."

10 Shindell (2015) similarly proposes the term "Social Cost of Atmospheric Release" (SCAR) to refer to the combined climate and air quality damages from emissions of multiple pollutants. FSCC thus can be defined as SCAR per ton of CO_2 emissions from fossil fuel combustion.

Table 1.4 $PM_{2.5}$ intensity by industrial sector, United States

Industrial sector	Population-weighted $PM_{2.5}$ per ton CO_2	Minority share (%)
Primary metal manufacturers	19.7	47.5
Non-metallic mineral product manufacturers	8.6	39.8
Petroleum refineries	8.4	59.5
Chemical manufacturers	5.2	43.9
Power plants	3.0	38.8

Source: Boyce and Pastor (2013).

Equity Implications

From an equity standpoint as well, air quality co-benefits have significant implications for climate policy. In the US, for example, racial and ethnic minorities and low-income communities tend to bear disproportionate air pollution burdens (see, for example, Ringquist, 2005; Mohai, 2008; Morello-Frosch et al., 2011; Cushing et al., 2016).

Executive Order 12898, "Federal Actions to Address Environmental Justice in Minority Populations and Low-Income Populations," issued by President Bill Clinton in 1994, directs every US government agency to take steps to identify and rectify "disproportionately high and adverse human health or environmental effects of its programs, policies, and activities on minority populations and low-income populations." This directive made equity an explicit objective in federal environmental policy. Many US states now have similar environmental justice policies (Bonorris, 2010).

The extent of air pollution exposure disparities varies across facilities, industrial sectors and locations (Ash and Boyce, 2011; Ash et al., 2009; Zwickl et al., 2014). Nationwide, racial and ethnic minorities bear 59.5 per cent of the impact of particulate emissions from petroleum refineries, for example, compared to 38.8 per cent in the case of power plants, the latter figure being closer to their 34.2 per cent share in the national population in 2005–09 (see Table 1.4). An equity-oriented climate policy would aim to achieve greater emissions reductions not only from those sources where air quality co-benefits are greater, but also from sources where co-pollutant damages are more unequally distributed by race, ethnicity and income.

Incorporating Air Quality Co-Benefits into Climate Policy

Two broad categories of direct policy instruments can be used to reduce fossil fuel combustion: quantitative regulations, such as fuel economy standards for automobiles and renewable portfolio standards for power plants; and price-based policies, such as a carbon tax or marketed carbon permits. These are not mutually exclusive, since the public policy mix can and usually does include both. Anti-smoking policies, for example, combine restrictions on who can buy tobacco and where smoking is permitted with excise taxes to discourage smoking. Similarly, policies to cut sulfur dioxide emissions from US power plants have combined mandated technologies and emission standards with a cap-and-trade permit system.

One reason to include price-based tools in the policy mix is that they create incentives not only to adopt existing pollution control technologies but also to invest in research and development of new ones. Price-based policies often have encountered opposition, however, from environmental justice advocates on the grounds that they may allow co-pollutant "hot spots" to persist, and possibly even worsen, in overburdened communities. Environmental justice organizations in California, for example, filed a lawsuit in an attempt to block implementation of the state's CO_2 cap-and-trade program for this reason (Farber, 2012). Hot-spot concerns can be addressed both by quantitative regulations and by allowing prices to vary across sources, for example by a zone system with tighter caps (or, equivalently, higher prices) in priority locations (Boyce and Pastor, 2013).

Some economists have argued that co-pollutants should not be factored into climate policy design because they are best regulated separately (Schatzki and Stavins, 2009). Of course, pollution control technologies, such as scrubbers in smokestacks, can reduce co-pollutant emissions without reducing the use of fossil fuels (and hence without reducing CO_2 emissions). The adoption of such technologies by advanced industrialized countries is a major reason why their premature deaths from fossil fuel emissions per ton of carbon are much lower than in China and India, as seen in Table 1.3. But the fact that the CPCC remains high even in advanced industrialized countries with relatively stringent environmental regulations – ranging from \$64 to \$134 per ton CO_2 in Table 1.3 – means that it remains an important component of the full social cost of carbon, and hence relevant for assessing the full social benefits of carbon reduction.

