PRELIMINARY ESTIMATION OF THE POTENTIAL ANCILLARY BENEFITS FOR CHILE

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1. Introduction

There is no doubt than human activity is responsible of the increasing atmospheric concentrations of greenhouse gases (GHG), including carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O). The main activities responsible of this increase are fossil fuel combustion, which has grown at a rate unprecedented in human history, and changes in land use and agricultural practices. In the absence of emission controls for GHG, their atmospheric concentrations will rise in the next decades to levels that may induce changes in the climate of the earth. The Intergovernmental Panel on Climate Change (IPCC) estimates that human-induced climate change will increase surface temperatures by about 2°C by the year 2100 (Houghton, Meiro Filho *et al.*. 1996), although many uncertainties exist about this estimate.

The climate change protocol signed in the Third Conference of the Parties in Kyoto in December 1997 set goals for emissions reduction for countries included in Annex I, which includes only developed countries. Non Annex I countries, mainly developing countries, do not need to abide to any emission reductions. The protocol set up an emissions trading framework that would allow countries (mainly Annex I) to invest in GHG reduction projects in other countries, and share part of the emissions credits. The implementation of such schemes, like “Joint Implementation” and “Clean Development Mechanisms” have been widely discussed at the subsequent Conference of the Parties held in Buenos Aires and Bonn.

In order to stabilize the global concentrations of GHG, it will be necessary for all countries, including developing countries, to make reductions in their emissions. However, developing countries shall make the most progress in reducing the growth of their greenhouse gas emissions by implementing measures that are consistent with their development objectives and that provide near term economic and environmental benefits. Within the existing framework, it is not clear for a developing country if it is beneficial to enter voluntarily in an emission reduction scheme. Our own analysis for Chile (Montero, Cifuentes *et al.*. 2000) shows non-conclusive results, with the economic convenience depending heavily upon the initial emissions baseline assigned to the country.

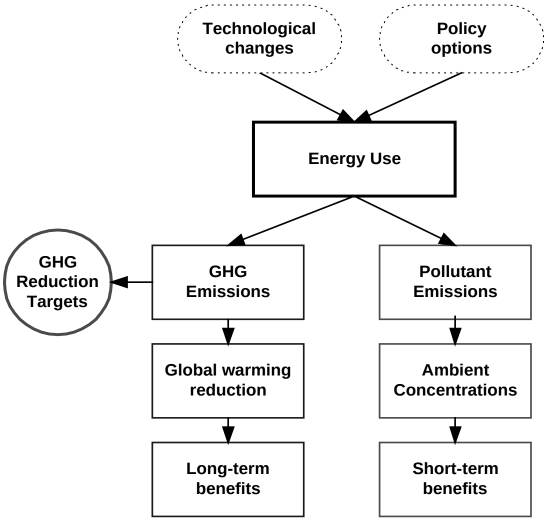
While many developing countries have conducted extensive analysis of possible greenhouse gas mitigation measures, relatively little attention has been given to full characterization of the more immediate environmental and health benefits that would result from these measures. Understanding those benefits has been a critical gap in past efforts to help developing countries estimate the cost of GHG abatement policies. Improving a country's understanding of the scope and potential magnitude of those direct public health benefits can help policy makers to take better decisions, by considering the full impact of adopting alternative climate change mitigation policies.

Figure 1 shows a schematic view of the potential social benefits resulting from measures aimed at reducing GHG emissions. Technological changes and policy options aim at reducing energy use to achieve the target in GHG emissions reductions. The path in the left side of the diagram shows that global warming reductions lead to long-term benefits, such as reduced extreme weather events, sea level rise, and communicable diseases spread, among others. However, these benefits are uncertain, at least to the same extent that global warming itself is uncertain. Also, from the standpoint of a single country, reducing the threat of global warming can be seen as a public good. Therefore, it is not strange that developing countries are more worried with local, immediate environmental and human health needs, such as control of air and water pollution, than with long-term problems such as global warming.

Nonetheless, the right path of the diagram in Figure 1 shows another set of benefits stemming from the measures aimed at reducing GHG emissions. In fact, the same combustion processes that lead to emissions of GHG also produce local and regional pollutants, like particulate matter (PM), sulfur dioxide (SO_2), and nitrogen oxides (NO_x). Thus, any measure aimed at reducing GHG emissions that also produces concomitant reductions in those pollutants, will lead to short and mid-term benefits from air pollution reduction, such as reductions in health effects associated to air pollution, reduction in vegetation and materials damages, and visibility improvements. Since these benefits can be considered a 'side effect' of the GHG mitigation measures, they are referred to as 'ancillary' benefits.

If properly assessed, consideration of these ancillary benefits may allow for implementation of policy measures that would otherwise have not been taken. If the ancillary benefits exceed the mitigation costs, they may even allow for "no regrets" GHG abating measures, in which taking immediate action to reduce GHG will be justified only by those benefits, even without consideration of the long-term benefits from GHG emissions reduction. Of all these benefits, those associated to health effects are probably the more important ones. They are the benefits considered in this report.

Figure 1. **Short and long term social benefits derived from measures aimed at reducing GHG emissions**

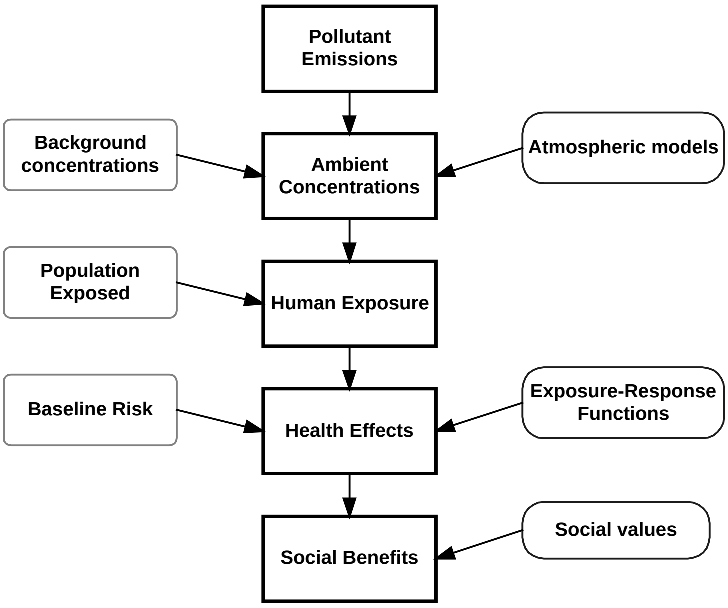


2. Methods

There are several levels at which the analysis of ancillary benefits of GHG mitigation can be conducted. The most detailed would be an analysis of individual mitigation measures, in which the changes in GHG and pollutant emissions associated to each policy or technological measure are estimated, and linked to a change in health effects in the population. This requires a great deal of data. The impacts of different mitigating measures are likely to vary according to the location and duration of the reduction in emissions, the population density close to the sources, and the prevailing meteorological conditions. In this work we took a global approach, conducting the analysis at an aggregate level for the whole country.

