ANCILLARY BENEFITS ESTIMATION IN DEVELOPING COUNTRIES: A COMPARATIVE ASSESSMENT

by David O'CONNOR

1. Introduction

The possibility of reaping ancillary benefits from climate policy is generally accepted. The questions of their quantitative importance and how factoring them into the analysis might alter policy choices are still being actively explored. Thus far, most work has focused on public health benefits of reduced emissions of air pollutants associated with carbon dioxide (CO_2) as by-products of fossil fuel combustion – SO_2, NO_x, suspended particulates, volatile organic compounds (VOC), carbon monoxide (CO), and ozone (O_3). A smaller body of work has looked at other damages, notably crop damage from O_3, forest damage from SO_2 and materials damage from that and sulphate aerosols. The early studies of ancillary benefits of reducing CO_2 emissions were done in Europe and the United States, where the prospect of quantitative restrictions on those emissions appeared to be imminent. More recently, work has begun to focus on developing countries. The policy rationale is similar, viz., that decisions about desirable levels of CO_2 abatement need to be informed by as full an accounting as possible of both costs and benefits, and that the nearer-term, more certain and local ancillary benefits may well carry more weight with policy makers than the longer-term, more uncertain and global ones from climate change mitigation.

This paper reports on work-in-progress at the OECD Development Centre (*DevCentre*) that makes use of computable general equilibrium (CGE) models to integrate ancillary benefits estimation into an economy-wide assessment of climate policy. To date, a study has been completed on the ancillary benefits of climate policy in Chile (Dessus and O'Connor 1999), and one is currently underway for India (Bussolo and O'Connor 2000 forthcoming). Both focus on public health benefits of climate policy in consequence of reduced local air pollution. A study on China is also planned, which will focus on the effects of reduced air pollution on crop yields, considering in the first instance O_3 but with possible extension in a second phase to include particulate haze. The paper compares, wherever possible, both the methodology and the results of this research effort with those of other studies undertaken for the same set of countries. In particular, Cifuentes *et al.* (1999) provides another set of estimates for Chile, using a "bottom-up" engineering approach, while Garbaccio *et al.* (2000) offers a CGE-based assessment of public health benefits of climate policy for China. These two studies provide useful comparators for the *DevCentre* studies, allowing the examination of the sensitivity of results to specific parameter or variable values. In conjunction with global studies like Abt Associates (1997), methodological analyses like Markandya (1998), and the European and U.S. empirical studies mentioned above, the developing country studies permit the exploration of certain hypotheses about cross-country and cross-regional differences in the relative magnitude of ancillary benefits.

The paper is organised as follows: Section 2 presents a simple analytical framework relating ancillary benefits to CO_2 abatement costs and pointing to the importance of different regulatory baselines in determining the magnitude of expected benefits. Section 3 takes a comparative look at methodological issues, notably in the context of the *DevCentre* studies and other CGE-based studies for developing countries. Section 4 compares results of various studies, discussing sources of significant variation, while Section 5 concludes with a reflection on what is needed to give policy makers greater confidence in the reliability and robustness of the estimates provided by this body of research.

2. Simple analytics of ancillary benefits

By climate policy we refer to any set of policies whose primary purpose is to slow the growth of net greenhouse gas (GHG) emissions (including through sink enhancement), including those that result in an actual reduction of such emissions relative to some base-year level (as for most Annex 1 Parties to the Kyoto Protocol). (In what follows, we focus on CO_2 abatement, as this is by far the most significant GHG in most countries.) Figure 1.A presents a stylised picture of how costs vary with the abatement level, suggesting that they increase at an increasing rate. In other words, *marginal abatement costs are increasing in abatement effort.* The figure also depicts a stylised ancillary benefits curve, which is shown as a ray from the origin with constant positive slope, suggesting as a first approximation that marginal ancillary benefits are equal to average ones. This follows from the epidemiological studies on mortality and morbidity effects of particulate exposure, many of which find that reductions in risk bear a roughly constant relationship to reductions in ambient concentration irrespective of the initial concentration level. The figure – and the subsequent analysis – abstracts from the primary benefits resulting from climate change mitigation, not because they are not considered important but because they are thought to be too uncertain and distant in time to influence significantly policy making in countries faced with more immediate and pressing concerns. Health of the population is one such concern, and while reducing air pollution exposure may not be the most urgently needed health intervention in countries where infectious diseases are rampant, in many parts of the developing world respiratory diseases are among the leading causes of mortality and morbidity – and air pollution is certainly among the aggravating factors in many cases. Indeed, acute lower respiratory infections rank first of all diseases in the world in terms of disability adjusted life years (DALYs) (WHO 1999), a measure which combines the burden from premature mortality with that from living with disability (Murray and Lopez 1996).

Through inversion of the net cost curve in Figure 1.A, Figure 1.B shows net benefits of CO_2 to be positive over some range, peaking at abatement rate *a* before declining, becoming zero at point *b* (the so-called "no regrets" rate of abatement) before turning steeply negative. An "optimal" climate policy would, needless to say, seek to maximise the net benefits (again bearing in mind the absence from consideration of primary climate benefits), and so "optimal" abatement would be somewhat lower than the "no regrets" rate.

The costs depicted in Figure 1 are those of limiting an economy's emissions of CO_2, which can be done only through one or more of the following: (a) reducing energy consumption; (b) fuel switching from high-carbon to low-carbon fuel; (c) lowering the carbon intensity of a given activity or set of activities; (d) reallocating resources away from energy- (specifically, carbon-) intensive activities. If the economy was operating efficiently in an initial equilibrium, any one of these actions involves an opportunity cost. It is only when one assumes pre-existing inefficiencies – e.g., in energy input per unit of output – that the gross abatement cost curve could be expected to dip below the *x*-axis over an initial range of abatement. In this event, the net cost curve also shifts down proportionally and the "no regrets" level of abatement is further increased.

The ancillary benefits curve is a construct involving several intermediate steps between the policy shock (say, a carbon tax) and the change in real disposable income, our welfare measure. These steps are depicted in Figure 2. The crucial link in the chain is from the carbon tax to the impact on other pollutants. Taking TSP for purposes of illustration, we need to know how a carbon tax – levied for example on the carbon content of fuel – translates into reductions in particulate emissions, in other words, the cross price elasticity of particulates with respect to carbon ($\bullet_{pc}$). The higher is $\bullet_{pc}$, the greater will be the effect on particulate emissions of a given carbon tax. What determines the value of $\bullet_{pc}$? Most importantly, it depends on the extent to which the two pollutants have been "de-linked" in the baseline through prior controls specifically targeted at particulates – in other words, on the stringency, and the strictness of enforcement, of particulate standards. Since growth in carbon emissions is still fairly closely linked to GDP growth (though with some variation in elasticities across countries), de-linking particulates emissions from carbon emissions implies de-linking their growth from GDP growth.