The health costs of air pollution also remain large relative to the costs of pollution control, implying that actually existing environmental policies are far from what economists would characterize as efficient. A cost–benefit analysis for the European Union's Thematic Strategy on Air Pollution (TSAP) found current policies to be suboptimal in all EU member countries (Holland, 2012, 2014). The "extraordinary high net benefits and benefit-cost ratios" reported in the TSAP study, the OECD (2014, p. 76) remarks, "suggest that something has gone wrong with the decision-making process."

Unless and until one can reasonably assume that co-pollutant impacts are efficiently and equitably addressed by separate regulations, climate policy should take them into account. This approach to climate policy is consistent with the growing embrace of multi-pollutant strategies for air-quality management.[11] The authoritative US government document on regulatory impact analysis, Office of Management and Budget (OMB) Circular A-4, for example, explicitly directs federal agencies to consider co-benefits (also known as "ancillary benefits"):

> Your analysis should look beyond the direct benefits and direct costs of your rulemaking and consider any important ancillary benefits and countervailing risks. An ancillary benefit is a favorable impact of the rule that is typically unrelated or secondary to the statutory purpose of the rulemaking (e.g., reduced refinery emissions due to more stringent fuel economy standards for light trucks) . . . (OMB, 2003, p. 26)

[11] See National Academy of Sciences (2004) and McCarthy et al. (2010).

In a similar vein, a study by the European Environment Agency (2006) concluded that climate policies could significantly reduce both health damages from air pollution and the costs of controlling air pollutant emissions.

Administratively, it is not terribly difficult to incorporate air quality co-benefits into climate policy design, particularly in settings where co-pollutant damages are concentrated in a relatively small number of facilities, sectors or locations, as in the case of US industrial point source emissions (Boyce and Pastor, 2013). Policy options include the following:

1. *Monitor impacts on co-pollutants*: A minimalist option is simply to monitor co-pollutant emissions with a view to instituting remedial measures if the climate policy has unacceptable impacts, such as exacerbation of environmental disparities across racial, ethnic or income groups. This was the approach taken by the California Air Resources Board (2011) in its adaptive management plan for the state's cap-and-trade policy.
2. *Zonal tax or permit systems*: Carbon permit or tax systems can ensure emissions reductions in high-priority zones where potential public health benefits are greatest. Zone-specific caps were used, for example, in California's Regional Clean Air Incentives Market, which was initiated in 1994 to reduce emissions of NO_x and SO_2 in the Los Angeles basin (Gangadharan, 2004).
3. *Sectoral tax or permit systems*: Similarly, sector-specific permit or tax systems can be designed to ensure emissions reductions in those economic and industrial sectors with the highest co-pollutant intensities or the greatest disproportionate impacts on minority and low-income populations.
4. *Trading ratios*: In a tradable permit system where damages per unit of emissions vary across sources, the exchange rate at which permits are traded can serve as another policy instrument for achieving greater reductions from specific sources. For example, if the full social cost of carbon (CO_2 plus co-pollutant damages per ton CO_2) are twice as high in location A as in location B, the exchange rate ("trading ratio") would be 1:2 (Muller and Mendelsohn, 2009).
5. *Community benefit funds*: Finally, some fraction of the revenue obtained from carbon taxes or permit auctions (the "carbon rent" discussed below) can be channeled into community benefit funds to mitigate pollution impacts and protect public health in overburdened and vulnerable communities. This strategy has been enacted for revenues from permit auctions under California's climate policy.[12]

3. CARBON RENT ALLOCATION

Putting a price on carbon emissions by means of a cap or a tax is widely viewed as a central element of climate policy for good reason. Although "command-and-control" regulatory instruments often figure in the policy mix, too – as in the implementation of

[12] California Senate Bill 535, signed into law in 2012, mandates that 25% of the revenue from the state's carbon permit auctions is to be spent on projects that benefit disadvantaged communities. For discussion, see Truong (2014).

California's Global Warming Solutions Act – carbon pricing can be an effective, if not indispensable, instrument both to drive emissions reductions in the short run and to create incentives for technological innovation in the long run. In addition, carbon pricing offers an opportunity to build and sustain public support for climate policy if the revenue – here termed "carbon rent" – is allocated in a manner that is transparent and widely regarded as fair.