The first step to estimate the short-term health benefits is to link each policy or technological measure to the reduction in emissions pollutants. Once the changes in pollutant emissions have been assessed, it is necessary to link them to changes in ambient concentrations, population exposure, health effects and social benefits, using the Damage Function Method, showed schematically in the Figure 2.

Figure 2. **Damage Function framework used to estimate the social benefits of a reduction in emissions of primary air pollutants**



Fine particulate matter ($PM_{2.5}$) was used as a sentinel pollutant to estimate the change in health effects. We choose to concentrate on $PM_{2.5}$ because studies conducted in the U.S. (Schwartz, Dockery *et al.* 1996) as well as our own studies in Santiago (Cifuentes, Vega *et al.*. 2000) have shown that the fine fraction of particulate matter is more strongly associated to health effects, especially mortality effects, than the coarse fraction of PM_{10}.

To estimate the potential health benefits for the whole country, we assembled a database of the current exposure of the Chilean urban population to particulate matter. Several studies led by the National Commission of the Environment (CONAMA) have measured particulate air pollution in cities that comprise almost half of the country´s urban population.

The changes in ambient concentrations of particulate matter were estimated using two methods: one based on source apportionment of ambient fine particles concentrations; the other was based in statistical associations between atmospheric pollutants. Although both methods were developed using data specific for Santiago, we applied their results to the whole country. This assumes that the atmospheric processes for the rest of the country are similar to Santiago's, which is a crude assumption. Unfortunately, due to limited data, this was the only option available to us at this time.

With the projected ambient concentrations for each policy scenario, we computed the population exposure in each year. Based on data of a previous study in which we estimated the social losses due to particulate air pollution in Santiago, we estimated the health damages for the CP and the BAU scenario, obtaining the social benefits as the difference of the two. In the next sections we describe in detail the methods used in each step.

3. Emissions scenarios

We have considered two emissions scenarios: the Business-as-usual scenario (BAU), in which no GHG mitigation measures are taken, and a Climate Policy scenario (CP), in which measures are taken to reduce emissions of GHG.

We have relied on the results obtained in a previous study contracted by the Chilean Environmental Commission to the Research Program on Energy of the University of Chile (PRIEN 1999). The study projected the emissions for several greenhouse gases, including carbon dioxide (CO_2), nitrous oxide (N_2O) and methane (CH_4) and several primary pollutants, including sulfur dioxide (SO_2), nitrogen oxides (NO_x) and non-methane hydrocarbons (NMHCs). Those projections were based on an engineering, bottom-up approach, considering technological measures like efficiency improvements and fuel switching to obtain emissions reductions. For the base case, policies that are currently in place and those which are scheduled to be applied were considered. In particular, all the measures of the Decontamination Plan for the Metropolitan Region that are scheduled to be implemented in Santiago were considered (Comisión Nacional del Medio Ambiente 1997), as well as the future investments in infrastructure contained in the national strategic plan developed by the Transportation Ministry (MOP 1997).

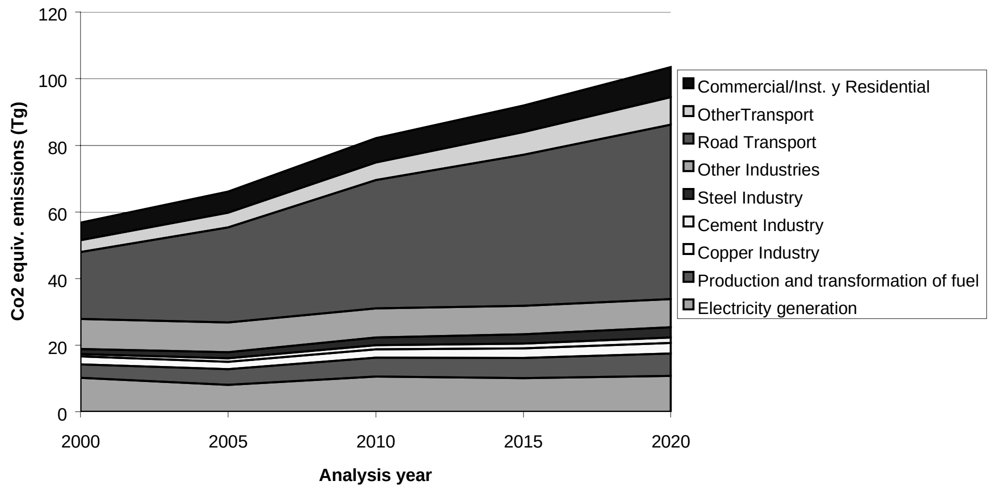
The most relevant assumptions considered for the projection of the base scenario from 2000 up to 2020 are:

- An average annual GDP growth of 4.5% for the whole period of analysis (2000 - 2020).
- A urban population increase of 1.9 % per year during the whole period.
- A constant rural population of around 2.5 million people.
- No substantial variations in the prices of energy.

3.1 Business as usual scenario

The Business-as-usual scenario projected the emissions of CO_2, CH_4 and N_2O for the years 2000 to 2020, in 5-year intervals. Based on the Global Warming Potential (GWP) recommended by IPCC for each greenhouse gas (IPCC 1996) we computed the CO_2 equivalent emissions for each period. The emissions were projected for the different sub-sectors of the economy. For the analysis, we aggregated the data into the most relevant sub-sectors, as shown in the next figure.

Figure 3. **CO_2-equivalent emissions by sector for the BAU scenario**



Source: Aggregation of data from (PRIEN 1999).

It can be observed in the figure that the baseline CO_2-equivalent emissions would grow around 80% during the period 2000-2020 for the BAU scenario. This high growth is explained mainly by the explosive growth of the emissions in the road transport sector, as is clearly seen in the figure.

3.2 Emission reduction potential in the Climate Policy scenario

We consider as Climate Policy (CP) scenario the mitigation scenario developed by the Research Program in Energy of the University of Chile (PRIEN 1999). This mitigation scenario was developed following the bottom-up (or engineering) approach, considering the introduction of newer, more efficient technologies and computing the incremental cost and emissions reductions. Since the objective of PRIEN's study was to estimate emissions reductions that could be achieved through "no-regrets" implementation of technologies, the adoption and rate of penetration of the technologies was determined such that they would represent a net cost saving to the user. New technologies were considered for all sectors: residential, commercial, industrial and transport. Technologies considered in the residential/commercial sector included for example improved appliances and compact-fluorescent lamps. In the industrial sector, the main technologies considered were more efficient electric motors and the increased use of co-generation. In the transport sector the main mitigation measures were mode switching to cleaner means of transportation, and improvements in the fuel efficiency of the existing means of transport.

