COMPARING THE NET BENEFITS OF INCENTIVE BASED AND COMMAND AND CONTROL REGULATIONS IN A DEVELOPING CONTEXT: THE CASE OF SANTIAGO, CHILE[1]

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Abstract

There are numerous studies that establish the magnitude of the static efficiency gains made possible through the use of a cost effective ambient permit system (APS) compared to command and control (CAC) or other suboptimal instruments such as an emission permit system (EPS). However the cost effectiveness of APS rests both on the efficiency gains related to equalizing marginal costs of reduction and a lower degree of required control. As a result of this latter factor, CAC and EPS generally impose concentration reductions higher than required by the target air quality standard and also by APS. In developing contexts, as a result of high levels of pollution and only recent introduction of control policies, health benefits of reducing pollution significantly can be expected to be high whereas the costs may still be relatively low. Consequently the excess reductions may produce net benefits -benefits of improved air quality minus compliance costs-. This paper evaluates for Santiago whether reduced concentrations below the level of the standard as a result of suboptimal policies result in health improvements that produce greater net benefits than incentive based approaches. The results show that considering uniform air quality targets and for the range of technologically plausible control options in Santiago, suboptimal CAC and EPS policies result in higher net benefits than APS.

Key Words: Cost benefits analysis, cost effectiveness, air pollution, economic instruments,

JEL Classification: Q25

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1. INTRODUCTION

There are numerous studies based on simulation models which establish the magnitude of the static efficiency gains made possible through the use of marketable permits for fixed sources compared to command and control (CAC) instruments[2]. In a developing country context, O'Ryan (1996), established the cost reductions from using ambient based tradeable permits (APS). These studies suggest that in most cases the cost reductions of APS over CAC are substantial. An important caveat however is that many of these reductions are the result of lower emission reduction requirements under an optimal APS, i.e. APS, while complying with a given air quality standard in all receptors, allows more emissions than other instruments in non binding receptors. The cost effectiveness of APS rests then both on the efficiency gains related to equalizing marginal costs of reduction – a true efficiency gain – and a lower degree of required control (Tietenberg(1985)).

As many have pointed out (see for example Tietenberg (1985) and Oates et al. (1989)), if there is no value assigned to this overcontrol[3] CAC instruments will not improve at all on incentive based approaches and will indeed be more expensive. If "however, reduced concentrations below the level of the standards bring with them improvements in health or the environment, CAC approaches will produce greater benefits than incentive based approaches" (Oates et al. (1989), p.1233) . Consequently the comparison among instruments without correcting for these benefits is unfair and may be misleading. Considering the fact that most of the time regulators propose air quality standards that are uniform and not socially optimal, it may be the case that suboptimal CAC policies will result in higher net benefits (benefits of improved air quality minus compliance costs).

To overcome this problem there are two approaches. One is to eliminate the "lower degree of required control component" by imposing on all instruments the compliance with the same air

[2] See for example: Atkinson and Lewis (1974), Hahn and Noll (1982), Krupnick (1986), Mc Gartland and Oates (1984), Portney(1990), Seskin, Anderson,and Reid (1983 and Spofford and Paulsen(1988).
[3] The problem is that cost effective approaches implicitly assign a shadow price of zero to improvements that exceed the standard .

quality in all receptor locations, as is done in O'Ryan (1996)[4]. A second is to determine the net benefits, i.e., the difference between the costs and benefits, for each instrument, allowing a more complete comparison.

This paper evaluates the net benefits associated to the use of market-based incentives (MBIs) and CAC policies to control TSP emissions from fixed point sources[5] in Santiago, Chile. The authors are aware of only one paper that undertakes this comparison for the US (Oates et. al (1989)) and none for developing contexts. The results shed light as to the importance of including both costs and benefits in the comparison of different regulatory instruments, in particular in developing contexts where costs of reduction are not too high yet because little control effort has been undertaken, and correspondingly health benefits of improving air quality are high. The second section presents an overview of the air pollution problem in Santiago. The third section addresses the compliance costs of reaching given air quality targets using MBIs and CAC instruments. Using a linear programming model, the total costs of achieving a desired air quality standard have been established for each policy. The fourth section presents the population based health benefits associated to each instrument. Section five compares the net benefits of applying APS and two second best policies. The last section presents the main conclusions and future research lines.

2. The Air Pollution Problem in Santiago.

The city of Santiago, Chile, like many large cities in developing countries, suffers critical air pollution problems. In particular, in winter the concentrations of total suspended particulates (TSP) and smaller PM-10 particulates constantly exceed the established ambient standards. Significant adverse health effects on the city's 5.2 million inhabitants, associated with the high levels of pollution by PM10 in winter, have been established (Ostro et. al (1996), Ostro et. al (1999)). Additionally, ozone pollution is significant in spring and summer. The city's policy-makers have been struggling since the beginning of the nineties to improve air quality, objective that would require reducing emissions to approximately half the current levels to

[4] In practice this is done by running a model with a command and control instrument, and using the vector of air quality obtained as the targets to be met by APS.
[5] Point sources are sources that emit a gas flow greater than 1000 m^3N per hour, through a chimney.

reach the desired air quality targets[6]. Although they have been successful in halting the increase in pollution that could have been expected due to the significant increase in economic activity in the city during the decade, air quality is still far from the target as can be seen in the following figure.

To examine the spatial configuration of emissions from fixed point sources in Santiago, the city can be divided into a 34 x 34 km grid. The grid comprises 289 (2 x 2 km) cells which contain the relevant sources of the air pollution problem in Santiago, as well as most of the exposed population. In this area there is a total of 1098 fixed point sources. Total TSP emissions in the city reached **2.55** tons/day in 1998 . Figure 2 presents average daily PM10 emissions, measured in kilograms per day, from each cell in the grid, for 1999. Clearly, polluting sources are clustered in a few specific zones. The cell with highest emissions is located in the northwestern part of the city and emits 594 Kg/day, 22 % of the total PM10 emitted in the city[7]. Of the 289 cells of the grid, only seven are highly polluting[8], and the fourteen most polluting cells emit 65% of total emissions. These emissions spread out smoothly to the rest of the city affecting air quality in each cell. Figure 3 presents the corresponding concentrations levels, measured in ($\mu g/m^3$) in each cell. The daily PM10 standard is 150 ($\mu g/m^3$), that cannot be exceeded more than once in a year. The USEPA annual standard of 50 ($\mu g/m^3$) which will become an official standard in Chile by year 2012 can be used for purposes of comparison.

[6] Set, as is usual, based on health effects, not cost-benefit analysis.
[7] In this cell there is a power station with both natural gas and diesel powered generators.
[8] Emit more than 3% of total emissions.

