Distributional Issues in Climate Policy:
Air Quality Co-benefits and Carbon Rent

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Abstract

The case for climate policy typically is made on grounds of inter-generational equity, with a presumed tradeoff between the environmental interests of future generations and the economic interests of the present generation. This framing of the problem neglects the scope for designing policies that not only mitigate climate change but also yield net benefits for all or most people who are alive today. This chapter considers two avenues by which climate policy can bring substantial immediate gains to the present generation, via (i) air quality co-benefits from reduced use of fossil fuels; and (ii) recycling of rent created by carbon pricing. Both avenues pose important issues of equity within the present generation.

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1. Introduction

Climate change mitigation policy often is framed as posing a difficult tradeoff between the welfare of the present generation and that of future generations. Policies that aim to curtail climate change – most importantly, by reducing carbon dioxide emissions from 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 gains from mitigation will outweigh the present pain. Opponents make the opposite argument.[1]

This framing of climate policy has been most popular among elite environmentalists. But its appeal across a broader spectrum of the public – including among working people whose support is necessary to enact and sustain effective climate policies in democracies – has proven to be more limited.

It is possible, however, to design climate change mitigation policies that overcome this tradeoff by bringing near-term benefits to people – and voters – who are here today. Potential benefits of climate policy for the present generation extend beyond the substantial numbers of jobs that can be created through investments in the energy efficiency and renewable energy,[2] and include benefits for the public as a whole. This chapter discusses two of these: the air quality co-benefits from reduced use of fossil fuels, and the income gains from egalitarian distribution of the rent created by carbon pricing.

Inter-generational equity – in particular, the choice of a discount rate in integrated assessment models – has been a contentious issue in climate policy debates. The Stern review, commissioned by the UK government, advocated a discount rate of about 1.4%, for example, while William Nordhaus of Yale has advocated rates of 4% or higher.[3] The argument for using a higher discount rate hinges, in part, on forecasts of the average incomes of future generations. Citing the Stern review's assumption that global per capita income will rise from $10,000 today to around $130,000 (in today's dollars) two centuries from now, Nordhaus (2008) remarks: "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." If, however, climate policy can yield positive net benefits for the present generation as well as future generations, the choice of the discount rate becomes less consequential. Instead questions of *intra-generational* equity come to the fore, including the distribution of air quality co-benefits and carbon rent.

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[1] In this chapter, my focus is solely on climate change mitigation policy. Climate change adaptation also poses important distributional issues; for discussion, see Boyce 2014.

[2] On employment creation potential of "green growth," see Pollin 2015.

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

2. Air Quality Co-Benefits

In addition to reducing carbon dioxide emissions, policies to curb the use of fossil fuels generate air quality co-benefits via reduced emissions of co-pollutants such as particulate matter, sulfur dioxide, nitrogen oxides, carbon monoxide and a host of other toxic chemicals. Indeed, exercises that assign monetary values to the long-term damages from carbon dioxide emissions and the short-term damages from co-pollutant emissions often put higher values on the latter. Because co-pollutant intensity (the ratio of co-pollutant damages to carbon dioxide emissions) varies across pollution sources, attention to the distribution of air quality co-benefits is important from the standpoints of both efficiency and equity.

Magnitude

The World Health Organization (WHO) characterizes air pollution as "the world's largest single environmental health risk," concluding that outdoor and indoor air pollution together are responsible for one in eight deaths. The WHO reports that outdoor air pollution is responsible for 3.7 million premature deaths annually worldwide. Observing that air pollution often is a by-product of unsustainable policies in transport, energy and other sectors, the WHO remarks that in most cases, more sustainable strategies "will also be more economical in the long term due to health-care cost savings as well as climate gains" (WHO 2014).

A number of researchers have assessed the costs of outdoor air pollution. A multi-country study conducted by the Organisation for Economic Co-operation and Development (OECD) estimated that outdoor air pollution (specifically, particulate matter and ozone) was responsible for 2.45 million premature deaths in the OECD countries plus China and India in 2010 (see Table 1). China accounted for roughly half of this total, followed by India; among OECD countries, the US ranked highest with roughly 110,000 deaths. To monetize health impacts, the OECD multiplied the number of deaths by the value of a statistical life (VSL), calculated as a concave function of per capita income, adding 10% to account for the additional costs of non-fatal illnesses. The study put the total annual cost in 2010 at $3.5 trillion, about half in the OECD countries and the other half in China and India (OECD 2014).