Due to the way the mitigation scenario was constructed, we can assume that mitigation costs are negative or close to zero. Therefore, the associated reductions in greenhouse gas emissions are relatively small. This may be a serious limitation, since we are then computing the ancillary benefits for the first mitigating measures in terms of control cost, without going up the marginal cost mitigation curve. If the mix of GHG and local pollutant emission reductions change for this measures, then the estimates for the ancillary benefits will also change.

The next table presents the projected CO_2 equivalent emissions reductions for the CP scenario, compared to the BAU scenario, for the years 2010 and 2020. The Steel Industry and the Other Industries sub-sectors show the biggest percentage reductions, of 23% and 20% respectively, for the year 2020. However, the biggest reduction in mass corresponds to the road transport subsector.

Table 1. **CO_2-eq emissions reductions by subsector of the economy (Tg)**

Sector	**Emissions (Tg)**			**Reductions CP with respect to BAU**	
	2010	**2010**	**2020**	**2010**	**2020**
Electricity generation	10.2	10.6	10.8	6.2%	14.9%
Production and transformation of fuel	4.1	5.6	6.7	3.4%	8.4%
Copper Industry	2.4	2.5	3.2	4.9%	10.3%
Cement Industry	0.7	1.2	1.6	0.0%	0.0%
Steel Industry	1.4	2.3	3.2	16.1%	23.2%
Other Industries	9.1	8.8	8.4	9.8%	20.3%
Road Transport	20.1	38.4	52.4	6.2%	14.9%
OtherTransport	3.6	5.4	8.2	-0.6%	-0.9%
Commercial/Inst. y Residential	5.2	7.3	9.1	3.7%	8.1%
Total	**56.8**	**82.2**	**103.5**	**5.9%**	**12.9%**

Source: aggregation of data from (PRIEN 1999).

The next table presents the emissions reductions for the CP scenario, compared to the BAU scenario, for the GHG and the primary pollutants. The percentage reductions for all primary pollutants is similar to the reductions in CO_2-equivalent, except for SO_2, for which the reduction is slightly higher, due to the introduction of compressed natural gas in the country (starting at Santiago and other major cities) and to the sulfur reduction program in liquid fuels (gasoline and diesel).

Table 2. **Emission reductions for each primary pollutant in 2010 and 2020**

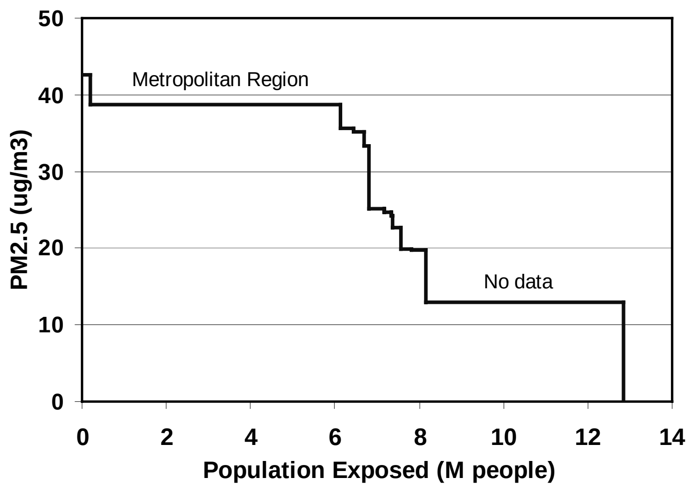
Pollutant	**2010**		**2020**	
	Gg	**% red**	**Gg**	**% red**
CO2eq	4,833	5.9	13,392	12.9
CO2	4,537	6.0	12,404	13.1
CH4	2.95	2.4	7.12	5.4
N2O	0.75	5.5	2.71	14.4
CO	33.14	4.0	102.30	11.2
SO2	7.15	6.7	15.88	14.9
NOx	15.92	5.2	40.87	10.7
VOCNM	5.10	4.3	15.52	11.6

4. Human exposure to air pollution in Chile

Chile has a widespread ambient particulate matter pollution problem. Santiago, the capital of Chile, is one of the world's most polluted cities by particulate matter. Regular daily measurements of $PM_{2.5}$ and PM_{10} using dichotomous samplers began in 1988 in five stations across the city. The original network was expanded in 1997 with a new network of eight monitoring stations. Regular monitoring is not currently conducted in any other city, except a few localities close to megasources like copper smelters and power plants, where the law mandates regular monitoring to ensure that ambient air quality levels are not violated. Some research projects [(SESMA 1999), (CIMM 1998), (Gredis 1999), (Cosude 1999)] have conducted sporadic measurements in several other cities though.

We gathered all the available concentrations data to estimate the current level of exposure of the Chilean population to fine particulate matter. All cities which have some particulate matter measurement, either $PM_{2.5}$ or PM_{10}, comprise a total of 7.8 million people, or about 63% of the urban population of Chile in 2000. For cities that did not have measurements of $PM_{2.5}$, we estimated them from the PM_{10} concentrations, based on the national average ratio of $PM_{2.5}$ to PM_{10}. For those cities with no measurements at all, we assumed a level equal to the cleanest city measured, that is, 13 $\mu g/m^3$. This assumption is probably an underestimate, since some industrial zones that currently lack measurements, probably have a higher concentration. The next figure shows the estimated exposure to fine particulate matter for all the urban population in Chile. The figure underlines the relative importance of the Metropolitan Region of Santiago in the total exposure of Chilean population. The total population exposure in 2000 will be 344.5 million person* ($\mu g/m^3$) of $PM_{2.5}$, of which 66% correspond to the Metropolitan Region.

Figure 4. **Population exposure of the Chilean urban population to $PM_{2.5}$ in 2000**



5. Changes in air pollutant concentrations due to changes in emissions of primary pollutants

This step is a crucial part of the method linking emissions of primary pollutant to social losses. For a detailed analysis, it should rely on atmospheric dispersion models, specifically in models that incorporate the complex set of chemical reactions occurring in the atmosphere. None of those models is available for Chile at this time. For this analysis, we estimated the impacts of emissions changes on PM concentrations based on two approximate methods, described in the following sections.