Figure 1: Air Quality in Santiago 1989-1999, PM-10.

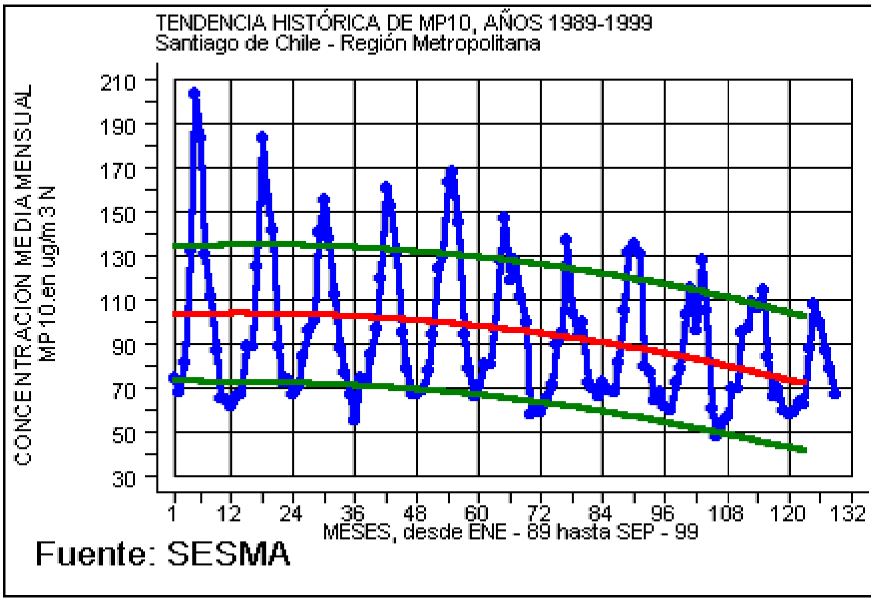


Figure 2: Baseline emissions of PM10

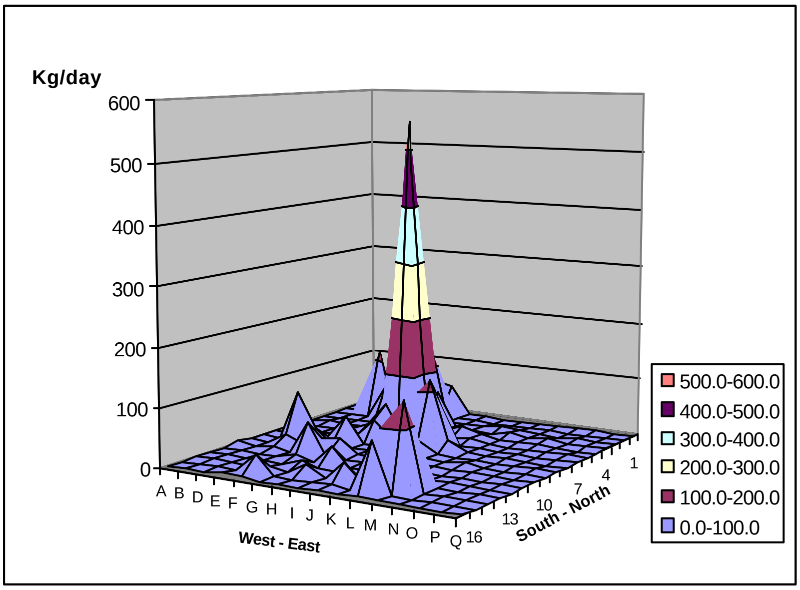
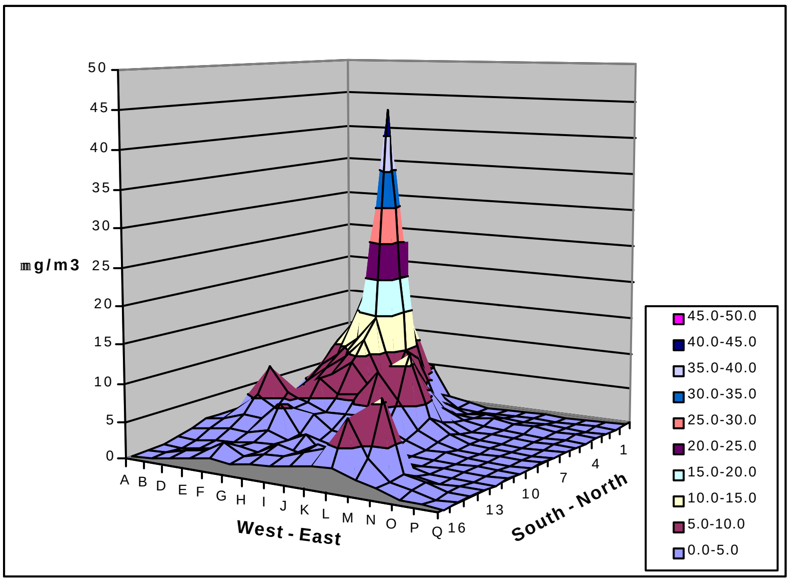


Figure 3: Baseline concentration of PM10



The main polluting sources of PM10 are summarized in Table 1. Industrial processess are the main emission sources, corresponding to 55 % of total emissions, while industrial boilers account for 40% the total. Heaters are also significant contributors (5%). The main emitting fuels are natural gas[9] and particles emitted by industrial processes using electricity[10]. Diesel oil is also an important source of emissions.

Table 1 Percentage of PM-10 Emissions by Source Category and Fuel

	Electricity	Coal	Wood	Fuel 6	Fuel 5	Diesel	Natural Gas	Total
Industrial processes	30,64%	1,13%	0,04%	1,03%	0,42%	3,65%	18,24%	55,1%
Boilers	0,00%	0,65%	0,57%	0,45%	5,70%	15,07%	17,53%	40,0%
Heaters	0,00%	0,00%	0,02%	0,00%	1,70%	2,12%	0,89%	4,7%
Bakeries	0,00%	0,00%	0,00%	0,00%	0,00%	0,13%	0,02%	0,16%
TOTAL	**30.64%**	**1.78%**	**0.63%**	**7.48%**	**12.82%**	**22.97%**	**36.68%**	**99.24%**

Source: Bravo (2000).

The main instruments used to control air pollution have been command and control: emission standards for industry, homes and new cars, as well as the elimination of highly polluting buses and increasingly tighter emission standards for diesel motors. For fixed point sources, these standards allowed reducing emissions from 9 tons per day in 1994, to approximately 2.6 tons in 1998[11].