It is generally the case that the OECD countries (i.e., the bulk of Annex 1 countries under the 1997 Kyoto Protocol) have gone farther than developing countries in de-linking local pollution from GDP growth. Another way of putting this is that they have moved farther out along their inverted-U-shaped environmental Kuznets curves for pollutants like particulates and SO_2. This observation suggests a hypothesis about the relationship between a given carbon tax and the size of expected ancillary benefits, viz., that *the lower a country's level of development*, *the larger are the expected ancillary benefits of a carbon tax*. This is because, given the limited prior abatement of local pollution, a tax on carbon translates into a bigger reduction in the more closely linked local pollutant. In short,

$$(\mathrm{GDP})_i < (\mathrm{GDP})_j \Rightarrow (\bullet_{\mathrm{pc}})_i > (\bullet_{\mathrm{pc}})_j \quad \Rightarrow (\mathrm{AB}_i |\, t_c) > (\mathrm{AB}_j |\, t_c)$$

where t_c is the rate of carbon tax and $\mathrm{AB}_{i,j}$ are the ancillary benefits for countries *i* and *j* (measured in physical units – e.g., premature deaths avoided per tonne carbon reduction). Whether this translates into larger monetised welfare gains depends on the relative incomes of the two countries, hence, on their respective willingness to pay (WTP) for the expected health improvements.

Figure 3 presents this analysis in graphical terms, showing the marginal abatement cost curves for local pollution for a low-income (MAC) and a high-income (MAC*) country. The latter is shifted to the left because of the prior abatement of local pollution, so the response to a carbon tax already finds the high-income country on the steeply ascending portion of the MAC* curve. Also in the high-income country, because of the relatively low cross price elasticity of carbon and local pollution, a given carbon tax translates into a lower effective tax on the latter – t_e* *versus* t_e. The combination of these two effects implies a lower *post*-tax equilibrium level of local pollution abatement, hence, lower ancillary benefits in the high-income country than in the low-income one.