5.1 Method 1: Use of a box model to develop emission concentration relationships

A simplified methodology was used to estimate the future impacts of PM_{10}, $PM_{2.5}$ and coarse fractions. The starting point is the Eulerian Box model approach that reads:

$$\frac{dC_i}{dt} = \frac{q_i}{H(t)} + R_i - \frac{V_{d,i}}{H(t)} C_i + \frac{u(t)}{\Delta x}\left(C_i^U - C_i\right) + \frac{(C_i^A - C_i)}{H(t)}\frac{dH}{dt} \qquad (1)$$

where C_i is the pollutant concentration ($i = CO, SO_2, PM_{10}, PM_{2.5}$, etc.), $H(t)$ is the mixing height, q_i the surface emission within the box, R_i the net production rate by chemical mechanisms, $V_{d,i}$ the deposition flux (dry and wet) at the ground surface, $u(t)$ the average wind speed in the box and the superscripts U and A stand for upwind and aloft advected concentrations, respectively. The rightmost term on the right hand side of (1) is only applicable whenever the mixing height is rising, that is, from sunrise until early afternoon (Seinfeld and Pandis 1998).

The above equation describes mathematically the concentration of species above a given area, accounting for emissions, chemical reactions, removal, advection of material in and out of the airshed and entrainment of material during growth of the mixed layer. The strongest assumption is that the corresponding airshed is well mixed.

If equation (1) is integrated for a pollutant like CO or SO_2, and assuming a first order decay process, it can be shown that the following relationship holds:

$$\langle C_i \rangle = \left\langle \frac{q_i}{k_i \cdot H + v_{d,i} + \frac{u \cdot H}{\Delta x}} \right\rangle + \left\langle \frac{C_i^o}{1 + \frac{k_i \cdot \Delta x}{u} + \frac{v_{d,i} \cdot \Delta x}{H \cdot u}} \right\rangle + \delta_i \qquad (2)$$

The above equation is a linear relationship between emissions and concentrations, and it was used to generate long-term forecasts of CO and SO_2 for Santiago for 2000-2020. The emissions of CO and SO_2 come from fuel consumption, so the model parameters were calibrated using measured ambient concentrations and historical data on fuel consumption and fuel sulfur content.

Next, in order to model the emission term for particulate matter fractions, it was assumed that the emissions of particulate matter can be expressed as a sum of contributions coming from mobile sources, stationary sources and other sources in the following manner:

$$q_{PM10} = \alpha (q_{CO})_{Mobile\ Sources} + \beta (q_{SO2})_{Stationary\ Sources} + \gamma \qquad (3)$$

where

α = ratio of PM_{10} to CO emissions in the mobile sources
β = ratio of PM_{10} to SO_2 emissions in the stationary (industrial, commercial and residential) sources
γ = emissions not directly linked to mobile or stationary source emissions

Therefore, α stands for the ratio of PM_{10}/CO in the emissions from the fleet in Santiago, β represents the ratio of PM_{10}/SO_2 emissions in industrial and residential sources and γ is a term independent of those emissions, and it is associated to mechanisms such as construction activities, wind erosion, agricultural activities, forest fires, etc.

Using equation (2) for CO and SO_2 and inserting equation (3) within the box model equation for PM_{10} leads, after some manipulation and simplification to:

$$\langle C_{PM10} \rangle = a \langle C_{CO} \rangle + b \langle C_{SO2} \rangle + \frac{c}{u} + d\frac{P}{u} + e \qquad (4)$$

Where u is the average wind speed and P the total precipitation recorded (this takes into account of the wet deposition term). Therefore, a linear regression for the daily averages of PM_{10}, $PM_{2.5}$ and coarse fractions against the daily averages of CO, SO_2, (1/u) and (P/u) will produce estimates of the unknown parameters in the model.

5.1.1 Parameter estimation and model validation

In order to validate the above model, data gathered at Santiago for the fall and winter seasons from 1990 to 1994 were used to fit the model (in some cases, data from 1995 and 1996 were used to increase the database, as was the case in Station C). The air quality data came from the MACAM monitoring network, and included hourly measurements of CO, SO_2, and surface wind speed u plus daily measurements of PM_{10}, $PM_{2.5}$ and coarse fractions. A substantial amount of time was devoted to extracting daily averages of the different terms appearing in equation 2, considering missing values, analyzing partial scatter plots to detect outliers, and so on. Model parameters were obtained by using classical, linear regression analysis of equation (4). In this fashion, we could estimate the model parameters for stations A, B, C and D of the MACAM network, and for the three fractions: PM_{10}, $PM_{2.5}$ and coarse particles. In particular, we can estimate the different contributions to the total, ambient particle concentrations coming from:

a) Advected and secondary particles, lumped together in the c/u term in equation (4)

b) Directly emitted by mobile sources, and so proportional to CO concentrations

c) Directly emitted by stationary sources, and so proportional to SO_2 concentrations

d) Deposited onto the ground by wet precipitation removal

From the 1997 Emission Inventory for Santiago (EIS), as developed by CENMA (1997), the estimated ratio of PM_{10} emissions from mobile sources to total CO emissions is:

$$\frac{2730(ton/yr)}{244921(ton/yr)} = 0.011(g/g)$$

The *a* coefficients for the CO concentration in the PM_{10} model have the values 10.25, 9.97, 19.57 and 7.78 at stations A, B, C and D, respectively, when CO is measured in ppm and PM_{10} in ($\mu g/m^3$). In units of (g/g), the coefficients take the values 0.009, 0.0087, 0.017 and 0.0068 for stations A, B, C and D, respectively. All coefficients are significant ($p<0.05$). The similar results among monitoring sites and their reasonable agreement with the value estimated above from the annual emission inventory for Santiago show that the box model is capable of reflecting these relationships among primary emissions.

From the 1997 EIS the ratio of PM_{10} to SO_2 emissions for the stationary sources is:

$$\frac{3175/ton/yr)}{21169(ton/yr)} = 0.15(g/g)$$

In this same units, the fitted values for *b* are 0.24, 0.21, 0.50 and 0.58, with all of them being significant ($p<0.05$) for stations A, B, C and D, respectively. The reason for *b* values higher that the value given by the emission inventory is that the ratio of PM to SO_2 is enhanced by the faster removal of SO_2 from the gas phase. In other words, by the time emissions reach a monitor site, a significant amount of SO_2 has already been deposited or degraded by chemical or physical mechanisms. This is more evident for stations C and D, which are rather away from major traffic lanes and so tend to be impacted by rather aged plumes, associated with regional scale dispersion of sulfur in Central Chile. On the other hand, stations A and B are located near busy streets, so they are impacted by fresh emissions coming from mobile sources. By contrast, this effect does not show up for CO, because its rate of chemical oxidation is fairly low, and so is its deposition velocity (Seinfeld and Pandis, 1998).