There have been lukewarm attempts to introduce flexibility for fixed point sources through the introduction of an offset system for particulates. The system set an emission standard for all existing point sources and allows them to trade any excess reductions. New fixed point sources (entering the city after 1992), are required to offset all their emissions and recently are being required to offset 120% . Trades are undertaken on a one to one basis (i.e. it is an emission permit system, EPS) even though it is recognized that particulates behave as a non-uniformly mixed pollutant.

[9] There is a mega-source that generates electricity within the city limits, that uses natural gas and emits by itself, almost 11% of total PM10 emissions in the city.

[10] The emissions are not generated by the energy source in this case, but are due to the characteristics of the process (smelters for example).

[11] These numbers must be taken as a reference only. In particular the figure for 1994 is an estimation that is not based on actual measurements of emissions.

The offset system has not worked as expected[12]. There have been very few trades most of them internal to each firm, and as a result the market has not fully developed. Additionally, the introduction of natural gas as a new fuel in the city in 1997, has reduced the costs of decreasing emissions dramatically for many sources, to the point where it is economically convenient for many to switch to this clean fuel that allows them to comply with the emission standard. Some regulators are concerned that allowing one-to-one trades can result in hot spots.

Finally, in the last two years, the industrial sector has become increasingly vocal about the need to stop pressing for more restrictions to fixed sources, arguing that they have reduced their contribution to air pollution significantly since the beginning of the nineties, while other sectors have not.

Consequently the question addressed in this paper is given that there is a national air quality standard established not by cost benefit considerations, is there an advantage, from a cost-benefit perspective, of using marketable permits, in particular an ambient permit system (APS), in Santiago? Or is it appropriate in the current situation to use simpler EPS or even command and control instruments?

3. Compliance Costs of Improving Air Quality Under Different Policies

This section presents the model developed to evaluate compliance costs and the main results related to the costs of applying MBIs and CAC policies in Santiago .

The Problem

From a cost-effectiveness perspective, the regulator's problem is to obtain the desired air quality in each receptor location at a minimum cost. Considering there are n polluting sources, K zones in which emissions are generated, and the same number of receptor

[12] The main reason for this have been implementation failures. For a performance evaluation of the Offset system in Santiago see Montero, J.P. et. al (2002).

locations (say localized at the center of each zone), the problem to be solved is the following:

$$\min \sum_{i=1}^{n} c^i (e^i) \qquad (1)$$

subject to

$$Q = f(e^1,...,e^n) \qquad (2)$$

$$Q \leq Q^* \qquad (3)$$

$$e^i \geq 0 \quad i = 1, ..., n \qquad (4)$$

where,

e^i = emissions by source i after the policy is applied;

$c^i(e^i)$ = compliance cost function for source i reducing to e^i;

Q = a vector of ambient concentration at the K receptor locations;

$f(e^1,...,e^n)$ = a natural systems model relating emission levels by all sources to ambient concentrations at each receptor;

Q^* = a vector of desired ambient concentration levels.

To solve this problem, it is necessary to know both $c^i(e^i)$ and $f(e^1,...,e^n)$, i.e. how abatement costs relate to different emission levels for each process, and the natural systems model relating the vector of concentrations to emissions from all sources.

Abatement Costs and Transfer Coefficients

The cost of abatement for each source depends on the control alternatives that are applicable. Based on the literature (Bretschneider and Kurfurst (1987), Vatavuk (1990a, 1990b), Aranda (1996), and Bravo (2000)) and expert opinions, two categories of abatement alternatives were identified for the main processes in Santiago: (i) collection devices such as cyclones, multicyclones, bag filters and wet scrubbers; and (ii), for some sources, a change of fuel. To estimate the costs of collection devices, the net discounted cash flow of total capital investments and net annual operating costs incurred each year over the useful life of the equipment were estimated. To estimate the present value of switching to cleaner fuels, the cost of transformation and the cost differential associated with using a different fuel were estimated. Different sized control devices were costed. As a result, analytical cost relations were established for each control alternative (see appendix 1). These costs depend on the size of the source (gas flow) and hours of operation. Each control option was also assigned the abatement efficiency presented in the following table, based on expert opinion.

Table 2: Control Efficiencies of Abatement Options for Main Processes in Santiago (in Percentage)

Process	Cyclones	Multicyclones	Bag Filters	Venturi Scrubbers	Electrostatic Precipitators	Switch to Fuel Oil 5	Switch to Fuel Oil 6	Switch to Diesel	Switch to gas
Coal Heaters and Boilers	33	70	99	96	97	65	80	94	99
Wood Heaters and Boilers	15	50	99	89	95	63	79	94	99
Fuel Oil 6 Heaters and Boilers	5	30	99	86	94	-	43	83	97
Fuel Oil 5 Heaters and Boilers	5	30	99	86	94	-	-	69	94
Diesel Heaters and Boilers	5	30	99	86	94	-	-	-	86
Coal Furnace	40	82	99	98	99	-	-	-	-
Wood Furnace	15	50	99	89	95	-	-	-	-
Fuel Oil 6 Furnace	5	30	99	86	94	-	43	83	97
Fuel Oil 5 Furnace	5	30	99	86	94	-	-	69	94
Diesel Furnace	5	30	99	86	94	-	-	-	88
Al, Cu, Br Foundries	31	80	99	95	90	-	-	-	-
Stone and Grain Crushing	41	85	99	98	91	-	-	-	-
Stone and Grain Cleaning	35	60	99	98	98	-	-	-	-
Asphalt Plants	40	89	99	97	94	-	-	-	-

Source: Aranda (1996), page33.

To relate concentrations to emissions, the natural systems model is substituted by an environmental "transfer" coefficient, α_{ik}, relating changes in emissions by source i to changes in concentration at receptor k. To obtain these coefficients a simplified "cell" dispersion model, available for Santiago, was used. The wind fields had to be averaged over the day for this, and meteorological conditions which reflected episode conditions had to be selected[13]. Twenty two episode days were used and the corresponding transfer coefficients averaged. As a result, the transfer coefficients obtained reflect the impact of a unit of emissions on concentration levels in each cell of the grid, for adverse meteorological conditions. The results, which were surprising because it was previously thought (by whom?) that the main impacts were in a different direction, are presented in Appendix 2.

The Policies Evaluated

With information on emissions, location of each source, costs of abatement for each individual source and the transfer coefficients, the overall costs of two MBI policies and one CAC policy were evaluated. The policies considered for this exercise were:

(i) The spatially-differentiated ambient permit system (APS). Under this policy, permits defined in units of concentration at each receptor, are distributed so as to achieve the desired air quality goal at each receptor which corresponds to the air quality standard valid nationwide.

(ii) A marketable emission permit system (EPS). Under EPS, total allowable emissions from fixed sources in the airshed are established. Permits, equivalent to these emissions, are distributed to polluters, who can then buy and sell the permits from any part of the city on a one-to-one basis.

