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AGRICULTURAL AND HUMAN HEALTH IMPACTS OF CLIMATE POLICY IN CHINA: A GENERAL EQUILIBRIUM ANALYSIS WITH SPECIAL REFERENCE TO GUANGDONG

by

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PREFACE

Size can be a mixed blessing. China is now ranked among the largest economies in the world, with a GNP roughly on a par with Italy's and only slightly smaller than France's (at current exchange rates). The size of China's market, actual and potential, has proven a powerful magnet to foreign investment over the past few decades and the scale economies made possible by that domestic market bestow cost advantages on a number of Chinese industries in global markets.

Being big, however, can also mean emitting lots of pollution. When the pollutants in question are global pollutants, there is nowhere to hide. China's CO_2 emissions in the year 2000 were already as big as those of the entire European Union and close to those of the rest of Asia (including Japan) combined.

Thus, for Chinese policy makers, inaction on the climate change front is not an option. The challenge becomes one of devising policies that minimise any adverse welfare impacts from slowing greenhouse gas emissions growth. As the analysis in this technical paper demonstrates, it is possible to devise a climate policy for China that yields sizeable ancillary benefits in terms of health and agricultural productivity improvements, and up to a point these can offset the economic costs of moving to a less carbon-intensive growth path.

While the presence of ancillary benefits cannot be expected on its own to motivate climate policy, plausible estimates of their magnitude can be a useful input into the determination of an economically efficient and politically feasible level of abatement effort. Policy makers in both OECD countries and developing countries are beginning to factor these indirect benefits not just into their calculations of an appropriate level of carbon abatement but also into the formulation of policy mixes that aim to maximise the joint (global and local) environmental benefits from a given amount of mitigation expenditure.

This study has been produced as part of the Centre's research activity on "Responding to Local and Global Environmental Challenges" and in the context of ongoing involvement in the Organisation's horizontal work on Sustainable Development.

Jorge Braga de Macedo
President
OECD Development Centre
24 March 2003

RÉSUMÉ

La politique climatique de la Chine dans les prochaines décennies aura un impact majeur sur les initiatives visant à freiner le réchauffement de la planète. La progression des émissions de CO_2 a ralenti dans les années 90, mais il est trop tôt pour conclure aux prémisses d'une tendance durable à la baisse de l'intensité de l'utilisation de carbone dans l'économie chinoise.

Les décisions politiques en matière climatique doivent prendre en compte l'intégralité des coûts et bénéfices économiques d'un ralentissement de la croissance des émissions de gaz à effets de serre. Comme pour d'autres pays en développement, la préoccupation à moyen terme de la Chine est d'assurer la croissance économique tout en réduisant la pauvreté ; la politique climatique devra donc être à la fois efficace et cohérente avec cet objectif de développement.

Cette étude des effets sur la santé et la productivité agricole d'une taxe sur l'émission de dioxyde de carbone montre que les moyens d'action pour ralentir la croissance des émissions de gaz sans réduire la prospérité économique sont considérables. La diminution de la pollution locale a des effets significatifs pour la santé et les gains de bien-être résultant d'une amélioration de la productivité agricole sont au moins aussi importants. Si l'on considère les bénéfices sur l'agriculture et la santé provenant d'une taxe sur les émissions de gaz carbonique, la Chine pourrait réduire l'émission de CO_2 jusqu'à environ 15 pour cent par rapport à l'objectif fixé pour 2010, sans aucun regret.

La prise en compte des avantages agricoles a d'importantes conséquences en termes d'équité. En effet, les principaux bénéficiaires — que ce soit en termes de hausse des revenus et diminution des coûts d'alimentation — sont les ménages ruraux, en moyenne plus pauvres que leurs voisins urbains. Ces derniers bénéficient en outre de la moindre pollution de l'air en ville. Les conséquences au niveau régional d'une taxe sur les émissions de gaz carbonique sont également prises en considération : notre analyse montre que les effets sur les termes des échanges au sein de la région peuvent être considérables, les zones importatrices d'énergie sous forme de charbon (par exemple Guandong) étant favorisées au détriment de celles qui exportent cette énergie.

Ces résultats dépendent des choix d'élasticités de substitution en ce qui concerne la production et également des règles appliquées à la redistribution des revenus provenant de la taxe sur l'émission de gaz carbonique. Si l'on suppose que le degré de flexibilité de l'économie chinoise ne va pas changer et donc que sa capacité à remplacer le charbon, fortement émetteur de CO_2, par d'autres sources d'énergie ne va pas progresser, les coûts de la politique climatique en termes de bien-être augmenteront de façon significative, toutes choses égales par ailleurs. Cependant, si l'on redistribue tous les revenus d'une taxe sur l'émission de gaz carbonique aux entreprises *via* une réduction de l'impôt sur les sociétés, on constate que les coûts supplémentaires sont largement couverts à l'horizon 2010, jusqu'à des taux de réduction de l'ordre de 20 pour cent des objectifs d'émission pour 2010. Néanmoins, une progression de la consommation est le revers d'une hausse de l'investissement des entreprises et la croissance du PIB.