In addition, the intercepts (*e* coefficients) on the lineal regression equation produce estimates of the background levels of PM_{10}, $PM_{2.5}$ and coarse fractions. This is relevant information to be used in the estimation of future concentration impacts. We have estimated that background levels of PM_{10}, $PM_{2.5}$ and coarse particles are around 45, 27 and 18 ($\mu g/m^3$), respectively. These three values compare very well with the measurements made by (Artaxo 1998) at Buin, a rural site 35 km south of Santiago considered representative of upwind, background values for the greater Santiago area. The values reported by Artaxo *et al.* in the winter 1996 campaign were 52, 29 and 23 ($\mu g/m^3$), for PM_{10}, fine and coarse particles respectively. The major difference lies in the coarse fraction, but (Artaxo 1998) measured $PM_{2.0}$ as fine fraction, thus explaining their larger estimates of the coarse particle background.

We have to recall that the box model cannot account for the generation of secondary aerosols (mostly sulfates and nitrates), because the chemistry of these processes is far too complex to be included within a simplified model like this one. We cannot estimate the magnitude of this uncertainty until a comprehensive simulation of those processes is carried out for Santiago. Nevertheless, the model parameters were fitted using actual data recorded at the monitoring network, so the model should represent reliably the PM levels within the city.

5.1.2 Projection of future impacts

From the previous results, the working equation to estimate future concentrations under new emission scenarios is obtained from (4) in the following way

$$\langle C_{PM10} \rangle_{2XYZ} = a\langle C_{CO} \rangle_{2XYZ} + b\langle C_{SO2} \rangle_{2XYZ} + \left(\frac{c}{V}\right)_{HISTORICAL} + d\left(\frac{P}{V}\right)_{HISTORICAL} + e_{HISTORICAL} \qquad (5)$$

Where 2XYZ stands for any future scenario. In addition:

a) The CO and SO_2 concentrations are forecasted using the box models calibrated with historical data from 1990 to 1998.

b) It will be assumed that Santiago will follow the same trend in emissions as the whole country in the PRIEN annual emission forecasts.

c) The estimates of contributions of resuspended dust and wet scavenged particle concentrations will be assumed to stay in the same values as in the model calibration period. That is, we assume that the emission factor for resuspended particles will stay the same. Given the uncertainties in estimating this type of emission factor, we consider the above approximation reasonable; for instance, (Venkatram 1999) have reported estimates for this emission factor between 0.1 and 10 g/VKT for a metropolitan area (VKT are the total kilometers traveled by all vehicles in a given period).

d) The proportion of particles that are deposited by wet mechanisms is assumed to be the same as the values computed from the regression analyses: about 1 to 2% for most of the fractions. This means that, at least for Santiago, these quantities can also be incorporated in equation (5) as fixed proportions of the total, average concentration <Ci> therein.

5.1.3 Results of the simulated scenarios

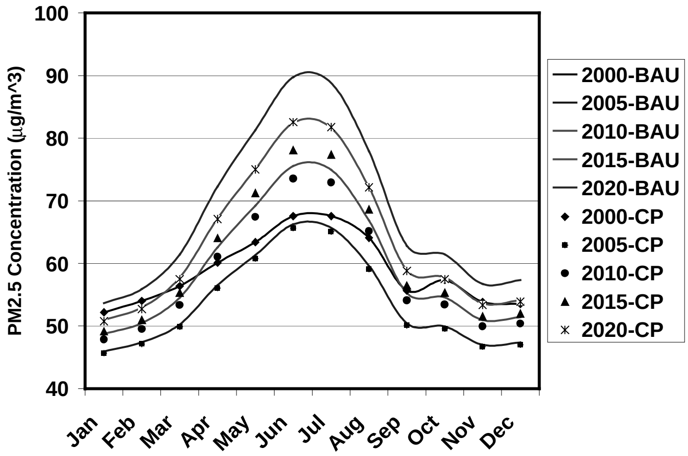
In order to simulate impacts for the BAU and CP scenarios, the following specific assumptions were made:

i) Background concentrations were kept at the same values as 1994. Although (Artaxo 1998) have estimated long range contributions from copper smelters that will undergo emission reduction plans, these plans will be pursued regardless of the long-term GHG policies (if any) in the country, that is either under BAU or CP scenarios.

ii) The parameters obtained for the different monitor stations will be kept fixed at their estimated values for the calibration period (1990-1996).

The next figure shows the projected impacts of $PM_{2.5}$ at monitoring station B; similar results hold for the other stations, so they are not shown here. It is clear that by 2020 the two scenarios achieve different impacts, with CP concentrations being lower by up to 7 $\mu g/m^3$.

Figure 5. **Projections of $PM_{2.5}$ concentrations at monitoring Station B**



5.2 Method 2 : Source apportionment of fine particular matter concentrations

In this approach, we estimated the changes in ambient PM concentrations due to changes in primary pollutant emissions using an alternative method. The method is based on source apportionment data of $PM_{2.5}$ concentrations to primary pollutants conducted in Santiago in 1996 and 1998 (Artaxo 1996; Artaxo 1998; Artaxo, Oyola *et al.*. 1999). We computed the fraction of $PM_{2.5}$ concentrations in Santiago attributable to each primary pollutant, based on those measurements, and obtained the fractions shown in the next table.

Table 3. **Percentage of $PM_{2.5}$ concentrations attributable to each primary pollutant in Santiago, 1998**

Primary Pollutant	**Percentage attributable**	**90% CI**
Resuspended Dust	5.0%	(0.5% - 10%)
SO2	20.0%	(15.5% - 25%)
NMHC	0.0%	(0% - 0%)
NOx	30.0%	(21.1% - 39%)
PM10	33.5%	(24.6% - 42%)
Other	11.5%	

Source: own estimates based on (Artaxo 1996) , (Artaxo 1998) and (Artaxo, Oyola *et al.*. 1999).

In the above table PM_{10} should be understood as primary emission of PM, whereas SO_2 and NO_x are associated to secondary sulphates and nitrates, respectively. Assuming that the contribution of each primary pollutant remains fixed over time in the value given in Table 3 above, then the relative change in ambient $PM_{2.5}$ concentrations can be expressed as:

$$\Delta\%[PM_{2.5}] = \sum_i F_i \cdot \Delta\%[P_i] \qquad (6)$$

where

- $\Delta\%[PM_{2.5}]$ is the relative change in $PM_{2.5}$ concentrations.
- $\Delta\%[P_i]$ is the relative change in pollutant i concentrations.
- F_i is the fraction of $PM_{2.5}$ apportioned to pollutant i, according to Table 3.

This equation should be applied only to the fraction of the $PM_{2.5}$ concentrations above background concentrations. However, we should consider only the natural background, not the background due to emissions occurring elsewhere in the country. In effect, if we are conducting an analysis for the whole country, assuming a relatively uniform distribution of pollutant sources within the country, the background concentration in any given city will also change when the level of emissions changes within the whole country.