(iii) A uniform concentration standard for all sources (STD). All point sources are required to emit at concentrations lower than a single concentration standard.

[13] Episode days are those in which the air quality standard is exceeded.

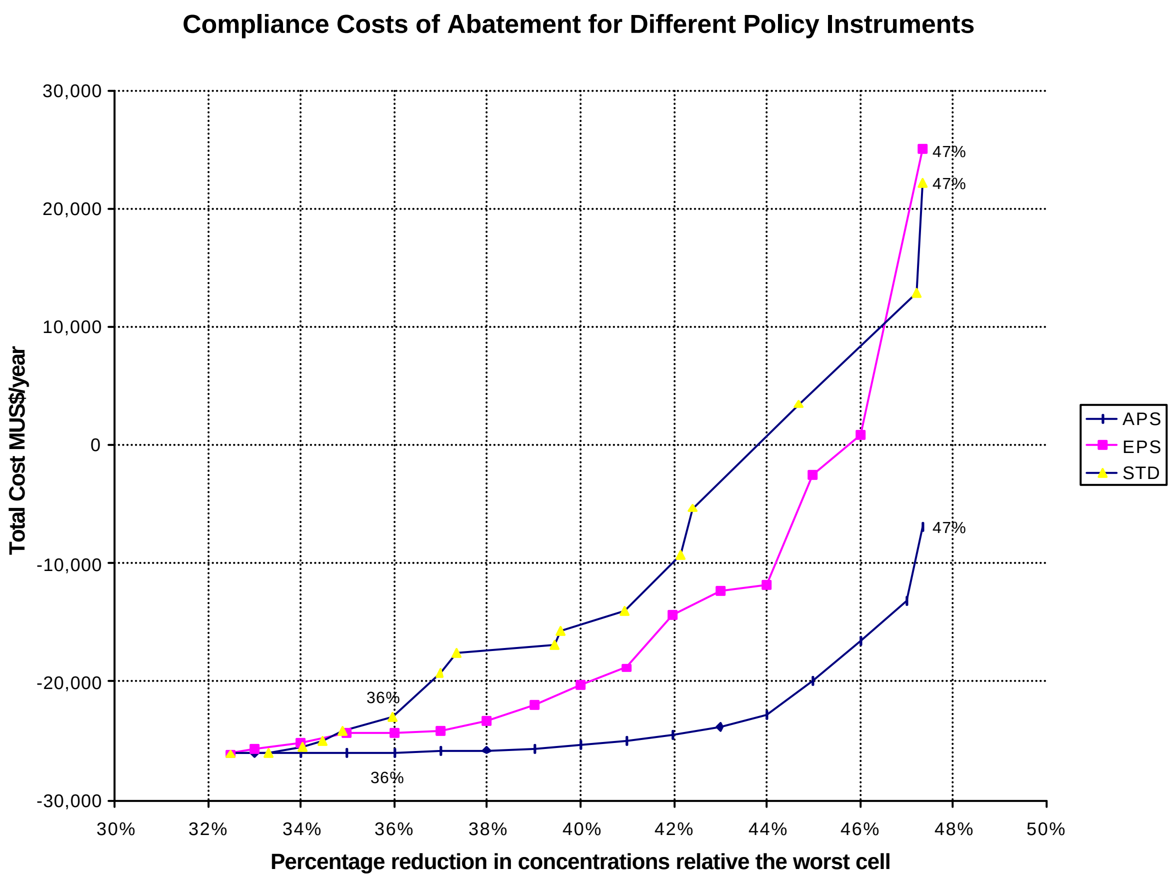
To compare the costs of different policies, the most widely used criterion are the compliance costs, under each policy, of meeting a uniform concentration standard at all receptor locations in the city (each cell of the grid in this case). Any policy must at least reach this standard everywhere. This is the success criterion used in this section. Allowed concentrations ranging from as low as 5% of current concentration in the cell with highest concentration , to as high as 47% were evaluated. Reductions higher than 47% are not possible in the worst receptor location without reducing activity of some sources or closing them down, option that has not been considered in this study.

Compliance Costs Under Different Policies

The compliance costs and reductions in allowed concentrations at the worst receptor location(s), under each of the three policies, resulting from different expenditure levels by fixed sources in Santiago are presented in Figure 4. The recent introduction of natural gas as a control option clearly imposes some surprising results. First, many sources that have not switched to natural gas could do so and actually obtain net benefits, i.e. their total production costs would fall. The explanation for this is that at current prices, many sources using diesel and fuel oil for combustion processes, can switch to natural gas and produce the same amount of energy at a lower cost. This lower operating cost (discounted over twenty years) covers the required transformation cost, and the source actually obtains net benefits. Consequently 32% of current emissions could be reduced if all those firms that can switch with net benefits, do so[14].

[14] Not all the sources had immediate access to the natural gas since time passed between the arrival of the natural gas in 1997 and the development of the distribution network.

Figure 4:



Source: Bravo (2000).

Note: $460= 1US$ (1998).

Second, as expected APS is clearly the most cost-effective instrument. Moreover, over the whole range of reduction, total costs are negative[15] for this instrument. A fully operating APS policy would take full advantage of the win-win situation discussed before, obtaining significant net benefits. The maximum 47% reduction can be obtained with yearly net benefits of approximately US$13 million. The suboptimal spatially undifferentiated EPS system on the other hand would impose a significant cost of over US$28 million per year to

[15] Obviously marginal costs are not however. They increase steadily over the reduction range.

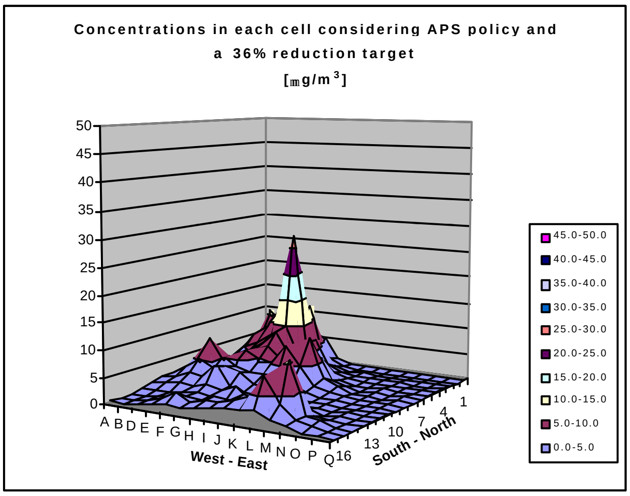
reach a maximum level of 47% of the current concentration at the worst receptor location. STD can only impose a 41% reduction at the worst receptor location, because it is assumed that the authority can only impose undifferentiated standards. Within the range of reduction of this instrument, it is clearly more costly than EPS[16].