From an administrative standpoint, a carbon cap or tax is most easily and efficiently implemented "upstream" where fossil fuels enter the economy: at tanker terminals, pipeline hubs, coal mine heads, etc. For each ton of fossil carbon that a firm brings into the economy, it surrenders a permit or pays a tax. In the US, an upstream system would entail roughly 2 000 collection points nationwide, far fewer than the number of compliance entities that would need to be monitored in a downstream system (US Congressional Budget Office, 2001).

What is Carbon Rent?

The revenue generated by a carbon price is depicted in Figure 1.1. A cap reduces the quantity of fossil fuel from Q_0 to Q_1. A tax raises the price of fossil fuel from P_0 to P_1. A cap sets the quantity of carbon emissions and lets the price adjust, while a tax sets the price and lets the quantity adjust. Apart from this difference the two are equivalent. Carbon rent, represented by the shaded area in the diagram, is the product of the carbon price and the quantity of carbon in fossil fuel entering the economy.

The price elasticity of demand for fossil fuels is low, especially in the short run: the percentage change in price exceeds the percentage change in quantity. Hence the tighter the cap or higher the tax, the bigger the carbon rent.

Carbon rent is sometimes confused with the resource cost of reducing emissions, but the two are quite distinct. Investments in energy efficiency and alternative energy use real resources. A number of studies have concluded that the resource costs of emission reductions in response to the introduction of a carbon price will be fairly modest. In an analysis of the Waxman-Markey bill, an unsuccessful attempt to enact federal carbon pricing legislation in the US, the Congressional Budget Office (2009) estimated that the resource cost in the year 2020 would amount to only 18 cents per household per day.[13] Indeed, a study by McKinsey & Co. (2007) concluded that substantial emissions reductions can be achieved initially at negative marginal cost – that is, the investments would pay for themselves at market interest rates.

The difference between the resource cost of emission reductions and carbon rent is depicted in Figure 1.2. The horizontal axis is the quantity of emissions, starting from zero reduction (100 per cent of current emissions); the vertical axis is the price. The rising curve represents the marginal abatement cost, here shown beginning at zero (rather than in the negative range reported in the McKinsey study). The figure shows the effect of

[13] The 18 cents/day figure comes from dividing the CBO estimate of "net annual economywide cost" of $22 billion/yr by the US population (335 million). In addition to resource costs of energy efficiency and alternative fuels, the CBO's $22 billion estimate included costs for the purchase of international offsets and the production cost of domestic offsets (both of which would have been allowed under the bill) and overseas spending on adaptation and mitigation.