6. Health impact estimates

There is a growing number of studies linking particulate air pollution with both mortality and morbidity all over the world. For short term effects, the work of Dockery and Schwartz in the late eighties has been replicated in more than 40 cities to date (and the number keeps growing), although still most of the studies come from US and European cities. For chronic effects, two prospective studies conducted in the US, the Harvard Six cities study (Dockery, Pope III *et al.*. 1993) and the Pope and colleagues study (Pope III, Thun *et al.*. 1995) have shown significant results, in agreement with results from earlier cross-sectional studies (Lave and Seskin 1977). Although the causal mechanism by which exposure to particulate matter can induce death is not yet know, there is not much doubt than the association is not a spurious one, and the US has moved towards more stringent standards based on the recent studies (EPA 1997).

For morbidity effects, studies in several countries have associated particulate matter with a number of health endpoints, including hospital admissions, emergency room visits, increased incidence asthma attacks, work loss days, restricted activity days, and minor symptoms, as well as increased incidence of chronic bronchitis (EPA 1996).

Most of the studies linking air pollution and health are based on a Poisson model. In this model, the relative risk (RR) associated with a change in the PM concentrations is given by:

$$RR(\Delta PM) = \exp[\beta * \Delta PM] \qquad (7)$$

The slope coefficient, β, is obtained from the epidemiological studies, as will be shown later. ΔPM is the change in PM concentrations from a reference concentration. The relative risk needs to be applied to a base number of effects, which is obtained from the observed number of effects on the population that is exposed to a given level of air pollution. Therefore, the number of health effects at a given concentration C, is given by:

$$\text{Effects}(C) = \exp\left(\beta \cdot (C - C_0)\right) \cdot \mathrm{R}_0 \cdot \text{Pop} \qquad (8)$$

where R_0 refers to the base rate of effects at concentration C_O, and is generally obtained from health statistics data, and Pop is the exposed population. The above formula assumes that there is no threshold in the effects. If there is a threshold in the effects, i.e. a concentration C_T below which there are no effects, the formula becomes:

$$\text{Effects}(C) = \exp\left(\beta \cdot (C - \max\{C_0, C_T\})\right) \cdot \mathrm{R}_0 \cdot \text{Pop} \qquad (9)$$

For some studies the above formula applies to daily effects, and the effects rate should be expressed as the number of effects per day. To obtain the number of excess effects in a year, it is necessary to add up the effects for all days of the year. If there is a threshold, the summation becomes more complicated. For computing the exact number of effects in this case it is necessary to know the form of the frequency distribution of the daily concentrations. Generally, it is assumed that daily concentrations follow a lognormal distribution (Ott 1990), although other distributions have been shown to better represent the physical process underlying air pollution concentrations (Morel *et al.*, 1999).

Exposure-response functions. We conducted the analysis based on exposure-response functions obtained from the literature, mainly from the estimation of benefits of the Clean Air Act performed by EPA (EPA 1997, EPA 1999) and from the recommendations of the World Health Organization by Ostro (Ostro 1996). We complemented these sources with exposure response functions from studies performed in Santiago. For mortality we used our own results (Cifuentes, Vega *et al.* 2000). For child medical visits, we used (Ostro, Eskeland *et al.* 1999) and (Illabaca, Olaeta *et al.* 1999). All of the studies correspond to short-term effects, except for chronic bronchitis and long-term exposure mortality. Following Ostro 1996, for mortality due to long-term exposure, we used the coefficient from the study of Pope *et al.* (Pope III, Thun *et al.* 1995) only for the high case, i.e., our mid estimate of mortality does not consider the chronic effects of pollution. Whenever possible, we used exposure-response functions based on $PM_{2.5}$. If they were available only for PM_{10}, we convert them to $PM_{2.5}$ using the relation $PM_{2.5} = 0.55\ PM_{10}$.

We considered three age groups in the analysis: Children 0-18 yrs, Adults, 18-64 yrs, and 65+ yrs, In some cases, we considered specific age groups, like for asthma attacks, in which the exposure-response functions are for children below 15 yrs. The summary of the exposure-response coefficients for the effects considered is shown in the next table.

Table 4. **Summary of exposure-response coefficients used in the analysis**

Endpoints	Age Group	β	σ_β	Source
Mortality (long term exp)	>30 yrs	0.00640	0.00151	Pope et al,1995
Chronic Bronchitis	> 30 yrs	0.02236	0.007891	Schwartz,1993
Mortality (short term exp.)	All	0.00120	0.000304	Cifuentes et al, 2000
Hospital Admissions RSP	> 65 yrs	0.00169	0.000447	Pooled
Hospital Admissions COPD	> 65 yrs	0.00257	0.000401	Pooled
Hosp. Adm Congestive heart failure	> 65 yrs	0.00135	0.000565	Schwartz & Morris, 1995
Hosp Adm Ischemic heart disease	> 65 yrs	0.00090	0.000400	Schwartz & Morris, 1995
Hospital Admissions Pneumonia	> 65 yrs	0.00134	0.000264	Pooled
Asthma Attacks	All	0.00144	0.000315	Ostro et al, 1991
Acute Bronchitis	8-12 yrs	0.00440	0.002160	Dockery et al., 1989
Child Medical Visits LRS	< 18 yrs	0.00083	0.000330	Ostro et al, 1999
Emergency Room Visits	All	0.00222	0.000427	Sunyer et al, 1993
Shortness of Breath (days)	< 18 yrs	0.00841	0.003630	Ostro et al, 1995
Work loss days (WLD)	18-65 yrs	0.00464	0.000352	Ostro et al, 1987
Restricted Act. Days (RAD)	18-65 yrs	0.00475	0.000288	Ostro et al, 1987
Minor Restricted Act. Days (MRAD)	18-65 yrs	0.00741	0.000704	Ostro et al, 1989

Base rate of effects. The other parameters needed to compute the total number of effects are the exposed population and the effects base rate. We projected the exposed population using the estimates of the Chilean Institute of Statistics, considering that the age distribution remains constant. For the base rate of the effects we used the rates for Santiago for all the cities.

7. Effects valuation

To estimate the social benefits associated to reduced health effects, it is necessary to estimate society's losses due to the occurrence of one extra effect. Several methods exist to value such losses. The most straightforward one is based on the direct losses to society stemming from the cost of treatment of each effect plus the productivity lost. This approach, known as the human capital method for mortality effects, and the cost of illness for morbidity effects, suffers from a serious limitation, by not considering the willingness to pay of the individuals to avoid the occurrence of an extra effect, or to reduce her risk of death. However, because values are easier to compute and defend, it has been used in previous analysis of quantification of air pollution effects, such as the economic valuation of the benefits associated to the Decontamination Plan of Santiago (Comisión Nacional del Medio Ambiente 1997).