Figure 1A. **Gross and net costs of CO_2 abatement**

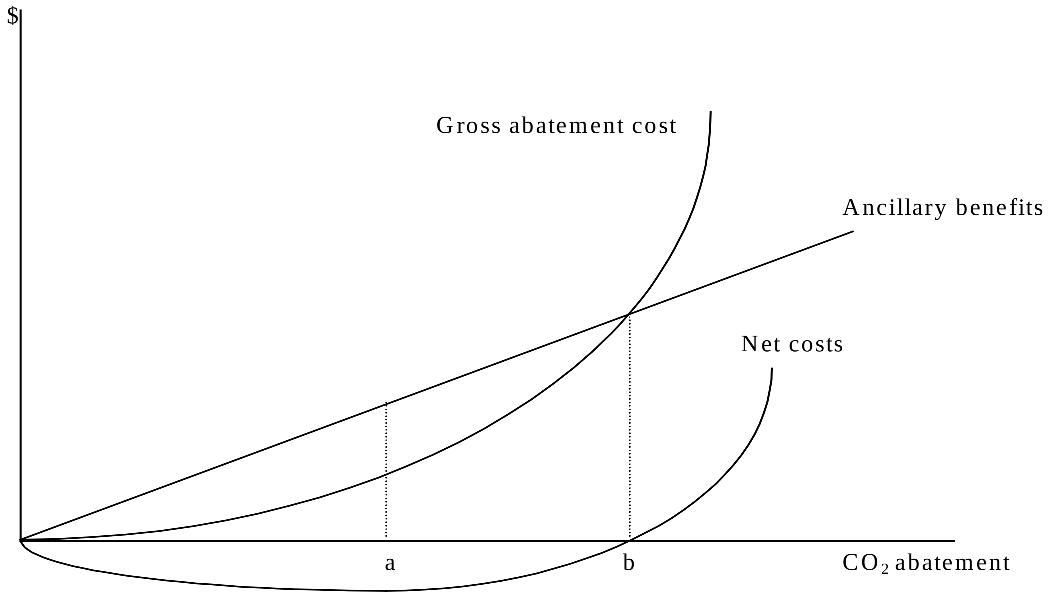


Figure 1B. **"Optimal" and "No Regrets" CO_2 Abatement**

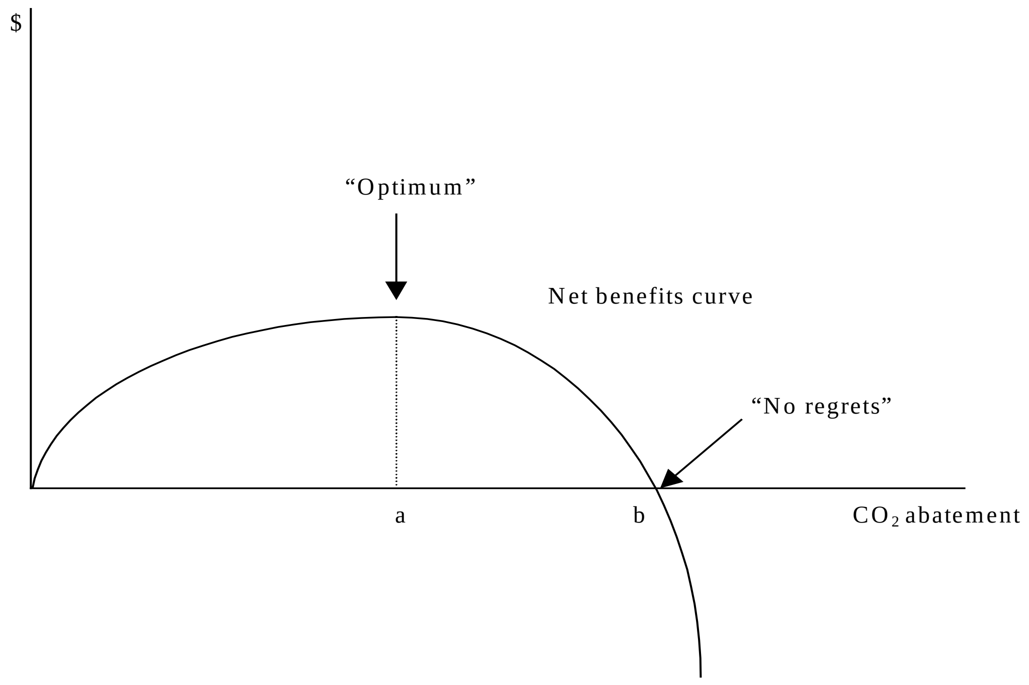


Figure 2. **Links in chain from policy measure to welfare change**

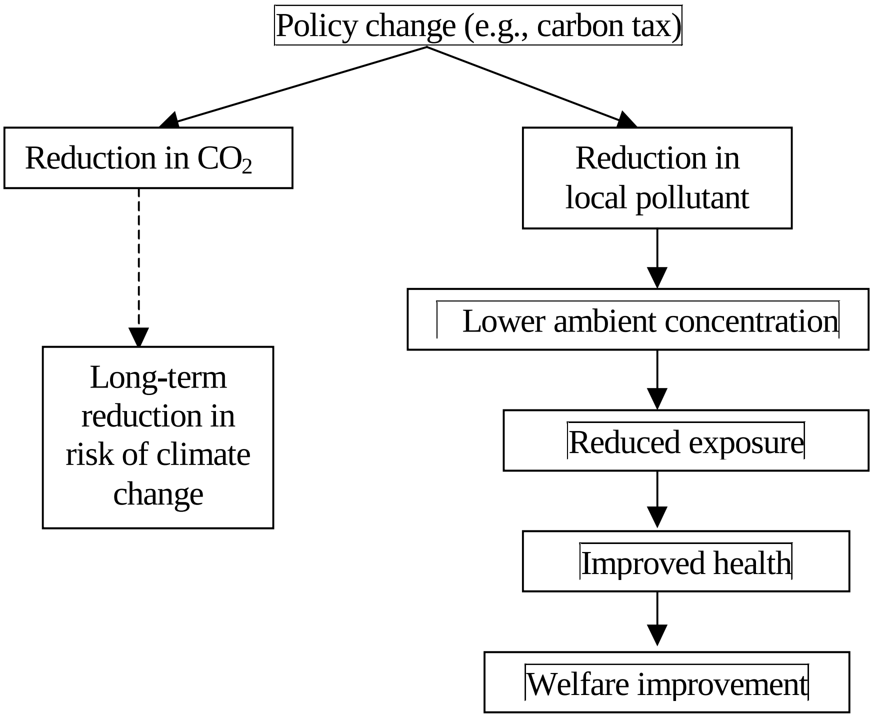
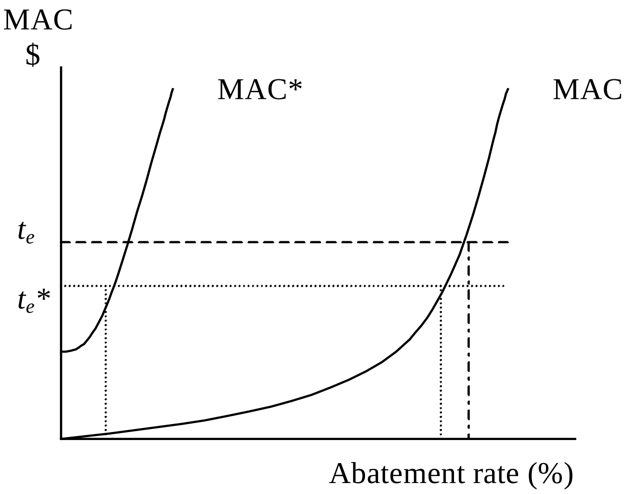


Figure 3. **Marginal abatement costs and abatement rates for local pollutants, developed and developing countries**



3. Methodological issues

Put simply, ancillary benefits analyses – like climate policy analyses more generally – can be dichotomised into top-down and bottom-up approaches. The *DevCentre* studies make use of the former, so that is the principal focus of discussion here, though at points reference is made to bottom-up methods for purposes of comparison. In any event, several of the methodological issues discussed relate equally to both types of approach.

The top-down approaches mostly make use of CGE models to look at economy-wide impacts of a given policy scenario relative to a no-policy baseline. These are the models of choice for most global climate policy modelling, where broad orders of magnitude of welfare change and rough comparisons across regions of the world are the most that is sought. Their principal virtue lies in their ability to capture feedbacks in the economic system, e.g., via relative price changes, that might lead to results other than those predicted from an examination of first-order, partial equilibrium effects alone. Their principal drawback is the paucity of technological detail, which makes them less than ideally suited for a thorough micro-level assessment of sectoral responses to a policy shock. Such models can, of course, be made more realistic, but at a cost in added model complexity. Another criticism levelled against CGE models is that they are not always strongly grounded in empirics – e.g., econometric estimation of the thousands of elasticities embedded in a typical model structure would simply be too data intensive, but on the other hand, elasticity values cannot simply be pulled out of thin air.

In the context of ancillary benefits estimation, one of the features of a typical CGE model is particularly noteworthy. It seldom incorporates a separate abatement technology for local pollutants. This implies that the only way to control those pollutants in the model is via inter-fuel substitution (e.g., switching from coal to gas in power generation) or via substitution of productive factors (e.g., labour) and/or other inputs for polluting energy in a given production process. At the level of the economy as a whole, structural change towards less polluting sectors can achieve the same results.

To the extent that end-of-stack or end-of-pipe abatement has already occurred, it is reflected in a reduced level of base-year emissions of local pollutants. In model simulations, however, further adoption of such technology cannot be readily accommodated. In reality, a carbon tax would most likely work its effects via fuel, factor and input substitution, so this is not a serious limitation in simulating climate policy. What is rendered difficult is any comparison of marginal costs of end-of-pipe/stack abatement with those of abatement via fuel/factor/input substitution. It is possible, for example, that in a country where no prior controls on particulate emissions are in place, there are capture technologies that would reduce emissions up to a point at significantly lower cost than would be possible incurred with a carbon tax. In that event, and referring back to Figure 3, model simulations would not be able to reflect movements along the shallow portion of the developing country's MAC curve but only along steeper sloping portions. In effect, MAC would be shifted leftward towards MAC*.