[insert Table 1 here]

Because the use of country-specific VSLs affects on international comparisons of the potential air quality co-benefits from climate policy, the OECD's rationale for this procedure is worth quoting at some length:

> *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 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 (OECD 2014, pp. 53-55).

An alternative valuation procedure would be to apply a uniform VSL in all countries, consistent with the ethical principle that the value of a person's life should not be based on individual wealth nor on income per capita in the country in which a person happens to reside.

Outdoor air pollution is caused not only by fossil fuel use but also by wildfires, burning crop residues and other biomass, dust from construction debris, and other industrial and agricultural practices. To assess the magnitude of health impacts attributable to fossil fuels therefore requires apportionment of air pollution across sources. Reliable source apportionment data at the national level are generally sparse. The OECD (2014, p. 63) reckons that road transport accounts for roughly half of outdoor air pollution in the EU24, and that in the US the road transport share is about one-third of the total due to a higher share from electric power generation.

A study by MIT researchers estimates total premature deaths in the US due to combustion emissions in 2005 at 211,875 (with a 90% confidence interval of 91,000 – 383,300), and apportions these deaths across economic sectors (Caiazzo et al. 2013).[4] Transportation and electric power generation account for 60% of the total, with the remainder coming from industrial, commercial and residential sources (Table 2). If we assume that all transportation and electric power emissions and one-quarter of emissions from other sectors result from fossil fuel combustion, then 70% of outdoor air pollution deaths in the US are attributable to use of fossil fuels.

[insert Table 2 here]

An international analysis by Lelieveld et al. (2015) that includes non-combustion sources of outdoor air pollution (such as ammonia releases from fertilizer and livestock, which form secondary particulates) reports similar estimates for the US, with 52% of mortality in 2010 attributed to electric power generation and land traffic. In other OECD countries included in their analysis, the share of these two sectors ranges from 27.3% (in Korea) to 35.5% (in the UK). In the cases of China and India, the study reports lower estimates for

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[4] The difference between the total US deaths estimated by Caiazzo et al. (2013) and the OECD (2014) may be attributable, in part, to emissions reductions between 2005 and 2010. See also Fann et al. (2013).

these sectors, 20.8% and 18.5% respectively, with residential and commercial energy accounting for larger shares. If again we attribute all transportation and power sector emissions and one-quarter of the remainder to fossil fuel combustion, this implies that fossil fuel use accounts for roughly 65% of premature mortality due to outdoor air pollution in the US, roughly 50% in other OECD countries, and roughly 40% in China and India. Insofar as carbonaceous particulates are more hazardous than other species of particulate matter (Caiazzo et al. 2013), these estimates may understate the share of fossil fuel emissions in total health impacts.

Applying these estimates to the data in Table 1, we can calculate the health impacts of fossil fuel emissions per ton carbon dioxide. I term this ratio the "Co-Pollutant Cost of Carbon" (CPCC). Three variants of the CPCC are reported in Table 3. The first, deaths/ton CO_2, ranges from less to 13 in the United States to more that 160 in India. The second, USD/ton using the OECD's valuation method, ranges from $50 in Mexico to $134 in Italy. By this measure, Italy and Turkey have CPCCs than India, despite the fact that India has more than three times more deaths/ton. The final measure recalculates monetary damages/ton by applying an equal VSL to all countries while holding the sum total of monetary damages unchanged. By this measure, which is directly proportional to deaths/ton, India's CPCC is more than $200/ton.

[insert Table 3 here]

By way of comparison, the "social cost of carbon" used in regulatory analyses by the US government in 2015 as an average measure of climate change damages ranges from $11 to $56 per ton of CO_2, depending on the discount rate; with a figure of $105/ton used as a sensitivity test "to represent the potential for higher-than-average damages" (US Environmental Protection Agency 2015).[5] The OECD valuation-based estimate of the co-pollutant cost of carbon for the US, $64/ton, lies above the highest of these average measures.