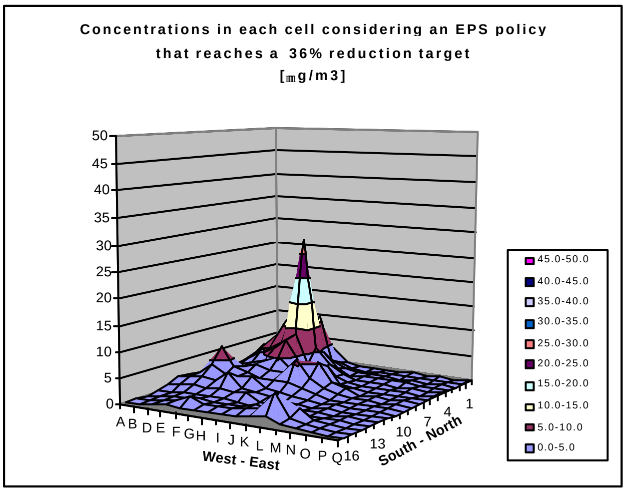
Air Quality In Each Receptor Location

Even if APS is the most cost effective policy, and both EPS and STD are much more costly, much of the cost reduction is *not* related to gains in efficiency but to the lower degree of required control component. Figure 5 presents the concentration reached in every cell with each instrument when a maximum concentration reduction of 36% with respect to the worse cell is imposed. As expected, as a result of APS, concentrations are higher than with both EPS and STD. For example, considering a cut all along the J cell presented in figure 6, it can be observed that APS allows higher concentrations in all cells relative to the other two instruments. EPS imposes the highest reductions.

[16] It must be borne in mind that negative total costs does not mean that all sources gain switching to gas. Actually, many individual sources have to incur in high costs to reduce emissions.

Figure 5: Air Quality Resulting from A 36% Reduction Target Considering Different Policy Instruments





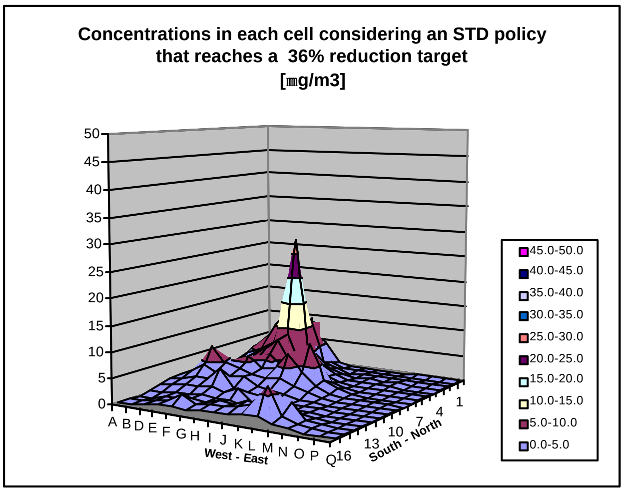
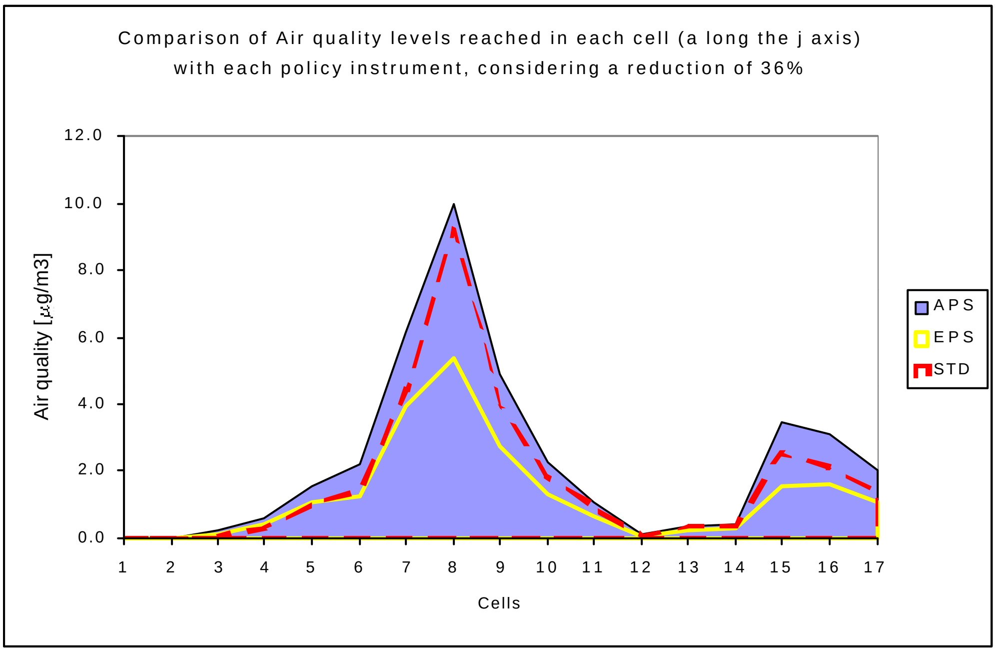


Figure 6



As a consequence, the health benefits from applying each instrument should be different, since the concentration reductions are different. These benefits are evaluated in the following section.

4. The Health Benefits of Improved Air Quality.

In this section we present the value of the population weighted health effects of improving air quality with each of the three instruments discussed before. The estimation of the benefits associated with air quality improvements has considerable difficulties due to fact that clean air is a non-market good and hence one for which there is no information available on prices and/or quantities traded in order to estimate a proper monetary measure for welfare changes. In this paper we followed the *indirect method or dose – response function approach* ,a methodology frequently used in environmental benefit estimation and specially to estimate health related benefits (see for example [Externe (1999)]). The methodology involves three stages:

Stage 1 : The impact on concentrations that result from the application of each policy instrument has to be estimated. The change of concentrations *with* and *without* the project is given by:

$$\Delta q = q(\mathbf{s}^{\mathbf{1}}) - q(\mathbf{s}^{0})$$

where Δq is the reduction in the concentrations of pollutants (for example, PM10) and the vectors $\mathbf{s}^{\mathbf{1}}$ and $\mathbf{s}^{0}$ correspond to the environmental policies to be evaluated ($\mathbf{s}^{\mathbf{1}}$) and those in place in the original situation ($\mathbf{s}^{0}$). The function q(.), corresponds, in general, to a pollutant dispersion model that predicts the behavior of concentrations as a result of the change in emissions that result from the policies included in **s** over the different sources. These results are presented in the previous chapter.