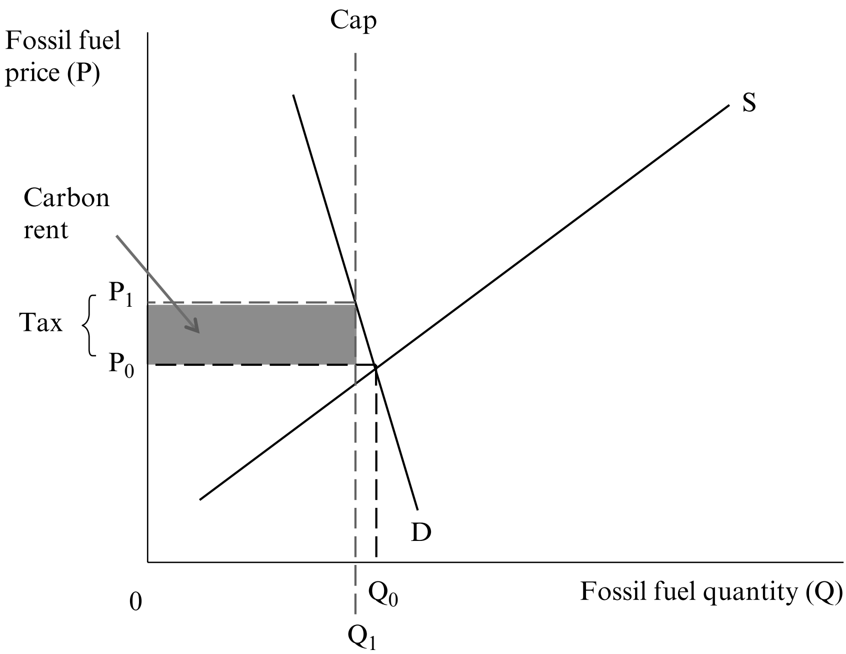


Figure 1.1 Carbon rent

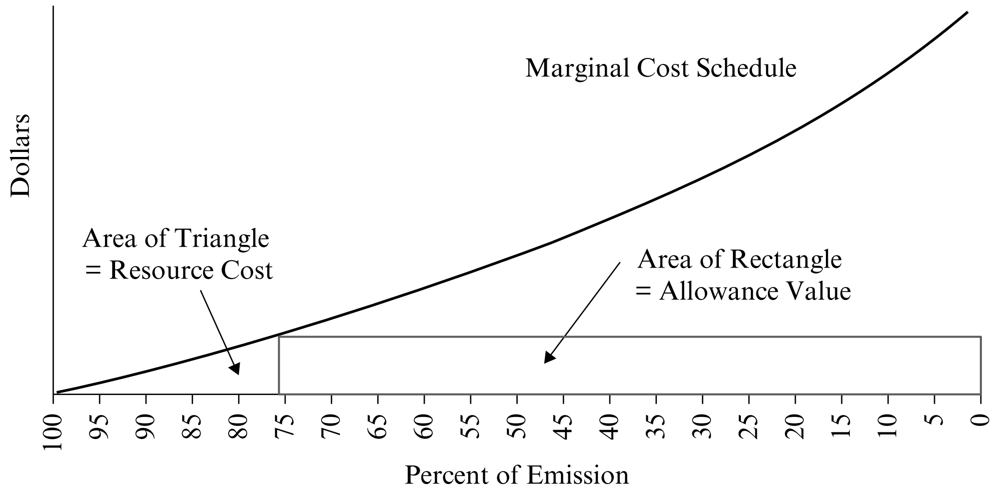
capping emissions at 75 per cent of their current level, or equivalently, setting a carbon tax at the level needed to obtain this outcome. The resource cost triangle is the cost of preventing emissions. The carbon rent rectangle is the price paid by fossil fuel users for emissions that are *not* prevented. This is often termed "allowance value" in discussions of cap-and-permit systems, where an allowance is a synonym for a permit.

As Figure 1.2 shows, the carbon rent generated by pricing policies is likely to be substantially larger than the resource cost of reducing emissions. In economic terms, carbon rent is not a cost: it is a *transfer*. The carbon rent is not spent on retrofitting buildings or installing solar panels. It is a surcharge paid on fossil fuel resources that would have been produced even in the absence of the policy.

Who Pays Carbon Rent?

The tax or permit price is paid by the compliance entities – fossil fuel firms in an upstream system – and then passed through to final consumers, either directly in the market prices of gasoline, heating fuels and electricity, or indirectly in the market prices of food, manufactured goods, and everything else that is produced or distributed using fossil fuels.[14] The extra money paid by consumers is the main source of carbon rent (as

[14] Most economic analyses assume that 100% of the carbon price is passed through to consumers. In practice, it is possible that "pass-through" would be a little less than 100% (or even



Source: Burtraw et al. (2009).

Figure 1.2 Resource cost versus allowance value (carbon rent)

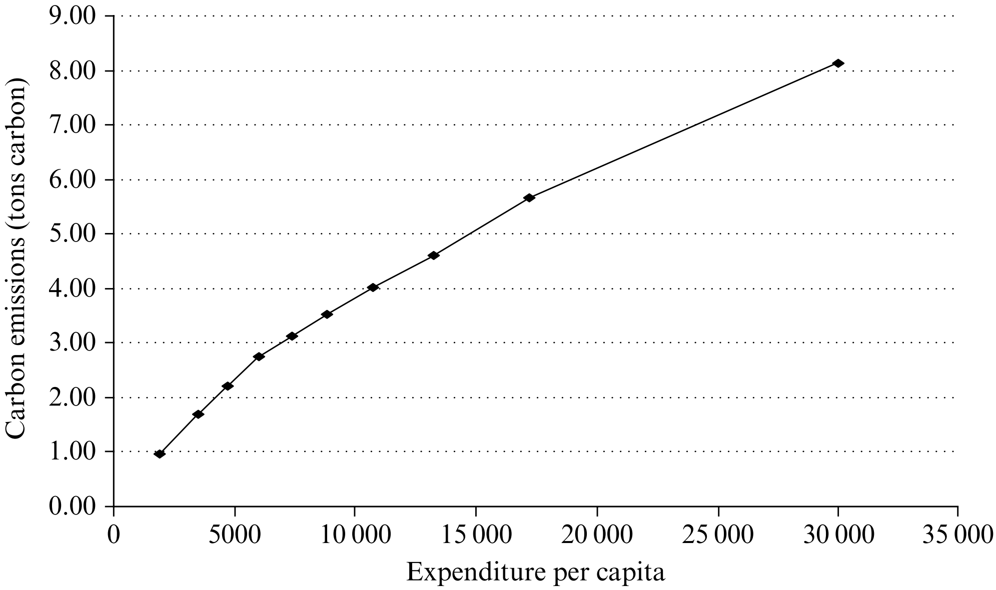
noted below, some also comes from non-household final users of fossil fuels). Consumers pay in proportion to their direct and indirect consumption of fossil fuels, their "carbon footprints." Because upper-income households generally have bigger carbon footprints, they pay more in absolute terms than other households. As a percentage of their incomes, however, lower-income households may pay more. If so, carbon pricing is a regressive tax.