Other studies have reached similar conclusions. An analysis of the health co-benefits of carbon emissions reductions in the European Union, conducted for the Netherlands Environmental Assessment Agency, concluded that the air quality co-benefits of a stringent climate policy would offset the policy's costs "even when the long-term benefits of avoided climate impacts are not taken into account" (Berk et al. 2006). Summarizing 37 studies of air quality co-benefits from around the world, Nemet et al. (2010) found a mean co-benefit of $49 per ton of CO_2. An IMF study (Parry et al. 2014) of domestic co-benefits of carbon pricing in the top twenty CO_2 emitting countries concluded that without counting global climate benefits the average nationally efficient carbon price would be $57.5/ton. An analysis of potential air quality co-benefits from "deep decarbonization" energy and transportation policies in the US found that they

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[5] For details, see US Interagency Working Group on the Social Cost of Carbon (2013). For a critique, see Ackerman and Stanton (2012).

would prevent roughly 36,000 premature deaths annually from 2016 to 2030, and concluded that these near-term health benefits exceed climate benefits valued on the basis of the US government's social cost of carbon (Shindell et al. 2016).

The salience of air quality co-benefits for climate policy may be even greater than these valuation exercises suggest. The former are predominantly national and near-term; the latter are predominantly global and long-term. As Shindell (2015) notes, "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."

Efficiency implications

From the standpoint of efficiency, two conclusions follow. First, inclusion of the air quality co-benefits of emissions reduction justifies more stringent regulatory interventions (higher carbon prices, for instance) than if policy were based on CO_2 emissions alone. Second, insofar as airs quality co-benefits per ton of CO_2 vary across sources and locations, efficiency can be enhanced by designing policies to achieve larger emissions reductions where the co-benefits are greater.

The full social cost of carbon (FSCC) – where this term signifies environmental damage per ton of fossil carbon – comprises both the climate change cost of carbon (CCCC) and the co-pollutant cost of carbon (CPCC). That is,

FSCC = CCCC + CPCC

Use of this more complete measure by policy makers in the US and elsewhere would buttress the usual efficiency case for curbing use of fossil fuels, and provide a yardstick for more ambitious carbon emission reduction targets.[6]

The rationale for taking into account differences in abatement benefits across emission sources can be illustrated by considering two facilities in California: a power plant located outside Bakersfield, and a petroleum refinery located in metropolitan Los Angeles. Each emits roughly 3 million tons per year (t/yr) of carbon dioxide. The power plant emits about 50 t/yr of particulate matter (PM) and has fewer than 600 residents living in a 6-mile radius. 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 carbon emission reductions are likely to be greater in the case of the latter facility.

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

This is a dramatic example, but it is not unique. In an analysis of variations in co-pollutant intensity for industrial point sources in the US, Boyce and Pastor (2013) found substantial differences across industrial sectors. For example, emissions of $PM_{2.5}$ (fine particulates with a diameter of 2.5 micrometers or less) per ton of CO_2, weighted by the population living within a 2.5-mile radius of the facility, on average are almost three times higher for petroleum refineries for than for power plants (see Table 4). Population-weighted $PM_{2.5}$ emissions from primary metal manufacturing facilities are more than six times higher than from power plants.

[insert Table 4 here]

Equity implications

Air quality co-benefits matter from the standpoint of distributional equity, too. In the US, for example, a substantial body of literature has documented the fact that racial and ethnic minorities and low-income communities tend to bear disproportionate pollution burdens (see, for example, Ringquist 2005; Mohai 2008; Morello-Frosch et al. 2011). Again, the extent of disparities differs across industrial sectors and locations (Zwickl et al. 2014). For example, Boyce and Pastor (2013) found that minorities bear 59.5% of the impact of population-weighted $PM_{2.5}$ from petroleum refineries in the US, compared to 38.8% of the impact from power plants and to their 34.2% share in the national population (see Table 4).