Also – and it is here that the degree of technological detail matters – how well one can capture substitution possibilities – e.g., between fuels – depends on how disaggregated a model one has of the energy sector, hence, of fuel types. For instance, while most models distinguish coal and oil as separate sectors, few can distinguish low-sulphur diesel oil from high-sulphur diesel or low-ash coal from high-ash coal. From the perspective of carbon emissions, such distinctions are not particularly important, but they are when one is concerned with impacts of policy on local pollutants.

3.1 Modelling emissions and dispersion

Another difficulty with the use of CGE models for ancillary benefits analysis results from a lack of spatial detail. For global climate modelling, only the largest countries are treated individually, with the rest of the world grouped into broad regions. Ancillary benefits estimation for an individual country obviously requires a separate national CGE model, but even this is not necessarily adequate to capture the local-level dynamics that affect the size of ancillary benefits. Carbon dioxide is a global pollutant, so a single, undifferentiated national model suffices for analysing climate policy on its own, but the pollutants of interest for ancillary benefits estimation are mostly local or regional. Geographic location of emissions, stack heights of emitting sources, local temperature and meteorological conditions, population distribution and location of valuable assets vulnerable to pollution damage all matter to the nature and size of impacts[1]. Needless to say, this richness of detail is not well captured in a national model, especially for a large country. It is for this reason that, in the case of the *DevCentre* studies, it was decided that, while for Chile a single national CGE model would suffice (given that most air pollution impacts are concentrated in the heavily populated capital city of Santiago), for India a multi-region CGE model would be more appropriate.

In any event, it is seldom possible with a CGE model to achieve a degree of disaggregation ideal for analysing local air quality and health impacts, viz., at the level of the individual metropolis. Wedding CGE models to adequate air dispersion models remains a research challenge, not least because even a regional CGE does not usually incorporate a detailed locational grid of emissions within the region of the sort needed for more sophisticated air modelling. To illustrate the problem, suppose that, while coal-burning power plants account for 50 per cent of regional particulate emissions and motor vehicles 20 per cent, the latter contribute 60 per cent to ambient concentrations in the main regional metropolis, while the former contribute only 30 per cent. Ideally, this locational effect on the emissions-concentration relationship should be reflected in the basic dispersion model, but without the benefit of a source-receptor matrix, one might mistakenly conclude that a 10 per cent reduction in power plant emissions would reduce concentrations and exposure in the big city by 5 per cent.

Geographically localised ancillary benefits studies are able to incorporate more sophisticated dispersion models – e.g., of the Gaussian plume variety (see Colls 1997, ch.3, for a presentation of the Gaussian model with worked examples). This approach, which is rather data-intensive, is adopted in Cifuentes *et al.*(1999) for Santiago, Chile.

[1] To illustrate the role of temperature and humidity, according to one estimate, concentrations of particulate matter from a fixed quantity of emissions in hot and dry regions are about one-third of what would be expected from the same emissions under most other climatic conditions (Working Group 1997).

3.2 Modelling concentrations and exposure

In the absence of detailed location-specific emissions data and a means of mapping these into ambient concentrations at different receptor points, some simplifying assumptions are needed to link emissions changes generated by policy simulations to changes in concentrations-exposures. In the extreme, for some pollutants (excluding those that are the product of atmospheric chemical reactions – e.g., O_3, sulphate and nitrate aerosols) one can assume a simple linear relationship between regional emissions and ambient concentration measures in major cities, making use of base-year emissions and concentration data to determine the coefficient on emissions, and assuming some background level of emissions unattributable to specific sources. If information is available on the proportion of emissions-generating economic activities located in each metropolis, the regional emissions figures can be scaled down to approximate more closely local emissions. This is essentially the approach adopted in the Chile study of Dessus and O'Connor (1999). A slightly more sophisticated approach, used by Garbaccio *et al.* (2000) for China is to assign different coefficients to different sectoral groupings, depending on whether emissions from those sectors normally occur at or near ground level (e.g., motor vehicles and small boilers), from stacks of medium height (large industry), or from high stacks (power plants) (classification based on Lvovsky and Hughes 1998). In short, the dispersion function is of the form:

$$Conc_{TSP} = a + b_1 (Emis_{Tall}) + b_2 (Emis_{Medium}) + b_3 (Emis_{Low}),$$

where $Conc_{TSP}$ refers to the average city-wide concentration of TSP, $Emis_{Tall,\ Medium,\ Low}$ the region-wide or metropolitan-area-wide TSP emissions from each of three groups of sectors differentiated by typical stack height. The constant *a* is an approximation of the effect of background emissions on ambient air quality (in short, what concentration would obtain assuming zero sectoral emissions). The b_is are the dispersion coefficients for emissions from each stack height, calculated using a simple dispersion model in which different atmospheric conditions are assumed to occur with given frequencies[2] and the key piece of additional data required is a metropolitan area's radius (see Lvovsky and Hughes 1998). The use of even this somewhat more sophisticated dispersion model still involves a gross simplifying assumption, viz., that the specific geographic distribution of emission sources within the area does not significantly affect area-average pollutant concentration.

Even if location-specific emissions data are not available, it is clearly necessary to have an estimate of total emissions of a given pollutant and to be able to allocate those emissions by sector. Fuel-specific emission factors are a useful starting point. With sector-wise data on fuel consumption by type it should be possible to estimate maximum combustion-related sectoral emissions. To these one needs to add any process emissions not directly linked to fuel use – e.g., fugitive dust from cement plants. This total then needs to be adjusted downward by a factor reflecting removal with end-of-pipe/stack controls. Also, it is probably a reasonable assumption that, as enterprises invest in new plant and equipment, average emissions (per unit of fuel unit or per unit of output) will fall. Thus, in the case of particulates, and drawing on work by Lvovsky and Hughes (1997), Garbaccio *et al.* (2000) incorporate lower emission coefficients for new capital stock than for old, though no account is taken of the cost of engineering lower emissions into new capital equipment and emission coefficient reductions are treated as additional to the autonomous energy efficiency improvement (AEEI) factor.

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[2] Ideally, region- or city-specific information on atmospheric conditions can be found to determine these frequencies, but if not then certain "default" frequencies can be used as an approximation.