SUMMARY

China's climate policy over the coming decades will be crucial to efforts to slow global warming. While CO_2 emissions growth slowed in the 1990s, it is too early to know if this represents the beginning of a long-term downward trend in the carbon intensity of China's economy.

Climate policymaking needs to consider the full range of economic costs and benefits of slowing greenhouse gas emissions growth. Like other developing countries, China's medium-term preoccupation is with ensuring poverty-reducing economic growth, so climate policy must be both effective and consistent with this developmental goal.

This study of health and agricultural productivity effects of a carbon tax shows that there is considerable scope for slowing emissions growth without diminishing economic welfare. The health benefits of reduced local pollution are significant, and the welfare gains from improved agricultural productivity are almost as large. When both health and agricultural benefits of a carbon tax are considered, China could reduce CO_2 emissions by around 15 per cent from their 2010 baseline level while incurring no regrets.

The inclusion of agricultural benefits has important implications for equity. For, the principal beneficiaries — in terms of both higher incomes and lower food expenditures — are rural households; on average, they are poorer than the urbanites who enjoy health benefits from breathing cleaner city air. The regional welfare implications of a carbon tax are also considered: the analysis shows that interregional terms-of-trade effects can be important, with carbon-energy-importing regions (in this case Guangdong) benefiting at the expense of exporting ones.

Results are sensitive to choice of substitution elasticities in production and also to the rule applied for carbon tax revenue recycling. If one makes conservative assumptions about the degree of flexibility of China's economy, hence its capacity to substitute away from carbon-intensive energy, welfare costs of climate policy are significantly increased, all else equal. However, by recycling all carbon tax revenue to enterprises through reduced corporate tax, these added costs can be more than offset by 2010 for rates of abatement up to around 20 per cent of baseline emissions. The price, however, of higher corporate investment and GDP growth is some foregone consumption in the early years.

I. INTRODUCTION

China is an indispensable participant in international efforts to address the problem of climate change. China is a signatory to the United Nations Framework Convention on Climate Change (UNFCCC) and its Kyoto Protocol. At the World Summit on Sustainable Development held in Johannesburg, South Africa, in August-September 2002, the Chinese government announced its intention to ratify the Kyoto Protocol. Qualifying as a non-Annex 1 party, China would not be bound in the initial control period (2008-12) to any quantitative restrictions on its GHG emissions. Its principal obligation is to monitor and report to the Conference of Parties on the status of GHG emission sources and sinks and of measures to dampen growth of net emissions in future. According to UNFCCC documents, China has not yet filed such a "national communication" but it is scheduled to do so in 2004. It is hoped that the current study would provide a timely information input into that exercise.

Next to the United States, China is the largest country source of greenhouse gas (GHG) emissions; in 1998, US carbon dioxide (CO_2) emissions from fuel combustion totalled 5.4 million tonnes to China's 2.9 million tonnes, or 24 per cent and 13 per cent of global emissions, respectively (IEA, 2000). China's economic growth continues to be among the fastest in the world: over the past two decades, its real per capita GNP rose an average 8.2 per cent per annum, compared with a world average of 1.1 per cent per annum (WDI, 2001). Rapid economic growth is expected to continue for many years (perhaps decades) and, even with substantial further progress in reducing the energy intensity of China's GDP, CO_2 emissions will probably also continue to rise[1].

Even with 20 years of breathtaking growth, China remains a relatively poor country, with 1999 GNP per capita of $780 at the current dollar exchange rate and $3 550 at the PPP exchange rate — the latter a mere 14 per cent of the high-income country average (WDI, 2001). Even another two decades of 8.2 per cent per annum growth would only raise China's PPP per capita income by 2020 to two-thirds the 1999 high-income country average.

China's national priority must remain sustained improvement in the living standards of its one-and-a-quarter billion people. The government is, nevertheless, acutely aware that rapid economic growth can bring with it severe local environmental degradation, with negative effects on health, resource (e.g. land) productivity, ecosystem integrity, recreational opportunities, and the general quality of life. China's industrial development is still largely powered by coal, with China the world's largest coal producer. The quality of its coal is low, and coal burning (whether for electricity, heating or cooking) is thus a major source of air pollution (both indoor and outdoor). Pollution has become a seemingly inescapable fact of life for many living in China's major cities.