Executive Order 12898, "Federal Actions to Address Environmental Justice in Minority Populations and Low-Income Populations," issued by President Bill Clinton in 1994, directs each US government agency to takes 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 order makes equity an explicit objective in federal environmental policy. Many US states have adopted similar environmental justice policies (Bonorris 2010).

If climate policy fails to incorporate air-quality co-benefits into its design, these findings suggest that it will fall short by the criteria of both efficiency and equity. It will be inefficient by virtue of forgone public health benefits, by setting overly modest targets for emission reductions and by failing to target reductions across pollution sources to take into account variations in the CPCC. It will be inequitable by virtue of foregone opportunities to redress disparities in air quality across the population.

Policy options

Two broad types of policy instruments are used to curb carbon emissions from 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 two types are not mutually exclusive, since policies can and often do combine both.

An attractive feature of price-based policies is that they provide incentives not only to adopt existing emissions-reduction technologies, but also to invest in research and development of new technologies. Price-based policies often have encountered opposition, however, from environmental justice advocates on the grounds that they can allow co-pollutant "hot spots" to persist, and possibly worsen, in overburdened communities. Environmental justice organizations in California, for example, filed a lawsuit in an attempt to block implementation of the state's cap-and-trade program for this reason (Farber 2012).

Some economists have argued that co-pollutants are best regulated separately, and should not be factored into climate policy design (Schatzki and Stavins 2009). But unless we can reasonably assume that co-pollutant impacts are adequately addressed by separate environmental policies, efficient and equitable climate policy should take them into account – an approach consistent with the growing interest in multi-pollutant strategies for air-quality management.[7] The large health costs of air pollution, compared to the relatively modest costs of pollution control, imply that it would be a mistake to assume that these are adequately addressed by other policies. As the OECD (2014, p. 76) remarks, the very high benefit-cost ratios in air pollution control "suggest that something has gone wrong with the decision-making process."

The administrative costs of incorporating air quality co-benefits into climate policy design could be modest, particularly in situations where co-pollutant damages are concentrated in a small number of sectors or facilities, as is the case for industrial point source emissions in the US (Boyce and Pastor 2013). Policy options include the following:

1. *Monitor impacts on co-pollutants:* The minimalist option is to monitor co-pollutant emissions, with a view to instituting remedial measures if the climate policy has unacceptable impacts. This is 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:* Zonal carbon permit or tax systems can ensure emissions reductions in high-priority locations where the potential public health benefits are greatest. Zone-specific caps were applied, for example, in California's Regional Clean Air Incentives Market, initiated in 1994 to reduce emissions of NO_x and SO_2 in the Los Angeles basin (Gangadharan 2004).

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[7] On multi-pollutant strategies, see National Academy of Sciences 2004 and McCarthy et al. 2010.

3. *Sectoral tax or permit systems:* Similarly, sector-specific tax or permit systems can be designed to ensure emissions reductions in the industrial sectors with the highest co-pollutant intensities and the most 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 be another policy instrument. 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 or sector A as in location or sector B, the exchange rate ("trading ratio") would be 1:2 (Muller and Mendelsohn 2009).

5. *Community benefit funds:* Some percentage of the revenue from carbon taxes or permit auctions (the "carbon rent" discussed in the next section) can be channeled into community benefit funds to mitigate pollution impacts and protect public health in overburdened and vulnerable communities. Such a policy has been enacted for revenues from permit auctions under the California's climate policy.[8]

3. Carbon Rent Allocation

Putting a price on carbon emissions via a cap or tax is often regarded as a central element of climate policy, for good reason. Although "command-and-control" regulatory instruments are often part of the policy mix, too – as in the implementation of California's Global Warming Solutions Act – carbon pricing is useful both to spur emissions reductions in the short run and to provide incentives for technological innovation in the long run. In addition, carbon pricing offers unique opportunities to build durable public support for climate policy if the revenue – here termed "carbon rent" – is allocated in a fair and transparent manner.

From the standpoint of administrative costs, a carbon cap or tax is most efficiently implemented "upstream," where fossil fuels enter the economy: at tanker terminals, pipeline hubs, coal mine heads, and so on. For each ton of carbon the firm brings into the economy, it must pay a tax or surrender a permit. In the US, the Congressional Budget Office (2001) has estimated that such a system would involve about 2,000 collection points nationwide, far fewer compliance entities than would exist in a downstream system.