3.3 Assessing health damages

To assess actual health damages, ideally one would like to be able to measure effective human exposure – i.e., numbers of people exposed to what concentrations over what period. In practice, the epidemiological studies of pollution's health impacts mostly relate variations in ambient concentration to variations in relative risk, e.g., the risk of premature death. In short, if pollution (say, PM_{10}) were reduced by a given amount from current levels, how many lives would be saved or, conversely, how many excess deaths would be caused by a given PM_{10} increase?

The results from multi-city U.S. studies of acute exposure to PM_{10} by Dockery, Pope and colleagues are quite consistent, finding an estimated 0.7-1.5 per cent increase in total mortality associated with a 10 •g/m^3 increase in PM_{10} concentration from mean levels in the range 38-61 •g/m^3 – i.e., several times lower than mean concentrations in many developing-country cities. A meta-analysis in Schwartz (1994) finds a consensus range for mortality increase estimates of between 0.7 and 1.0 per cent per 10 µg/m^3 increase in PM_{10} concentration. Comparing their estimates to those of other studies, Dockery *et al.* (1992) observe that the dose-response relationship between particulates and mortality is remarkably similar across a large range of concentrations, in a variety of communities, and with varying mixtures of pollutants and climatology. There is no evidence of a "no effects", or threshold, concentration – at least not within the range observed in U.S. cities.

The robustness of the estimates is borne out by non-U.S. studies, including a handful in developing countries. For instance, Ostro *et al.* (1996) find a significant relationship, for Santiago, Chile, between ambient particulate concentration (in this case, PM_{10}) and mortality, after controlling for confounding influences like temperature. In particular, the results from their basic OLS model suggest that a 10 µg/m^3 change in concentration around the mean (115 µg/m^3) is associated with a 0.6 per cent change in mortality[3]. They note that their results are consistent with findings of various U.S. studies on the PM_{10} – mortality link and suggest that, for this reason, applying the U.S. estimates to developing countries may be appropriate where local research is not possible, assuming those countries are not drastically different from the United States in terms of variables like time spent outdoors, baseline health status, and medical care and access.

An aspect of the particulates–mortality relationship that can be important for impact valuation is the age distribution of those whose lives are foreshortened. In the U.S. studies, those at highest risk are the aged and infirm and also the very young. In India, by contrast, Cropper *et al.* (1997) find that, while the overall mortality risk is somewhat lower in Delhi than in the U.S. studies, the 15-44 age group are at greater risk than those over 65 years. For one thing, the proportion of the population in the latter age group is much lower than in the United States; for another, most of the deaths before that age are from causes unrelated to air pollution. The age profile of those at risk is clearly more pertinent when the measure to be valued is life-years-lost or DALYs than when it is premature deaths averted irrespective of remaining life expectancy.

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3 The relationship between PM_{10} and mortality is non-linear, however, with a change evaluated at 50 µg/m^3 associated with a 1.4 per cent increase in mortality (i.e., closer to the U.S. means) and one evaluated at 150 µg/m^3 increasing mortality by 0.4 per cent.

Besides mortality, there are a variety of morbidity endpoints that may be affected by air pollution, though few relationships are borne out as consistently by the epidemiological literature as the PM_{10} - mortality link. There are two main ways in which the effects of pollution on morbidity are measured: as incidence of physical symptoms and illness and as behavioural responses to the symptoms/illness. The former are normally the object of interest in clinical studies, while epidemiological studies may report on symptom/disease incidence and/or effects on human activity. The most common measures for the latter are "restricted activity days" (RADs), "work loss days" (WLDs), hospital admissions, and emergency room visits. RADs are a more comprehensive measure than WLDs, including days spent in bed, days missed from work, and other days when normal activities are restricted due to illness (Cifuentes and Lave 1993). They are also a more subjective measure and thus subject to greater measurement error.

Reviewing briefly the epidemiological evidence (and drawing principally on Ostro, 1994), air pollution is most commonly associated with respiratory illnesses, though other illnesses linked to specific pollutants include cardiovascular illness and impaired neurophysiological development (in the case of blood lead in children). More specifically, particulate exposure has been found to be associated with lower-respiratory illness in children; particulate and ozone levels with exacerbation of asthma attacks among both children and adults; ozone in particular with eye irritation and respiratory symptoms; and long-term exposure to particulates and sulphate and nitrate aerosols with chronic bronchitis and reduced lung function.

3.4 Valuing impacts

There are three broad approaches to valuation of environmental benefits in general and ancillary benefits of climate policies in particular (see Freeman, 1993, for the classic text on valuation methods). The first approach tallies productivity losses or costs to the economy from illness, premature death, or damage to crops, materials and ecosystems. From a theoretical standpoint it is the least satisfactory, not being firmly grounded in welfare economics, i.e., on measurement of changes in individual welfare. It can, however, provide a lower bound estimate of "true" benefits. The other two approaches – the first based on revealed preferences and the second on stated preferences – avoid this problem. Of the two, more controversy surrounds the latter, since it is based not on observed but on hypothetical behaviour. This is not to suggest that revealed preference methods are problem-free; they are not (see following discussion).

In most valuation exercises, mortality benefits/costs tend to dominate morbidity benefits/costs, the main reason being the high value attached by most people to mortality risk reductions. These are captured by a concept known as the "value of a statistical life", reflecting the willingness to pay of individuals for a given reduction in *ex ante* risk of premature death.

The literature purporting to estimate *VSL* is vast and still growing (see Viscusi 1993 for an earlier review). The overwhelming majority of the studies have been done in Europe and the United States. The bulk employ a revealed preference method called hedonics to estimate the compensating wage differential paid to those workers in jobs with relatively high fatality rates. From this one can derive an estimate of *VSL*. For instance, if it is found that, on average, a worker receives a wage differential of \$350 per year for assuming an added risk of accidental death on the job of 1/10,000, then this implies a *VSL* of \$3.5 million. When one transfers this estimate out of the context in which it was derived – e.g., to one of mortality risk from pollution – there are at least three possible sources of bias, two having to do with different risk characteristics and the third with different affected populations. First, assuming complete information, job-related risk is voluntarily assumed, while risk from pollution exposure is involuntary, in the nature of a negative externality imposed by others' behaviour (or a combination of own and others' behaviour). Second, the time dimension of the risks can differ. For example, certain risks from pollution exposure are delayed until later in life, and people may value differently risks avoided now to those avoided later. Trying to capture the notion of delayed risk in a contingent valuation (CV) questionnaire, Krupnick *et al.* (1999) find from pre-test results that the discounted *VSL* is significantly below estimates from hedonic wage studies.