Stage 2 : The effects that pollutant concentration reduction have on different health outcomes have to be estimated. The change in the health effects are quantified using dose – response functions for a set of health effects for which there are well established statistical relations in the environmental epidemiological literature. These dose-response functions are applied to the exposed population to determine the population weighted health effects. For this, the exposed population in each cell is considered.

Stage 3: Lastly, the health effects found in stage 2 have to be valued in monetary units and they have to be summed up over the different effects, over the individuals exposed and over time since the benefits occur through time. In other words, the benefits associated with the environmental policy or project evaluated are estimated as the avoided costs due to the reduced cases of mortality and morbidity due to it. In this section we consider the population weighted health effects related to the average PM-10 concentration in each cell, resulting from each policy.

Dose-Response Functions.

The use of dose - response functions to estimate health effects can be described with the following equation in which the estimated impact in the health effect that is being analyzed (hospital admissions, mortality, etc.) is given by :[17]

[17] Ostro (1996).It is assumed that the effect is a lineal function of the dose, no matter the level of pollution .

(1) $$dH_i = b * POP_i * dA$$

where:

dH_i = change in the risk of the health effect i in the population under exposure.

b = slope of the dose-response function.

POP_i = population exposed and under risk of being affected by effect i

dA = change in atmospheric pollution for the pollutant under consideration.

Dose- response functions come from environmental epidemiology studies in which the aim is to try to estimate the health effects that can be attributed to atmospheric pollution after controlling for the effect of other variables on the probability of suffering health effects: like nutrition habits, smoking habits, temperature, time spent outdoors, supply of medical services, etc. Ideally the dose-response functions used for benefit estimation should be estimated with data generated forn the population and place where the project is going to be implemented . However, when these are not available, the functions are transfered from studies performed on other populations and locations and applied to the apecific case under study.

The use of dose-response functions is more appropriate to estimate acute health effects but not to capture chronic effects. This fact is reflected in the limited number of dose-response functions for chronic effects available in the literature even though there are good reasons to think that long term exposure to air pollution can cause chronic effects on human health. Therefore, if there are chronic effects in health as a result of long term exposure to atmospheric pollutants of the population of Santiago, the estimates obtained in this paper will be an underestimation of the true health effects attributed to air pollution.

In addition, it is worth noting that when dose - response functions estimated for other populations and places are transfered and used to estimate health effects associated with an environmental policy to be applied in Santiago, it is assumed that the function describes appropriately the relation existing between pollution and health in the area where the policy is to be implemented. This, in turn assumes that the base conditions in the area, like the general health status of the population, access to health facilities, dietary habits, time spent outdoors, and the chemical composition of the pollutants are similar to those prevalent in the are in which the original study was conducted. This is a strong assumption and constitutes the main criticism to the transfer of functions.

Another simplifying assumption is the linearity with which the dose-response function is applied, regardless of the level of the concentration of pollutants. If the true dose-response function is non-linear, then the effects could be larger at low concentration levels. For large changes in the pollution levels, in which the daily changes in concentration are strongly related to the annual changes, the linearity assumption is not a bad assumption.
The majority of the studies used in this paper, have estimated linear and log-linear models that imply a continuum of effects even at low concentration levels. This can be justified by the empirical fact that the studies that have tested for the existence of a threshold for effects associated to particulate matter have failed to find one. In addition, many recent epidemiological studies have found an association between particulate matter and health effects throughout the whole range of concentrations, even for levels under the USEPA primary air quality standards (Externe (1999)).

As a consequence, all the functions used in this study are applied linearly assuming that the slope of the dose response function is the same regardless of the concentration level.

Health Effects Considered

There is a large body of literature relating adverse health effects with ambient concentrations of PM10.The health effects for which there are well established dose – response functions are: Mortality, Hospital Admissions due to Respiratory Illness (CIE 460,480-486, 490-494,496)), Hospital Admissions due to Cardiovascular Illness (CIE 410, 413,427 y 428), Emergency Room Visits due to Respiratory Illness, Restricted Activity Days in Adults, Lower Respiratory Illness in Children, Chronic Bronchitis, Acute Respiratory Symptoms, Asthma Attacks.
Due to the uncertainty existing in the precise estimate of the dose-response parameter, it is possible to consider an interval of values to reflect a wider range of possible effects. However, for brevity, and because it does not affect the basic results of the paper, only the central value is reported here. The central value is typically obtained as the mean value

reported by the study or group of studies that have been selected as those that provide the most reliable results for the given health effects.

Monetary Valuation of Health Effects

To value mortality, there are three main approaches: *the contingent valuation approach, the wage differential approach and the human capital approach.* All of them are debatable and present limitations.[18] Since there are no reliable willingness to pay studies for reduced mortality in Chile, nor wage differentail studies, two alternatives are available. The first is to use the human capital approach, which is the simplest but also the least exact. However, its application gives a value for mortality extremely low, The second alternative, which was the one employed in this paper, was to use a values for a statistical life estimated from willingness to pay studies performed abroad after adjusting for the differences in GNP per cápita purcjasing power parity between the country where the estimation was performed (usually a developed country) and Chile. More specifically, the value used for mortality valuation was one used by EPA , which corresponds to the average for the value of the statistical life (VSL) from the 13 studies, selected by EPA , that report the lowest values for VSL. The values was deflated using the GNP per capita PPP estimated for 1999 by The World Bank. For morbidity, three alternative approaches are generally used: *direct costs of illness, defensive expenditures* and *contingent valuation*. In this paper we used the first approach since there are no willingness to pay studies available. [19]It considers the direct treatment costs plus the lost income as a measure of productivity loss during the episode of illness. This methodology has the advantage of being very simple but it has a number of limitations. The first is that it is a lower bound of the true willingness to pay for morbidity reductions due to the fact that it does not consider other costs such as pain and inconvenience. In addition it does not consider the fact that people can take a number of defensive actions.

Table 3 presents the unitary values for each health effect used for the monetary valuation in this paper.

[19] The values are estimated in (Holz and Sánchez (2000)).