The carbon rent created by a cap or tax on the use of fossil fuels is depicted in Figure 1. A cap reduces the quantity of fossil fuel from Q_0 to Q_1. A tax raises the price of fuel from

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[8] 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.

P_0 to P_1. A cap sets the quantity of carbon emissions and lets the price adjust; a tax sets the price and lets the quantity adjust. The policy choice between a cap and a tax often is a subject of spirited debate, but apart from this difference the two are equivalent. Carbon rent is the revenue created by the cap or tax; that is, by policy-induced scarcity in the supply of fossil fuels. Because the price elasticity of demand for fossil fuels is low, the percentage increase in price exceeds the percentage decrease in quantity. For this reason, the tighter the cap (or higher the tax), the bigger the carbon rent.

[insert Figure 1 here]

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 absorb 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 modest. An analysis of the Waxman-Markey bill, the last attempt to enact federal carbon pricing legislation in the US, by the Congressional Budget Office (2009) estimated that the resource cost in the year 2020 would amount to only 18 cents per household per day.[9] In fact, some studies have concluded that substantial emissions reductions initially can be achieved at negative marginal cost – that is, the investments would pay for themselves at market interest rates (McKinsey & Co. 2007).

The difference between the resource cost of emission reductions and carbon rent is depicted in Figure 2. The horizontal axis is the quantity of emissions, starting from zero reduction (100 percent 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 territory reported in the McKinsey study). The figure shows the effect of capping emissions at 75% of their current level, or equivalently, setting a carbon tax at the level to yield this outcome. The resource cost triangle is the cost of preventing emissions. The carbon rent rectangle is the price paid for emissions that are *not* prevented. The latter is often termed "allowance value" in discussions of cap-and-permit systems, where allowances are a synonym for permits.

[insert Figure 2 here]

As Figure 2 suggests, 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*. This money is not spent on retrofitting buildings

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[9] 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.

or installing solar panels; it is a surcharge paid on fossil fuel resources that would be produced even in the absence of the policy.

The tax or permit price is paid by the compliance entities – fossil fuel firms in an upstream system – and then passed through to 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.[10] The extra money paid by consumers is the main source of carbon rent (as discussed below, some also comes from non-household final users of fossil fuels, notably government). Consumers pay in proportion to their direct and indirect consumption of fossil fuels, their "carbon footprints." Because upper-income households generally have larger carbon footprints, in absolute terms, than other households, they pay more. As a percentage of their income and expenditure, 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 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 relationships have been found in studies of other industrialized countries.[11]

[insert Figure 3 here]

There have been fewer studies of the incidence 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 even as a percentage of total expenditure, by virtue of their very low consumption of fossil fuels. Figure 4 depicts the relationship between carbon emissions and household expenditure in China in the year 1995. The convexity of the curve indicates that at that time, the incidence of carbon pricing in China would have been progressive.

[insert Figure 4 here]

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

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[10] 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 a little more), if fossil fuel firms cut profit margins in an effort to protect their market shares (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.

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

- *Cap-and-giveaway-and-trade policies* (usually simply termed "cap-and-trade") give carbon permits to firms free of charge, based on a formula based on historic emissions.[12] The European Union's Emissions Trading System for power plants and industrial point sources is an example. The firms that receive free permits reap windfall profits, much as OPEC profits by restricting oil supplies: when the supply of fossil fuels is reduced, prices go up, and suppliers get the money. In this policy option, carbon rent ultimately flows to shareholders and executives of the firms that get free permits. Because the recipients generally are upper-income households, the distributional effect is regressive.

- *Cap-and-spend policies* auction carbon permits rather than giving them away, with the revenue going to the government. The total auction revenue equals the carbon rent, since what firms bid for permits is equal to what they can recoup in higher prices from the buyers of fossil fuels. Government then uses the revenue for public expenditures, tax cuts (or "tax expenditures"), or budget deficit reduction. 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 flow to the government treasury is an equivalent policy. In this policy option, the net distributional impact depends on how the government chooses to use the carbon revenue.