Third and finally, reducing mortality risk from pollution may be valued differently by individuals according to their ages. The population sampled in hedonic wage studies consists of active workers, while those most adversely affected by air pollution (at least in the United States and other OECD countries) are beyond working age. The direction of any resultant bias is unclear, however. On the one hand, one might expect the elderly to be relatively risk-averse, while on the other the willingness to pay (WTP) to save relatively few extra years of life may be lower than WTP to save an average of 30 or more expected by active workers. One empirical study does find that the WTP of the elderly to reduce mortality risk is somewhat lower than that of younger persons (Jones-Lee *et al.*, 1985). Since, as noted above, Cropper *et al.* (1997) find for the Delhi population that those at greatest risk from particulate air pollution fall into the prime working-age group (15-44 years), using hedonic wage estimates of *VSL* may not be a significant source of age-bias, though the other sorts of bias mentioned above could still be present.

A separate issue crucial for ancillary benefits estimation in developing countries is the relationship between WTP for reduced mortality risk (or *VSL*) and *per capita* income. Naturally, the former can be expected to rise with the latter, but is the rise proportional? The reason it is important is that, for most developing countries, there are few if any on-site *VSL* studies comparable to the hedonic wage studies done for the United States. It is common practice, therefore, to borrow *VSL* estimates from the U.S. studies, scaling them for differences in *per capita* income between the United States and the target study site. What scaling factor should we use? If the ratio of *per capita* incomes is 10:1, and if U.S. *VSL* is \$3.5 million, should we assume a *VSL* in country *x* of \$350,000. Doing so implicitly assumes that the elasticity of *VSL* with respect to income is unity. This does not, however, square very well with the evidence. Rather, it would appear that *VSL* rises less than proportionately to income; in other words, its income elasticity is less than unity. In their benefits transfer study of air pollution in Central and Eastern Europe, Krupnick *et al.* (1996) assume an elasticity of 0.35 (based on contingent valuation studies reported in Mitchell and Carson 1986). Also, in their study of mortality risk valuation in India, Simon *et al.* (1999) find evidence that the *VSL* is higher relative to *per capita* income than for the United States, dismissing as implausible the possibility that Indian workers are more risk averse than their American counterparts. Also, studies of morbidity risk find a relatively low income elasticity of WTP to avoid illness, ranging from 0.26 to 0.60 (Loehman and De 1982 and Alberini *et al.* 1997).

In sum, when applying benefits transfer for either mortality or morbidity risk reductions, it seems reasonable to assume an income elasticity of WTP well below unity, indeed, probably closer to 0.5. Thus, the *VSL* in a low-income country will be lower than in a high-income one, but by less than the ratio of their *per capita* incomes would imply.

While strictly speaking, one should derive measures of WTP for both mortality and morbidity risk reductions from welfare-theoretic principles, in practice there are seldom subjective measures of WTP for all relevant health endpoints. In their absence, it may be necessary to rely on such observables as "cost of illness". In any case, once mortality benefits and morbidity benefits of a policy change have been calculated, they need to be incorporated back into the economy-wide model to determine the resulting welfare change. In practice, this is done by calculating what reduction in disposable income would leave individuals indifferent between the *status quo* and a post-policy state with reduced air pollution, fewer premature deaths, and improved health. That change in disposal income represents the amount of the welfare gain from cleaner air, and it can be compared in turn to the costs of achieving that improvement, measured by the reduction in disposable income associated with a carbon tax or other policy (and abstracting from any ancillary benefits).

4. How far do results differ across studies and why?

The literature on ancillary benefits of climate policy dates at least to the early 1990s (cf. Ayres and Walter 1991). Ekins (1995, 1996) reviews a number of ancillary benefits studies, with an emphasis on Europe. While there are relatively few, they yield widely varying estimates. In part, the variance stems from different underlying estimates – e.g., of *VSL* – but there are additional sources: differences in the method of estimating benefits (e.g., damages avoided *versus* abatement costs avoided); differences in scope of benefits included (e.g., some studies include both emission-related benefits and non-emission-related ones like the reduction in traffic congestion, accidents, and noise resulting from reduced road transport); differences across study sites in population exposure to pollution (with generally higher population densities in Europe than in the United States and prevailing winds blowing pollution inland in Europe, but out to sea from the eastern United States); the assumed stringency of CO_2 control; the timeframe of the scenario and the date of measurement of benefits. Another potential source of variation are differences in definitions of local pollutant baselines, deemed to be important in U.S. studies reviewed by Burtraw and Toman (1997; see below). Considering only emission-related benefits, the values range (in 1990 $US) from a low of $20/tC (Barker, 1993, for the U.K., based on social preferences revealed from the marginal costs of implementation of existing abatement technologies) to a high of $212/tC (Alfsen *et al.*1992, for Norway; Pearce 1992 reports a similar figure for the U.K.: $195/tC). A mean value of emission-related benefits, based on the estimates reported in Ekins (1995), is around $100/tC.

With the exception of Ayres and Walter (1991), the other studies reviewed in Ekins (1995) were conducted in Europe. Ayres and Walter find ancillary benefits for the United States of around $23/tC (at 1990 prices) from fossil fuel emissions reductions in two sectors[4] (transport and electricity, which together account for about two-thirds of carbon emissions). While Ekins discounts this study because of its partial coverage, a more recent review for the U.S.A. by Burtraw and Toman (1997) reports on the results of eight studies whose mean estimate of ancillary benefits is virtually identical to the Ayres and Walter figure (i.e., $24/tC), with a low estimate of $2.64 and a high of $78.85. Burtraw and Toman state a preference – on methodological grounds – for estimates at the lower end of the range (i.e., below $7/tC) (though these cover only the electricity sector; the two economy-wide model-based estimates are above the mean but neither yields the highest estimate). Those studies producing the lower estimates also assume weaker control measures and smaller carbon reductions. Estimated abatement costs per tonne carbon reported in these studies are in the $10-20/tC range, so with ancillary benefits in the $3-$7/tC range (and with a similar differential between ancillary benefits and control costs found in studies assuming somewhat more vigorous abatement), Burtraw and Toman propose a "rule-of-thumb" (for the United States) that ancillary benefits can be assumed to be roughly 30% of the cost of carbon reduction for low to moderate rates of abatement. Observing that over some range the marginal costs of GHG reductions are likely to be close to zero, the authors conclude that the existence of ancillary benefits even as small as $3/tC could significantly increase the volume of emissions reduction that is considered "no regrets" in the sense of having negative or zero net cost.