Table 3

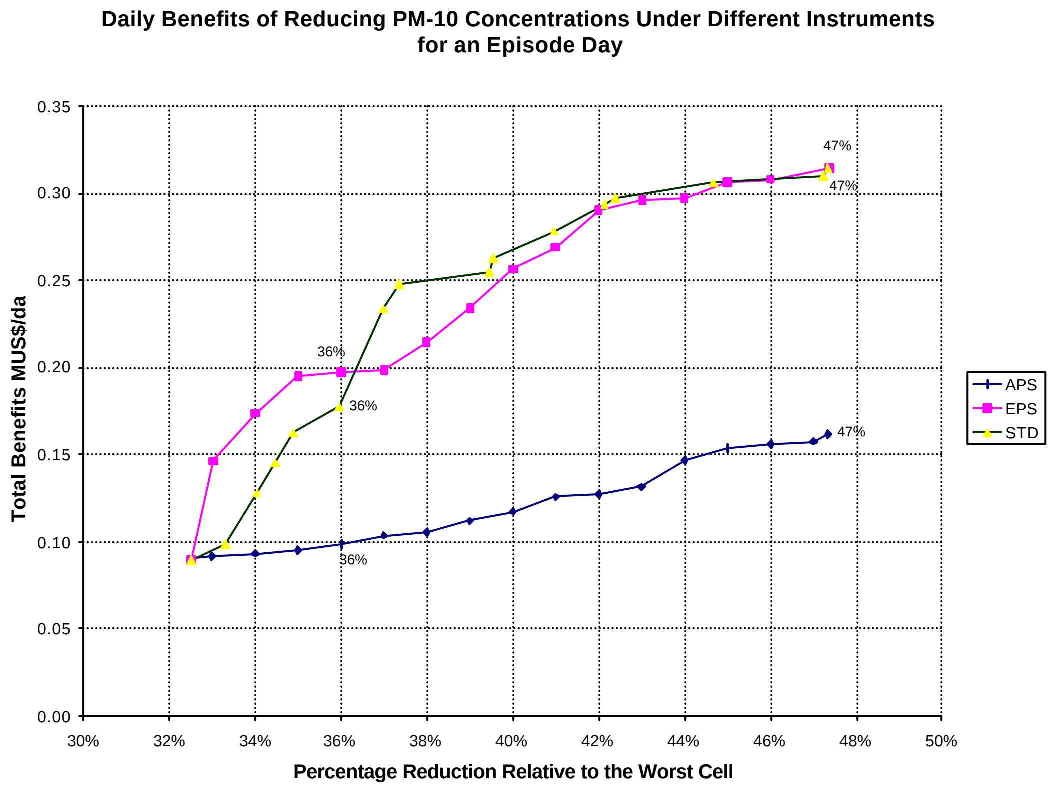
Unitary costs of Health Effects	
Health Effects Considered	Dollars of 1998
Mortality	713.514
Chronic bronquitis	142.988
Hospital admissions due to respiratory illness	1.669
Hospital admissions due to cardiovascular illness	3.534
Emergency room visits due to respiratory illness	77
Astmha Attacks	172
Lower Respiratory Illnes in Children	174
Acute Respiratory Sympthoms	9
Restricted activity days	16

Source: Holz and Sánchez (2000).

Health Benefits

Figure 7 presents the daily population weighted health benefits obtained from improving air quality with each instrument. As can be seen, each instrument results in very different health benefits. These differences are basically due to the fact that each policy imposes different reductions in each cell. All policies start out with health benefits of US$90 thousand per day, for a 32% reduction in the worst cell. However, considering a 47% reduction, APS results in approximately total benefits of US$162 thousand per day, whereas EPS has a total benefit of almost US$314 thousand per day. These results are very interesting since APS has the *lowest* benefits of the three instruments considered, almost 50% lower than EPS and STD for an important fraction of the reduction range!

Figure 7:



5. Comparing the Costs and the Benefits

In section 3 the annualized cost of each instrument was estimated for different reduction targets. To compare these costs with the benefits of improving air quality, requires an estimation of the yearly values of benefits is required. To do this, it is necessary to estimate the health benefits of reducing emissions for every day of the year. Unfortunately this is not as simple as multiplying the value obtained in the previous chapter by 365, since weather conditions are key to the dispersion of the emitted pollutants. The episode conditions considered to estimate the dispersion factors discussed in section 3, are obtained for the 28 worst days in terms of weather conditions for dispersion in winter. During other seasons, dispersion conditions are much more favorable and as a consequence, similar emissions result in significantly lower concentrations of PM-10. Health benefits from reducing emissions in summer months can thus be expected to be significantly lower than in winter.

For this reason, to determine the yearly benefits resulting from the application of each instrument, it is necessary to estimate the concentrations associated to the resulting emission levels in each season. To do this, the following simplified procedure is used. Jorquera (2000) has recently estimated a factor that represents average dispersion conditions for each month in Santiago at four different receptor locations. His results show that these factors do not vary much between locations, hence we have used the average results from the four locations. As can be seen in the table 4, average dispersion conditions in the worst winter month (June) are more than four times worse than in January,. To estimate the benefits of reducing emissions each day, it is assumed that these factors represent the average dispersion conditions each month relative to the episode conditions (that has a factor of 1). Consequently, the benefits in a day of November, for example, of reducing emissions will only be one fourth of those obtained in a day in June. As a result, total monthly benefits are obtained multiplying the daily health benefits of the corresponding month by the number of days in the month and by the relative dispersion factor. The yearly benefits are the sum of the benefits obtained for each month. These benefits are equivalent to the benefits of 182 winter days.