We choose to use values that reflect the willingness to pay of individuals to reduce the occurrence of one extra effect. Since there are no such values available for Chile, the unit values of the effects are based on those used by the US EPA (EPA 1999), transferred to Chile using the ratio of the per capita income of both countries. By far, the more important effects are premature mortality. For these effects, we choose a lower bound from the range of values used by EPA, which became US$338 thousand after adjustment, for the year 1997. This value falls within the range of values that we have obtained in a pilot test of a contingent valuation study of willingness to pay for reducing mortality risks in Santiago (Cifuentes, Prieto *et al.* 1999). For The summary of values used in the analysis in shown in the next table. The values were updated annually using a projected growth in real per capita income of 2.6%.

Table 5. **Unit values for each effect for the year 1997 (1997US$ per effect)**

Endpoint	**mid**	**90% CI**
Mortality (long term exp)	281,209	(111,956 - 707,906)
Chronic Bronchitis	45,556	(22,192 - 68,921)
Mortality (short term exp.)	338,549	(134,785 - 852,252)
Hospital Admissions RSP	2,796	(2,796 - 2,796)
Hospital Admissions COPD	3,624	(3,597 - 3,651)
Hosp. Adm Congestive heart failure	3,832	(3,815 - 3,849)
Hosp. Adm Ischemic heart disease	4,755	(4,742 - 4,767)
Hospital Admissions Pneumonia	3,670	(3,654 - 3,686)
Asthma Attacks	7	(3 - 11)
Acute Bronchitis	10	(4 - 16)
Emergency Room Visits	54	(33 - 74)
Child Medical Visits	165	(133 - 198)
Shortness of Breath (days)	1	(0 - 2)
Work loss days (WLDs)	18	(18 - 18)
RADs	9	(5 - 12)
MRADs	8	(5 - 12)

Source: Values from EPA (1999) transferred for Chile using the ratio of per capita income.

8. Uncertainty and variability analysis

As has been shown in the preceding sections, each step of the analysis is fraught with uncertainty. Explicit consideration of all the uncertainties is crucial to illuminate the analysis for several reasons (Morgan and Henrion 1990):

- It lets us identify the important factors in the analysis.
- It can help us identify which steps of the analysis need to be improved the most.
- It points out potential sources of disagreement between different experts or analysts.

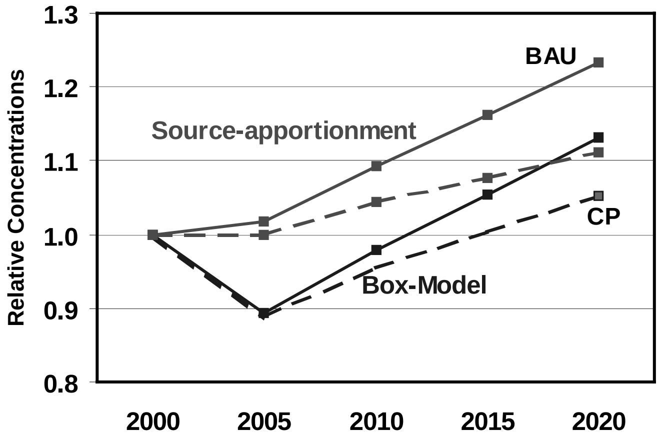
Uncertainty can be classified into parameter uncertainty, model uncertainty, and scenario uncertainty. In this analysis, we have considered explicitly only the first two. Parameter uncertainty can be modeled quantitatively treating the parameters as random variables. We have done so for the exposure-response coefficients for health effects quantification, for some parameters of the ambient concentration models, and for the unit values of the effects. A more difficult kind of uncertainty is model uncertainty. As discussed in Section 5, we have considered two different models to estimate the change in $PM_{2.5}$ concentrations due to changes in emissions, this was considered necessary because of the relevance of air pollution dispersion modeling in the framework depicted in Figure 2.

To consider quantitatively the uncertainty in the analysis, the model was implemented in the Analytica modeling environment (Lumina Decision Systems 1998), which is based on Montecarlo simulation. This very flexible modeling environment let us propagate and analyze the uncertainty of the parameters and the results.

9. Results

Based on the emissions changes presented in Section 3, we estimated the evolution of $PM_{2.5}$ concentrations in time for both methods proposed in Section 5. The next Figure shows the mid estimates of the projected $PM_{2.5}$ concentrations for each scenario, using both methods of estimating the concentrations. The concentrations are referred to the concentrations in the year 2000.

Figure 6. **PM2.5 concentrations relative to year 2000 concentrations, for both methods of estimating the concentrations**



The figure shows that both methods produce similar results for each scenario, BAU and CP, with the concentrations increase being driven mainly by the increase in NO_x and PM emissions. However, given the consideration of different primary pollutant emission changes, the difference between the BAU and CP scenarios is approximately 50% bigger for the source apportionment method. The more pronounced minimum in 2005 for the Box model approach is caused by the heavier weight given to CO and SO_2 concentrations, with respect to the source apportionment approach.

Applying the changes in $PM_{2.5}$ concentrations to the exposed population in each city it is possible to compute the excess health effects for each scenario. The next table shows the avoided excess health effects in the year 2010 and 2020. The excess effects have been computed assuming there is no threshold in any of the effects. The table shows the mid value of the effects for each policy scenario, grouped by type of effect, summed up over all age groups, and the 90% confidence interval. We show the results for the source apportionment method. The values for the Box model are smaller.

Table 6. **Avoided health effects for the years 2010 and 2020**

Endpoint	2010 mid	2010 90% CI	2020 mid	2020 90% CI
Premature Deaths	100	(62 - 431)	305	(189 - 1,290)
Chronic Bronchitis	710	(503 - 854)	2,157	(1,526 - 2,572)
Hospital Admissions	619	(480 - 797)	1,887	(1,450 - 2,423)
Emergency Room Visits	9,972	(6,431 - 14,882)	30,095	(19,654 - 44,984)
Child Medical Visits	4,837	(1,919 - 8,178)	14,642	(5,866 - 24,878)
Asthma Attacks & Bronchitis	133,022	(86,530 - 183,840)	399,351	(263,016 - 556,863)
Restricted Activity Days	2,878,743	(1,868,859 - 3,716,428)	8,804,442	(5,660,315 - 11,270,793)

Note: $PM_{2.5}$ concentration changes estimated using source apportionment method, equation (6).

The next table shows the total number of effects avoided from 2000 to 2020 for the BAU-CP scenario comparison.