Starting with the less optimistic assumption (from Nordhaus 1991) that even at low abatement levels costs are positive, Ekins (1995) calculates marginal abatement costs for small increments in the abatement rate, using an equation estimated by Nordhaus in a regression of the marginal abatement costs from several studies on their respective abatement rates. In so doing, he finds that even the lowest estimate of ancillary benefits on which he reports ($23/tC) would justify a CO_2 emissions reduction of nearly 15%, supporting the Burtraw and Toman conclusion.

There does not yet exist the same wealth of studies on developing countries, though a few have been completed and several more are in progress (the two major endeavours being spearheaded, respectively, by OECD Development Centre and the U.S. Environmental Protection Agency through its *ICAP*[5] initiative). Thus far, each has produced an ancillary benefits study for Chile (Dessus and O'Connor, 1999, and Cifuentes *et al.*, 1999). The former is presently undertaking a study of India (Bussolo and O'Connor 2000 forthcoming) and will soon initiate one for China, while the latter has several in progress – for Argentina, Mexico, China and Korea. Garbaccio *et al.* (2000) have produced preliminary results for China and Joh (1999) for Korea. A preliminary comparison of results of these studies – with each other and with the OECD-country studies – suggests that there is also a fairly wide range in value estimates of ancillary benefits, even for the same country (e.g., Chile), but that estimates of physical benefits (e.g., numbers of deaths averted) are less variable.

4 The benefits estimates are based on the assumption of a 20% reduction in air pollution from 1978 levels.

5 International Co-Control Analysis Program, initiated in 1998, benefits from financial and technical support provided by The National Renewable Energy Laboratory (NREL) and the World Resources Institute (WRI), as well as other contractors such as Abt Associates, all of the United States.

The latter is understandable in view of the fact that most ancillary benefits studies draw upon the same set of dose-response functions gleaned from the epidemiological literature reviewed briefly above. Thus, their assumptions about the effects on mortality or morbidity of a given reduction in ambient concentration of local pollutant *x* are very similar. For ancillary benefits analysis, as described in the preceding section, it is necessary to work back from changes in concentration (say, of particulates) to changes in particulate emissions to changes in carbon emissions. Only then can we calculate the ancillary benefits (in physical terms) per tC reduction. Table 1 reports on calculations of this measure based on several ancillary benefits studies. The first two are for Chile, and they show a number of lives saved (or premature deaths averted) ranging from 89 to 100 per MtC reduction. Bearing in mind the discussion of Figure 3 above, it is interesting to compare the results for Chile with those for the United States (with a significantly higher *per capita* income and stricter baseline environmental standards) and for China (with lower income and more lenient standards). The ordering of the size of mortality benefits – China highest, U.S.A. lowest, and Chile in between – is consistent with our hypothesis that the ancillary benefits (measured in physical units) of climate policy are likely to be higher the lower a country's baseline air quality.

Translating physical benefits into monetary values requires, in the case of mortality benefits, an estimated *VSL*. Since this is closely related to *per capita* income, it is naturally lower in poorer countries than in richer ones. Thus, if physical benefits are larger in the former than the latter, but *VSL*s are lower, there can be no *a priori* expectation about whether the monetary value of ancillary benefits per tC reduction will be higher or lower. If differences in physical benefits are sufficiently large, or differences in *per capita* income sufficiently small, the larger physical benefits in the poorer country could translate into larger monetary benefits than in the richer one (or at least comparable ones). The *per capita* incomes are likely to be closer in terms of purchasing power parity, or *PPP*, than in terms of market exchanger rate conversion (in some cases significantly so). Thus, the choice of conversion rate to a common currency can make an important difference to the results. In our view, *PPP* is the more appropriate rate since the trade-off of mortality risk against other items in an individual's utility function depends on real disposable income, which in turn depends on *real* purchasing power of goods and services.

Table 2 presents three different sets of *VSL* estimates for Chile, with the underlying assumptions. It illustrates well the importance of two choices: of the "best estimate" of *VSL* from U.S. studies for purposes of benefits transfer, and of the exchange rate to be used in converting *VSL*s to a common currency. Dessus and O'Connor (1999) employ a moderate *VSL* estimate from U.S. studies, while Markandya's (1998) figure of $4.8 million is at the high end of the range and Cifuentes *et al.* use a conservative estimate ($1.9 million). Dessus and O'Connor (1999) and Markandya (1998) both employ a *PPP* exchange rate, while Cifuentes *et al.* (1999) employ the 1995 market exchange rate. The combination of the high U.S. *VSL* and the *PPP* exchange rate make the Markandya estimate for *VSL* in Chile (1995) more than three times higher than Cifuentes' estimate. In a pairwise Dessus/O'Connor–Cifuentes *et al.* comparison, the discrepancy in 2010 *VSL* estimates for Chile arises essentially from one source: the different exchange rate, or conversion factor, used, with Cifuentes *et al.* using a ratio of U.S. to Chilean *per capita* income (1995) of 5:1, based on the market exchange rate, and Dessus and O'Connor using a ratio of 2.5:1 based on a *PPP* exchange rate taken from the World Bank's 1999 World Development Indicators (CD-ROM). In the 2000 version of WDI, Chile's 1995 *PPP per capita* income has been revised downward substantially, so the new ratio of U.S. to Chilean *PPP* income is 3.8, i.e., much closer to the ratio used by Cifuentes *et al*[6].

[6] Moral: if you want to do benefits transfer right, do your own estimation of *PPP* GNP!

Ideally, then, one would not need to rely on benefits transfer for estimating the *VSL* for use in a particular ancillary benefits study. As we have just seen, this may pose a problem if there is some uncertainty about estimated *PPP per capita* income. In addition, there remains some uncertainty about the income elasticity of *VSL* to apply in making the transfer. In Table 2, the assumed elasticity is unity, while as discussed above there may be reason to suppose it is much lower. Cifuentes *et al.* are apparently attempting their own contingent valuation survey to estimate WTP for reduced mortality and morbidity risks in Chile. Likewise, the India ancillary benefits study currently in progress (Bussolo and O'Connor 2000, forthcoming) can avail of the results of the Simon *et al.* (1999) hedonic wage study for Delhi which, as noted above, yields a *VSL* strongly at odds with a unitary income elasticity assumption.