The incidence of carbon pricing can be analyzed by combining consumer expenditure survey data with input–output tables that provide information on the quantities of fossil carbon embodied in different goods and services. Figure 1.3 depicts the results of such calculations for US households. The relationship is concave: carbon footprints rise with total household expenditure, as expected, but decline as a percentage of expenditure. Similar patterns have been found in a number of other industrialized countries.[15]

There have been fewer studies of the distributional effects of carbon pricing in low and middle-income countries. In some of these countries, low-income households may have smaller carbon footprints than upper-income households not only absolutely but also in relative terms, as a percentage of total expenditure, by virtue of their very low consumption of fossil fuels. Figure 1.4 depicts the relationship between carbon emissions and household expenditure in China in the year 1995. The convex curve indicates that at that time, the incidence of carbon pricing in China would have been progressive.

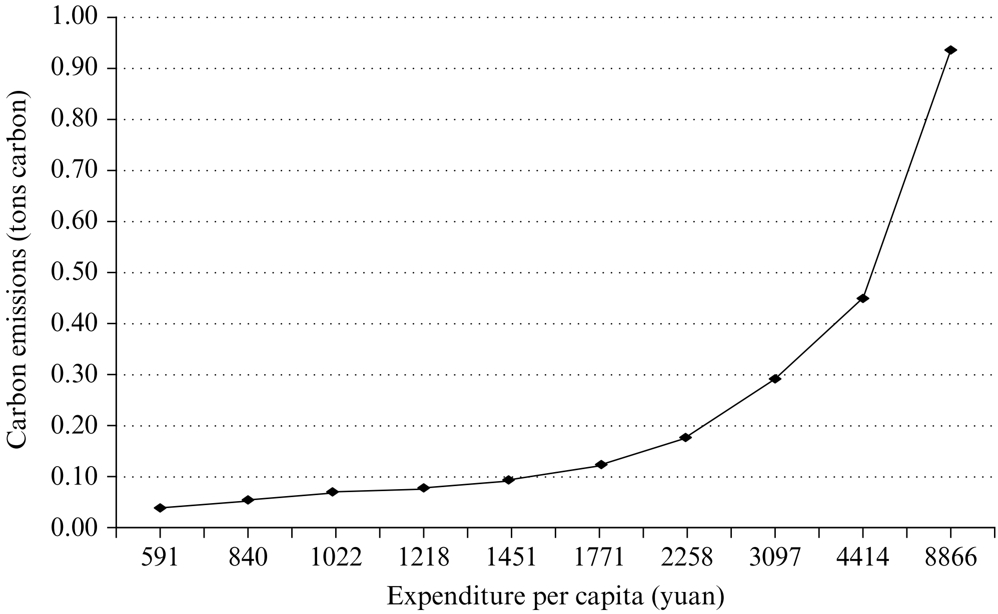
a little more), if firms cut their profit margins in an effort to protect their market share (or use the policy as a pretext to increase profit margins). For discussion of the effects of the degree of pass-through on carbon rent, see Boyce and Riddle (2007).

[15] See, for example, Cramton and Kerr (1999); Symons et al. (2000); and Wier et al. (2005).



Source: Boyce and Riddle (2007).

Figure 1.3 Carbon emissions by expenditure class, United States



Source: Based on data in Brenner, Riddle and Boyce (2007).

Figure 1.4 Carbon emissions by expenditure class, China (1995)

Who Receives Carbon Rent?