- *Cap-and-dividend policies* auction the permits, too, but in this policy option the revenue is returned to the public as equal per capita dividends rather than retained by the government. The underlying normative principle behind a cap-and-dividend policy is that the scarce carbon absorptive capacity of the biosphere (or more precisely, a nation's or state's share of it) belongs in common and equal measure to all its people, rather than to firms or the government.[13] A carbon tax in which the revenues are returned to the public in lump-sum payments (sometimes called "fee-and-dividend") is an equivalent policy. The net distributional impact would be progressive, since high-income households pay more in absolute terms than low-income households (as shown in Figures 3 and 4), while all individuals receive the same dividend payments.

The net impacts of a cap-and-dividend policy in the United States are shown in Table 5 under two policy scenarios. Both assume a carbon price of $25 per ton of CO_2. In the first scenario, 100% of the carbon rent is recycled directly to the public as equal per capita dividends. This was proposed in the climate policy bill introduced by

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[12] Because the permits are given to them free of charge, rather than distributed via an auction, 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 part of the cap-and-giveaway policy.

[13] A similar principle is applied to royalties from petroleum extraction in the Alaska Permanent Fund, from which annual dividends are paid to all state residents (Barnes 2014).

Congressman Chris Van Hollen in July 2014 and reintroduced in February 2015.[14] In the second scenario, 75% of the carbon rent is recycled as dividends and 25% is retained for public investment. This was proposed in the climate policy bill introduced by Senators Maria Cantwell and Susan Collins in December 2009.[15]

[insert Table 5 here]

The majority of households would receive positive net benefits from either policy: their dividends would exceed what they pay as a result of higher fossil fuel prices.[16] There are two reasons for this result. First, because US household income is highly concentrated in the upper deciles, so, too, are carbon footprints: the mean household carbon use is above the median, and dividends are based on the mean. Second, household consumption accounts for only about 2/3 of fossil fuel use in the United States. The remainder is accounted for mainly by government (federal, state, and local), and to a lesser extent by non-profit institutions and production of exports (Boyce and Riddle 2008). If more than 2/3 of carbon rent is returned to households as dividends, they receive a transfer from these other sectors.

Some economists maintain that carbon rent allocation poses a tradeoff between equity and efficiency (see, for example, Burtraw and Sekar 2014). Equal per capita dividends (that is, lump-sum payments) would be more equitable than a revenue-neutral "green tax shift" in which carbon revenues are retained by the government and offset by cuts in income and/or sales taxes, for the simple reason that lower-income households typically pay less tax. Distributing the carbon rent via tax cuts, however, is claimed to be more efficient based on the "double dividend hypothesis," which holds that such a tax shift not only reduces the negative externality of carbon pollution (the first dividend) but also, by cutting distortionary taxes that reduce the supply of labor and capital, will lead to higher output (the second dividend).

The validity of this purported equity-efficiency tradeoff can be questioned on three grounds. First, in real-world contexts characterized by labor unemployment and capital underutilization, increases in supplies of labor and capital do not translate axiomatically into increased output; instead they translate into more idle labor and capital. Second, as suggested by the relative income hypothesis, many people may work too much, rather than too little (Wendner and Goulder 2008). Third, higher output (as measured by GDP) is not synonymous with higher social welfare.

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[14] Text available at http://vanhollen.house.gov/sites/vanhollen.house.gov/files/VANHOL_011_xml.pdf.

[15] For details, see Boyce and Riddle 2011.

[16] 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).

Apart from its appeal on equity grounds, an attractive feature of the cap-and-dividend policy is that dividends could help to secure durable public support in the face of rising prices for transportation fuels and electricity. Getting an effective climate policy is not a one-shot game of passing a piece of legislation. The policy must be politically robust enough to endure over the decades needed for the clean energy transition, whatever parties control the legislative and executive branches of government. It must be so popular among the voters that no party will be able to overturn the policy. It is hard to imagine how such durable support can be maintained in the face of rising energy prices unless the carbon rent is returned to the public in a fashion that is transparent and widely perceived as fair.