In summary, a first best strategy for ancillary benefits valuation is to rely on original WTP estimates for the study site. Benefits transfer is distinctly second-best and fraught with potentially significant biases.

Table 1. **Comparison of mortality benefits estimates of CO_2 reductions**

Study	*Lives saved per MtC reduction*	*Scenario Assumptions*
Cifuentes *et al.* (1999)	89	Chile, 2020: 13% CO_2 reduction
Dessus and O'Connor (1999)	100	Chile, 2010: 10% CO_2 reduction
Garbaccio *et al.* (2000)	430	Chile, 2010: 15% CO_2 reduction
Abt Associates (1997)	82	USA, 2010: 15% CO_2 reduction

Table 2. **Comparisons of value of a statistical life (*VSL*) for Chile**

Dessus/O'Connor	*Markandya*	*Cifuentes et al.*
$2.1 million (2010)	$1.37 million (1995); $2.1 million (2010)	$ 375,000 (1995); $ 780,000 (2010)
1992 PPP $	1994 PPP $	1995 $ at market exchange rate
Ratio to 1995 U.S. VSL ($ 2.6 million) = 0.79	Ratio to U.S. VSL ($ 4 million) = 0.34; 0.53	Ratio to U.S. VSL ($ 1.9 million) = 0.2; 0.41

Source: Dessus and O'Connor (1999), Markandya (1998), Cifuentes *et al.* (1999).
Notes: World Bank estimate of 1995 ratio (Chile/U.S.A.) of *per capita* GNP = 0.4.
Assumed elasticity of *VSL* with respect to income (GNP) = 1 in all cases.

5. How to enhance policy relevance

The preceding discussion leaves some doubt about how useful the results of ancillary benefits estimation may be to policy makers, given the wide variation observed. A few general observations are offered here on the issue of credibility and how to enhance it (see also Davis, Krupnick, and Thurston 2000 for a discussion of the credibility of ancillary health benefit and cost estimates). Pearce (2000) notes that, in OECD countries, the ancillary benefits studies conducted so far have been largely academic and poorly integrated at best into climate policy making. The same could be said, but perhaps even moreso, in developing countries.

To be credible, any study of ancillary benefits must make its assumptions, methodology and scope as transparent as possible. The most likely criticism of results is that the benefits of climate policy are exaggerated by unrealistically optimistic assumptions, by neglect of certain costs, by an inappropriate baseline definition, or by incorrect estimation procedures. With respect to the baseline, it is perhaps best to give government the benefit of the doubt, i.e., assume that it will be successful in implementing any significant local environmental regulations already on the books or soon to be introduced. A useful method of dealing with parameter uncertainty is to conduct sensitivity analysis, varying parameter estimates over some range. To the extent that there are estimates from multiple studies, one can perhaps assume a sampling distribution to obtain a confidence interval around the mean and calculate benefits using parameter values marking the extremes. Clearly, the greater the variance in parameter estimates across studies, the wider the confidence interval will be.

In presenting results of an ancillary-benefits study, it is helpful to bear in mind the policy use to which it is to be put. The principal utility of such a study is to provide order-of-magnitude estimates of how large the expected local health (and/or other) benefits are from a given climate policy. This information may be valuable in and of itself and, in addition, it may be useful to be able to compare the value of ancillary benefits with the costs of GHG abatement (also on occasion with the primary benefits of climate change mitigation.) While for cost-benefit comparisons valuation of ancillary benefits is essential, in general, it is strongly recommended to report the physical impact estimates as well (premature deaths avoided, DALYs, symptom-days avoided, etc.), since these are apt to be somewhat less uncertain than the monetary values.

Where ancillary benefits estimation is done in a developing country, it may not be possible to find local epidemiological studies and/or WTP studies on which to base the analysis. In this case, one needs to proceed with extreme caution in undertaking benefits transfer. Transferring parameter estimates from epidemiological studies done elsewhere seems to be warranted (at least in the case of particulates), but even here there is the possibility that the age distributions of those affected may differ between original study site and new study site (recall the case of particulate-related mortality in Delhi *versus* U.S. cities). Potential problems with benefits transfer are more serious with respect to *VSL* and WTP for morbidity risk reductions. They include, to recap: (i) obtaining reliable estimates of *PPP per capita* income; (ii) choosing a suitable income elasticity of *VSL* for converting the chosen foreign *VSL* to a local one; (iii) ensuring reasonable comparability of populations and of risks between the original *VSL* study context and the pollution context of the ancillary benefits study. Ultimately, if resources permit, there is no good substitute for generating original WTP estimates at the new study site. If not, one may may want to consider, in addition to *VSL* transfer from elsewhere, calculating from local earnings data the foregone earnings from premature mortality, as a lower bound on mortality benefits (bearing in mind the well-known limitations of this “human capital” measure).

To provide a rough check on the plausibility of estimates obtained, it is useful to compare results with those of other studies in terms of a common metric. This applies both to the underlying *VSL* and WTP estimates employed in the study and to the final value of ancillary benefits. In the first case, a common practice is to compare *VSL* to *per capita* income of the country; another is to compare it to the present discounted value of foregone earnings as a result of premature death (in effect, comparing the age-adjusted *VSL* from hedonic wage or contingent valuation studies – or benefits transfer based on such studies – with the human capital measure). Where comparable ratios are available for several countries, this provides a broad consistency check, though an interpretation is not easy in the event that the ratios show no tendency to converge.

In the case of ancillary benefits *per se*, the most common metric is the value of benefits per tC reduction. As previously noted, this measure tends to vary fairly widely across regions, though within regions – e.g., among the U.S. studies alone – the variation is less extreme. There can be significant factors explaining inter-regional differences (population density, climatology, *per capita* incomes, etc.), but wide divergence of results within a given region or country is apt to be more problematic, requiring careful diagnosis of the source of discrepancy.

Finally, an important element of ancillary benefits analysis in developing countries should be the consideration of the relative merits (in particular relative costs) of climate policy *versus* more direct measures to control local pollution. As explained above, this is especially pertinent in the developing-country context because one cannot necessarily assume that low-cost abatement options, e.g., for particulates, have already been exhausted. In short, one would like to be reasonably sure, when pointing to the ancillary benefits of climate policy, that there are not significantly lower-cost options for a developing country to realise those same benefits.

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