The net impact of carbon pricing on income distribution, however, depends crucially on where the money goes: to whom the carbon rent is transferred. Broadly, there are three possibilities:

- *Cap-and-giveaway-and-trade* policies (usually called "cap-and-trade") distribute carbon permits to firms free of charge, allocating them on a formula based on historic emissions.[16] The European Union's Emissions Trading System for power plants and industrial point sources is an example. Much as OPEC increases profits for member countries by restricting oil supplies, the firms that receive free permits reap windfall profits by virtue of the cap: the supply of fossil fuels is reduced, prices go up, and suppliers keep the money. Under this policy the carbon rent ultimately flows to the shareholders and executives of firms that get free permits. Because these recipients generally are upper-income households, the effect of the rent distribution is regressive, compounding whatever regressivity exists in the incidence of the carbon price itself.
- *Cap-and-spend* policies auction carbon permits up to the limit set by the cap rather than giving them away. The auction revenue goes to the government. This equals the carbon rent, since what firms bid for permits is equal to what they will recoup in higher prices from buyers of fossil fuels. Government then can use the revenue for public expenditures, tax cuts (or "tax expenditures"), budget deficit reduction, or some combination of these. The Regional Greenhouse Gas Initiative for power plants in the northeastern US states is an example of such a policy. A carbon tax in which the revenues are deposited into the government treasury is another. Under this policy the net distributional impact of carbon pricing depends on how the government chooses to spend the carbon revenue.
- *Cap-and-dividend* policies auction the permits, too, but under this option the revenue is returned to the public as equal per capita dividends rather than being retained by the government. The underlying normative principle behind cap-and-dividend policy is that the scarce carbon absorptive capacity of the biosphere, or more precisely, a state's share of it, belongs in common and equal measure to all its people rather than to firms or the government.[17] Under this policy the net distributional impact is progressive, since high-income households pay more in absolute terms than low-income households (as shown in Figures 1.3 and 1.4), while all individuals receive the same dividends. A carbon tax in which the revenues are returned to the public as lump-sum payments (this is sometimes called "fee-and-dividend") has the same effect.

[16] Because the permits are given to them free of charge, rather than auctioned, some firms may find it profitable to sell permits to others who find it profitable to buy them. For this reason, permit trading is invariably allowed in the cap-and-giveaway policy.

[17] A similar principle can be applied to royalties from natural resource extraction. For example, oil revenues paid into the Alaska Permanent Fund provide annual dividends to all state residents (Barnes, 2014). For more on carbon dividends, see Boyce (2019).

Table 1.5 Net incidence of a carbon dividend policy in the United States

Expenditure decile	Net impact (USD/household/year)[1]	
	Scenario 1: 100% as dividends	Scenario 2: 75% as dividends[2]
1 (poorest)	289	190
2	253	154
3	225	126
4	201	102
5	175	76
6	148	49
7	117	18
8	77	−22
9	18	−81
10 (richest)	−109	−200

Notes:
[1] Net impact = dividend minus amount paid in higher prices for fossil fuels.
[2] Excludes impacts from the 25% of carbon rent allocated to other purposes.

Source: Author's calculations; for methods, see Boyce and Riddle (2011).

Table 1.5 shows the net impacts of carbon dividends in the United States under two scenarios. Both assume a modest carbon price of $25 per ton of CO_2. In the first scenario, 100 per cent of the carbon rent is recycled directly to the public as equal per capita dividends. This was proposed in a climate policy bill introduced by Congressman Chris Van Hollen in July 2014.[18] In the second scenario, 75 per cent of the carbon rent is recycled as dividends and 25 per cent is retained for public investment. This was proposed in the climate policy bill introduced by Senators Maria Cantwell and Susan Collins in December 2009.[19]

In either scenario, the majority of households would receive positive net benefits: their dividends would exceed what they pay as a result of higher fossil fuel prices.[20] There are two reasons for this result. First, because household income and expenditure are highly concentrated in the upper deciles, so are carbon footprints. The mean household carbon use is above the median, and dividends are based on the mean. Second, household consumption accounts for roughly two-thirds of fossil fuel use in the US. The remainder is consumed mainly by governments (federal, state, and local), and to a lesser extent by non-profit institutions and production of exports (Boyce and Riddle, 2008). If more than two-thirds of the total carbon rent is returned to households, they receive a transfer from these other sectors.