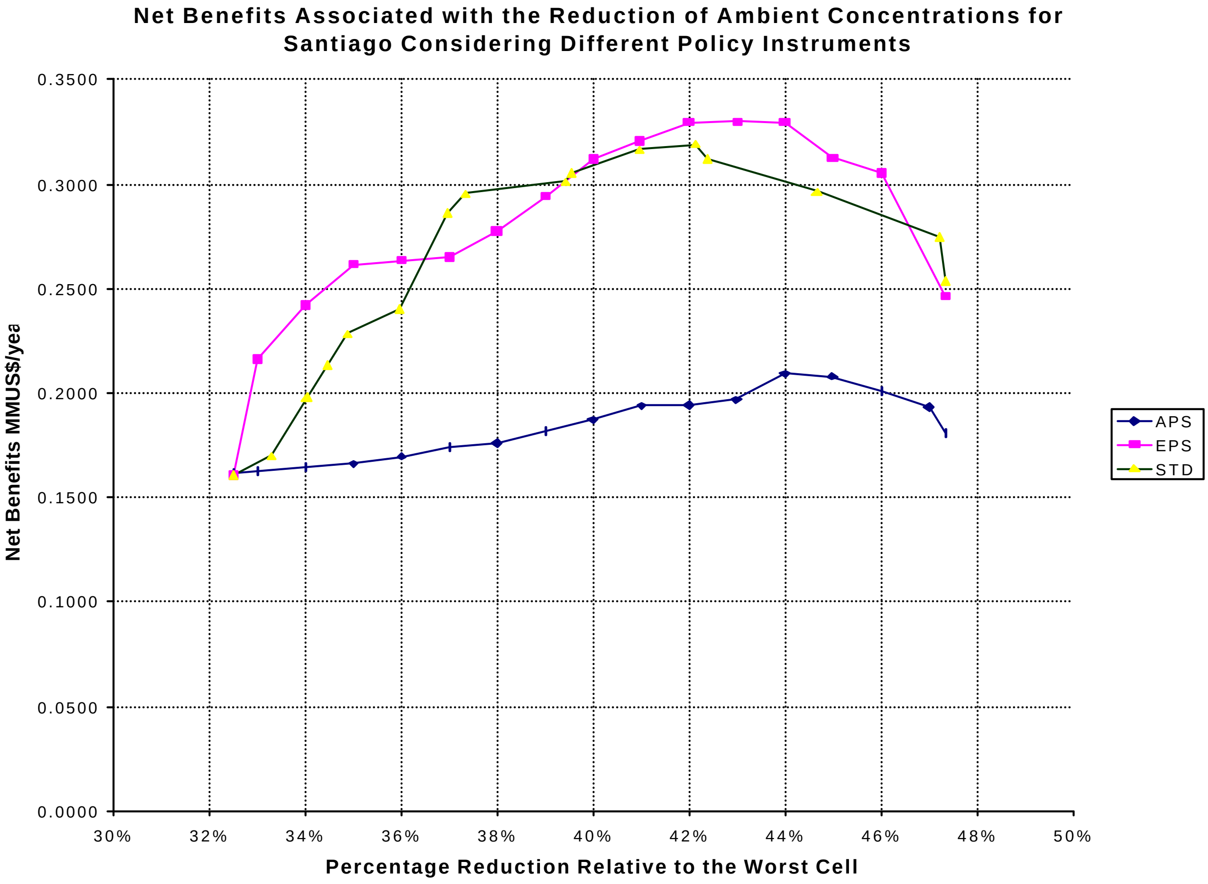
Table 4: Relative dispersion factors for each month

Month	Relative Dispersion Factor	Number of days
January	0,239	31
February	0,279	28
March	0,366	31
April	0,579	30
May	0,805	31
June	1,000	30
July	0,859	31
August	0,646	31
September	0,431	30
October	0,279	31
November	0,251	30
December	0,251	31

Source: Personal elaboration based on Jorquera (2002a and 2002b)

The yearly benefits obtained are in the order of the tens of millions of dollars per year, similar to the annualized costs of reducing emissions. Subtracting the costs of each policy instrument from the yearly benefits yields the net benefits to be expected from applying each instrument. This net benefit is presented in the following figure. The results are extremely interesting.

Figure 8:



The maximum net benefit is obtained for a reduction between 40 and 44% of concentrations at the worst receptor, *using an EPS policy*. These net benefits are approximately US$ 67 million per year. An emission concentration standard (STD) yields lower net benefits than EPS, but is still better than the cost-effective APS! In a developing country such as Chile, where little effort has been undertaken to reduce air pollution, the benefits of a better air quality associated to EPS and STD outweigh the relatively small efficiency gains of using a cost–effective instrument.

However, EPS is not better than STD for the whole range of possible reductions. For a 45% reduction both instruments have similar net benefits and for higher reductions STD is actually better. It is interesting to observe that between the optimal 40% reduction and a 47% reduction, net benefits fall approximately by 50 %. The implication is that the regulatory authority must determine the reduction targets very carefully so as to ensure that most of the net benefits are captured! A small difference of 10% in the required reduction target can imply very significant reductions in net benefits.

6. Conclusions and Future Research.

Correcting for the difference in benefits associated to each instrument makes a significant difference when choosing the policy instrument to be used. When only a cost effectiveness criterion is used, APS is clearly the preferred option, reducing costs significantly compared to EPS and STD, over a relevant range. However, when the benefits associated to the overcontrol of the latter two instruments, EPS has the highest net benefits, and APS is the instrument with lowest net benefits over a wide range of plausible reduction possibilities due to the fact that the air quality standard is fixed and uniform.

One of the main advantages of an APS plays against it in this context! Since it is able to impose reductions that very nearly reach the uniform standard in different parts of the city, it cannot take advantage of the significant health benefits from reducing concentrations more than required by the standard. This is precisely what occurs with the other two instruments. The efficiency gains of APS are much less significant than the economic losses due to health impacts resulting from the higher concentrations allowed. Of course, in principle it is possible to design an ambient permit system that exactly emulates the concentrations reached by the other two instruments, and in this case APS would definitely be the best option. However practical experience suggests that one cannot expect the regulator to set up such a system of differentiated standards within the city.

Since an EPS system is much simpler to implement, it can be suggested that a simple trading system that does not consider the spatial complexities of each trade be implemeted.

Additionally, the results shed light on the discussion about additional reduction requirements for fixed point sources. It has been shown that for Santiago, emissions from these sources can still be reduced by 40% with significant net benefits for society.

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Appendix 1: Control costs.

Table 1 Cost Function for Cyclones and Multicyclones ($ from 1998)

	Cyclones	**Multicyclones**
Annualized Investment Costs [$/year]	**62,34Q + 64.496**	**106,46Q + 113.809**
Direct Operational Costs [$/year]	**(0,113Q + 115,25)HRS**	**(0,113*Q + 115,25)HRS**
Indirect Operational Costs [$/year]	**503.729**	**503.729**

Table 2 Cost Function for Electrostatic Precipitators ($ from 1998)

Electrostatic Precipitators		
Annual Investment Cost [$/year]	**Direct Operational Cost [$/year/hour]**	**Indirect Operational Cost [$/year]**
$79.566Q^{0,6261}$	**(42,9Q – 357.671)HRS**	**4.336.985**

Table 3 Cost Function for Venturi Scrubbers ($ from 1998)

Venturi Scrubbers		
Annual Investment Cost [$/year]	**Annual Investment Cost [$/year]**	**Indirect Operational Cost [$/year]**
$116.407Q^{0,4037}$	**(110Q + 390.990)HRS**	**6.214.271**

Table 4 Cost Function for Bag Filters ($ from 1998)

Bag Filters		
Annual Investment Cost [$/year]	**Annual Investment Cost [$/year]**	**Indirect Operational Cost [$/year]**
$3.560Q^{0,7449}$	**(23Q + 386.720)HRS**	**6.912.527**

Appendix 2: Transfer Coefficients

Transfer Coefficients for Santiago

(values in percentage, referred to 100)

					6,16		
				5,39	12,71	5,93	
5,10	7,93	12,41	20,50	34,96	100,00	14,14	7,13
5,84	8,01	11,18	15,49	20,37	26,88	7,65	5,25
5,06	6,35	7,96	9,90	11,06	11,86	5,83	
		5,07	5,79	5,94	5,89		

Source: O'Ryan (1996).