4. Concluding remarks

Insofar as distributional considerations enter into discussions of climate change mitigation policy, these have often been posed as tradeoffs between the welfare of present and future generations. This chapter has argued, however, that climate policy design has important distributional implications within the present generation, too.

Two key intra-generational issues have been explored here. The first concerns the air quality co-benefits that accompany reductions in the 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 potential co-benefits per ton of carbon vary across space, sectors and polluters, 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. For this reason, designing climate policy so as to achieve greater emission reductions where they yield the greatest health benefits can promote equity as well as efficiency.

The second distributional issue explored here concerns allocation of carbon rent. Climate change mitigation policy is a form of property creation, in the sense that it converts the carbon-absorptive capacity of the biosphere from an open-access resource (for which property rights are entirely absent) into a resource governed by a regime of rights that limits its use. When carbon pricing is part of the climate policy mix – via either a carbon tax or a cap-and-permit system – the bundle of property rights being created includes the right to receive income from payments for use of the scarce resource. The allocation of this income poses important distributional issues, illustrated by the choice among the cap-and-trade, cap-and-spend and cap-and-dividend policy options.

If climate policy design addresses these distributional issues in an egalitarian fashion – based on the principles of equal rights to clean air and equal rights to carbon rent – the result can be positive net benefits for the majority of people in the present generation. These health and income benefits can attenuate or eliminate the climate policy tradeoff

between present and future welfare. This, in turn, could help to resolve one of the greatest political obstacles to taking effective steps to safeguard the world's climate.

Acknowledgements

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Table 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	1 371.4
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**	**3 558.4**

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 = Costs of mortality + morbidity.

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

Table 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	8 830	4.2
Rail transportation	5 040	2.4
Aviation	1 355	0.6
Total	**211 875**	**100**

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

Table 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)		
			Deaths	US dollars	
				OECD VSL	equal VSL
China	511 556	7 388.5	69.2	74.2	100.6
India	276 970	1 714.9	161.5	106.9	234.6
US	71 690	5 580.0	12.8	63.6	18.7
Japan	32 888	1 177.3	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	2 588.3	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 Feb 2016.

Table 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.