[18] See Boyce (2014a).
[19] For details, see Boyce and Riddle (2011).
[20] This result holds not only at the national level but also in each of the 50 states, although the percentage of households that would come out ahead varies depending, in particular, on the carbon-intensity of the state's electricity supply (Boyce and Riddle, 2009).

A 2017 study by the US Treasury Department's Office of Tax Analysis reached similar conclusions. The study estimated that a carbon price of $49 per ton of CO_2 would generate annual dividends of $583 per person. Roughly 70 per cent of households would experience net income gains, and the distributional impact would be strongly progressive, ranging from an 8.7 per cent gain in net income for the poorest decile to a 1.0 per cent loss for the top decile. In contrast, using the carbon revenue to reduce corporate tax rates would have a regressive impact, resulting in net income losses for the lowest nine deciles and net gains only for the top decile (Horowitz et al., 2017, Table 6).

An Efficiency–Equity Tradeoff?

Some economists argue that the choice between carbon dividends and a revenue-neutral "green tax shift," in which carbon revenues are offset by tax cuts, poses a tradeoff between equity and efficiency (Burtraw and Sekar, 2014). Equal per capita dividends would be more progressive, as illustrated above, but cuts in income or sales taxes would boost output since these taxes reduce the supply of labor and capital.

The validity of this ostensible equity–efficiency tradeoff can be questioned on three grounds. First, in real-world contexts characterized by unemployed labor and underutilized capital, increases in the supply of labor and capital do not translate into increased output; instead they translate into more unemployment and excess capital. Second, from a social welfare standpoint the main problem may be that people work too much, as predicted by the relative income hypothesis, rather than too little (Wendner and Goulder, 2008). Third, higher output (as measured by GDP) is not synonymous with higher social welfare.

Apart from its appeal on equity grounds, an attractive feature of the carbon dividend option is that it could help to ensure durable public support for climate policy even in the face of rising fuel prices. Effective climate policy is not a one-shot game in which success is simply enacting legislation. Once in place, the policy must be able to continue over the decades needed to complete the clean energy transition. It must be popular enough to survive no matter what party controls the government. It is hard to imagine how robust public support can be secured in the face of rising fuel prices unless the carbon rent is returned to the population in a way that visibly and fairly protects the net incomes of most households. Additionally, it is hard to imagine any outcome that would be more inefficient than the failure to curtail the use of fossil fuels.

4. CONCLUSIONS

The distributional issues in climate policy are often posed in inter-generational terms, as a tradeoff between the welfare of present and future generations. This chapter has argued that climate policy has important distributional implications within the present generation, too, and that policies that take these into account can attenuate the myopia and free-rider problems that have impeded efforts to curtail the use of fossil fuels.

Two crucial intra-generational issues have been explored here. The first relates to the air quality co-benefits of reduced use of fossil fuels. By some calculations, these benefits are as large or larger than the climate benefits themselves. Because air quality co-benefits are local rather than global, and because co-benefits vary spatially and across pollution sources, the ways in which these benefits are (or are not) integrated into climate policy can have important distributional implications. Co-pollutant damages often are greatest in lower-income and politically disenfranchised communities. Therefore, designing climate policy to achieve greater emission reductions where they yield the greatest public health benefits can promote equity as well as efficiency.

The second distributional issue concerns the allocation of carbon rent. Climate change mitigation policy is a form of property creation, in that it converts the limited carbon-absorptive capacity of the biosphere from an open-access resource (where property rights are absent) into a resource governed by rights and responsibilities. When carbon pricing is in the policy mix – in the form of a carbon tax or cap-and-permit system – the new bundle of property rights includes the right to receive income from payments for use of the scarce resource. The allocation of this income, or carbon rent, again poses important distributional issues. These are illustrated by the choice among the cap-and-giveaway-and-trade, cap-and-spend, and cap-and-dividend policy options.

If climate policy addresses these distributional issues in an egalitarian fashion – based on the twin principles of equal rights to a clean and healthy environment and equal rights to carbon rent – the outcome can be positive net benefits for the majority of people in the present generation. These health and income benefits can attenuate or eliminate the ostensible tradeoff in climate policy between present and future welfare. In turn, this could help to overcome one of the greatest political obstacles to taking effective steps to safeguard the world's climate.

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