Table 7. **Total number of health effects avoided in the CP scenario with respect to the BAU scenario during the period 2000 to 2020**

Endpoint	Total effects avoided mid	Total effects avoided 90% CI
Premature Deaths	2,771	(1,546 - 10,840)
Chronic Bronchitis	18,130	(10,710 - 22,170)
Hospital Admissions	15,000	(12,930 - 20,760)
Emergency Room Visits	247,200	(166,600 - 353,400)
Child Medical Visits	118,600	(47,560 - 205,400)
Asthma Attacks & Bronchitis	3,339,000	(1,981,000 - 4,998,000)
Restricted Activity Days	75,430,000	(43,650,000 - 96,670,000)

Note: $PM_{2.5}$ concentration changes estimated using source apportionment method, equation (6).

For the whole period of analysis, the mid estimate is around 2,800 deaths that can be avoided, with a 90% confidence interval of 1,500 to 10,800 (the upper bound of this interval is high because it includes long-term exposure deaths). Most of these effects will occur in the Metropolitan Region of Santiago.

Using the unit values shown in the preceding chapter, we computed society’s social losses due to these health effects. The difference of the damages for each scenario is the social benefit of the mitigation measures.

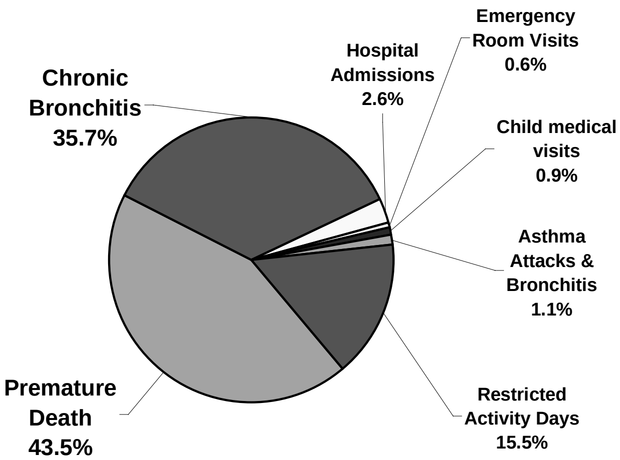
Table 8. **Social benefits for 2010 and 2020 (Millions of 1997US$)**

	2010		2020	
Endpoint	**mid**	**90% CI**	**mid**	**90% CI**
Premature Deaths	53.0	(15.1 - 371.3)	210.6	(60.3 - 1,494.0)
Chronic Bronchitis	41.8	(26.8 - 67.3)	168.4	(106.8 - 265.8)
Hospital Admissions	3.2	(2.6 - 3.9)	12.8	(10.4 - 15.7)
Emergency Room Visits	0.7	(0.4 - 1.1)	2.9	(1.6 - 4.6)
Child Medical Visits	1.1	(0.5 - 2.0)	4.3	(1.9 - 7.9)
Asthma Attacks & Bronchitis	1.3	(0.5 - 2.4)	5.3	(2.2 - 9.5)
Restricted Activity Days	18.4	(14.4 - 23.9)	74.0	(56.4 - 94.9)
Total	**119.6**		**478.2**	

Note: $PM_{2.5}$ concentration changes estimated using source apportionment method, equation (6).

Where do these benefits come from? The next figure shows the share of the social benefits for each effect. It is clear that the biggest share of the benefits comes from avoided premature mortality, although chronic bronchitis cases also have an important contribution in the mid value case. Premature mortality dominates the values for the upper bound of the confidence interval, representing around 70% of the benefits, mainly due to the consideration of long-term exposure deaths estimates in that case.

Figure 7. **Share of the present value of benefits for each type of effect (mid estimates)**



Note: Based on mid estimates using source apportionment method.

All the previous results have been obtained the using source apportionment model to estimate the change in $PM_{2.5}$ concentrations. The next table shows the net present value of the benefits, computed using a real discount rate of 12% (the rate used in Chile for evaluation of all social projects) for the two models for computing the changes in $PM_{2.5}$ concentrations.

Table 9. **Present value of social benefits for each method of emissions impacts estimation (Million of 1997US$)**

Method for estimating PM2.5 concentrations	mid	90% CI	
Source apportionment	710	314	2,472
Box Model	417	194	1,376

Finally, another way to look at these results is to compute the average social benefit accrued from the reduction of each ton of carbon. This is obtained by simply dividing the benefits by the equivalent carbon reductions in each year.

Table 10. **Average social benefit per ton of carbon (1997US$/tonC)**

	Atmospheric Model		
Year	**Source appmt**	**Box Model**	**Avg of two models**
2010	90 (42 - 337)	48 (21 - 190)	69 (21 - 337)
2020	129 (60 - 479)	79 (39 - 284)	104 (39 - 479)

10. Discussion

This work is a preliminary estimation of the potential ancillary benefits of greenhouse mitigation in Chile. We have conducted an aggregate analysis for the whole country, based on previously developed base (BAU) and mitigation (CP) scenarios.

The results show potentially high ancillary social benefits. The implementation of the CP scenario may prevent 2,800 deaths in the period 2000 to 2020, with a range from 1,500 up to 10,800. The mid estimate rests on generally accepted concentration-response coefficients, and which are in agreement with studies conducted in Santiago, Chile's capital, which accounts for most of the exposure to particulate matter. The upper bound of the confidence interval relies heavily on the mortality estimates from prospective studies performed in the U.S., under different conditions than in Chile, so their application is more uncertain.

From an economic standpoint, the potential ancillary benefits represent a substantial fraction of the potential costs of the mitigating options. For 2010, the benefits per ton of carbon abated range from 21 up to 337 dollars, depending on the models used to estimate the impact of emissions on concentrations. For 2020, the values range from 39 to 479 dollars per ton of carbon abated. The magnitude of these values is comparable to current estimates of abatement costs of carbon, for modest mitigation scenarios. Therefore, these ancillary benefits may offset a significant fraction of the costs needed to implement the measures. In the specific case studied here, in which all the measures considered do not impose a cost on the user, these ancillary benefits indicate a net benefit for society.

However, it is necessary to stress the limitations of the analysis. The main one is that it has been conducted at an aggregate level for the whole country, with no consideration of local conditions, like emissions, meteorology or population density surrounding the sources. Therefore, our estimates are average estimates across all these dimensions. Several factors can influence the analysis, making the impact of the emissions vary widely. Consideration of these factors is crucial to estimate the ancillary benefits associated with specific mitigation measures.

The modelling of atmospheric concentration reductions of $PM_{2.5}$ as a consequence of reductions in precursors emissions is a key link in the analysis. Our two approximate methods show results that differ in about 50%. Unfortunately, development of a comprehensive atmospheric model, was outside the scope of this project, and without considerable work may not offer results much better than those of the aggregated models.

Finally, the transference of the unit values from a developed country to a developing one implies some strong assumptions. Until results derived locally became available, this will probably be the weakest part of the analysis.

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