Table 5: Net incidence of cap-and-dividend policy, 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.
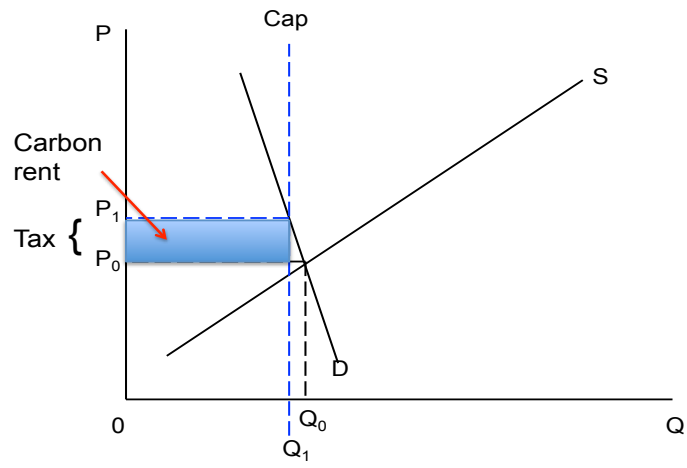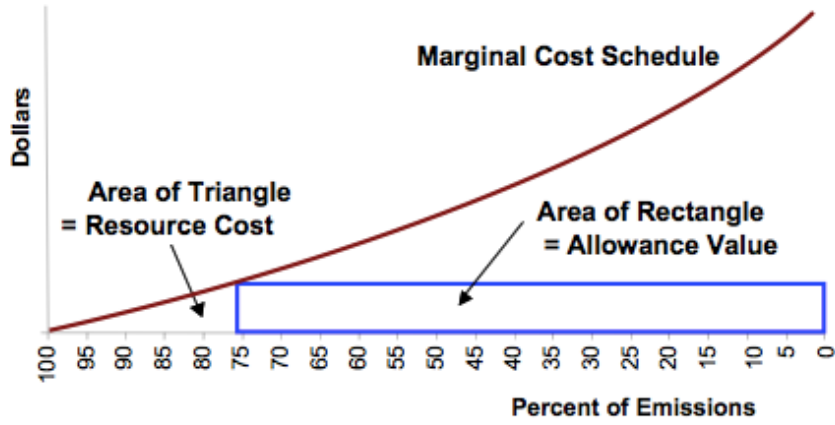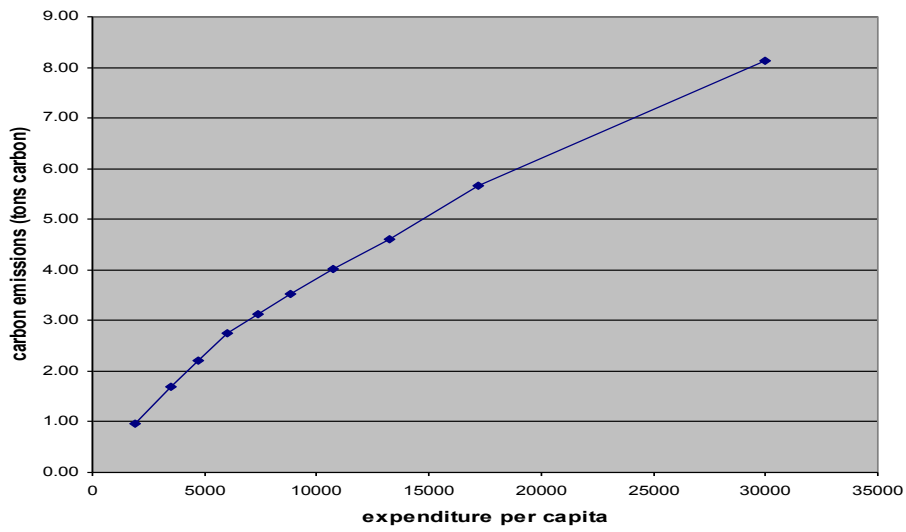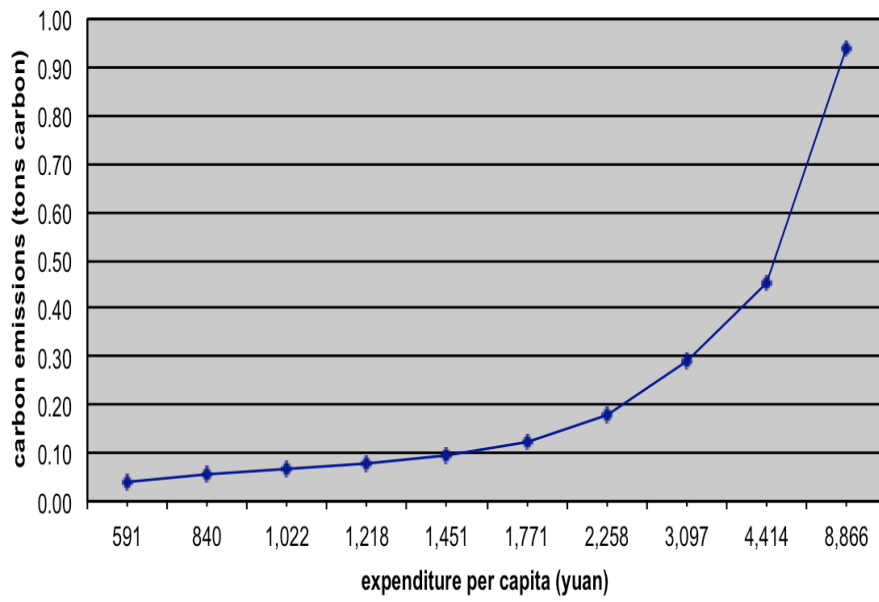
Figure 1: Carbon rent

Figure 2: Resource cost versus allowance value (carbon rent)

Source: Burtraw et al. 2009.

Figure 3: Carbon emissions by expenditure class, United States

Source: Boyce and Riddle 2007.

Figure 4: Carbon emissions by expenditure class, China (1995)

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