Beginning in the mid-1990s, the central government has undertaken a number of initiatives aimed at mitigating the most serious environmental problems. Environmental investments have already risen as a share of GDP from 0.7 per cent in 1997 to 1.1 per cent in 2000 and are projected to rise further, to 1.7 per cent, by 2010 (EIU, 2001). Investments and other measures to control pollution have yielded some results. Median concentrations of sulphur dioxide (SO_2) in air declined in both large and medium-sized cities between 1991 and 1998. Declines in median total suspended particulate (TSP) concentrations, however, were not nearly as large and, in some large cities like Beijing and Tianjin, concentrations rose significantly over this period. Actual human exposure increased due to rising urban populations. Since the most severe health effects are thought to be associated with particulate exposure, air pollution remains a major health problem. Meanwhile, growing transport traffic has caused nitrogen oxide (NO_x) emissions to rise and, while ambient concentrations generally remain low, they are particularly high in Beijing, Shanghai and Guangzhou (exceeding the Class 2 National Ambient Air Quality Standard by a factor of 2 or more) (World Bank, 2001). The effect of NO_x and volatile organic compounds (VOC) — also emitted *inter alia* by the transport sector — on ambient ozone levels in surrounding rural areas poses a potential threat to agricultural productivity, as does regional particulate haze. In short, air pollution affects the health and well-being of both the urban and the rural population.

Fossil fuel combustion is not only the main source of local air pollution, but it is the main source of greenhouse gases, a global pollutant. While the implicated emissions are mostly different in the two cases, measures that alter the amount of different fuels consumed will normally affect both types of pollution simultaneously. For example, more efficient conversion of coal into thermal energy will lower, per unit of power generated, emissions of SO_2, TSP and CO_2. This raises the question of what scope there may be for policies that "kill two birds with one stone", improving local air quality while contributing to slowing the rate of global warming.

The current study seeks to delineate that scope. The starting point is an assumption that economic growth takes precedence at this stage of China's development, that progress continues to control local and regional air pollution, while action to control greenhouse gas emissions remains a lower-order but unavoidable concern. Given the economic and environmental baseline to 2010, the study asks what difference an explicit climate policy — modelled as a tax on carbon energy — would make to growth on the one hand and to local environmental quality and damages on the other. From the perspective of a Chinese government policymaker, the interesting question the study seeks to answer is: Supposing we were to introduce a modest energy tax to encourage a less carbon-intensive development path, how much of the economic cost could be offset by benefits that accrue from improved human health and agricultural productivity?

The paper is organised as follows. The next section presents summary information on the energy structure of the Chinese economy and indicators of the severity of air pollution. It also presents basic data on the health status of the population and agricultural productivity. Section III then describes the economic model that is used for the analysis, which is a two-region computable general equilibrium (CGE) model of the Chinese economy. Guangdong Province is modelled as a separate region to allow

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for higher resolution in the modelling of local air quality impacts of energy/climate policy. Section IV presents the details of the dispersion modelling and the assessment of health and crop impacts. Section V presents the results of the base run of the model to 2010, assuming no major climate policy initiative (but some progress on strengthening local air quality standards). Then, section VI analyses the effects of the climate policy experiment, while section VII presents results of sensitivity analysis on key parameters and variable assumptions. Section VIII compares results with those of other studies, both for China and for other countries, while section IX considers whether there is an efficiency tradeoff — or perhaps complementarity — between greenhouse gas (i.e. global) and local air pollution control measures. Section X concludes with a discussion of policy implications and points to areas for further research.

II. ECONOMY, ENERGY AND EMISSIONS IN CHINA AND GUANGDONG

II.1. China's Economic Structure and Performance

China's economy is 2.2 times larger than India's, producing a GDP of $990 billion in 1999. Its 1999 PPP per capita income is 1.6 times India's, or $3 620. The difference has been widening since 1987, when China overtook India. Over the 20 years, 1979-99, real per capita GDP grew in China by an average 8.2 per cent per annum, whereas in India it rose by only 3.7 per cent per annum (WDI, 2001). Even if one allows for the likelihood of overestimation of China's per capita GDP growth (e.g. Maddison, 1998, estimates it to have been 6 per cent per annum, 1978-95), the difference in economic performance between these two mega-countries is striking. Major economic reforms began a decade earlier in China, but even if one compares growth rates for the past decade, when both countries were undergoing reforms, the picture does not change.

Following an initial spurt in agricultural productivity from tenure reforms, industrial sector growth — much of it export-oriented — has underpinned China's remarkable performance. China's industrial value added grew between two and three times as fast as agricultural value added from 1979 to 1999; even then, its agricultural sector outperformed India's over this period, growing by 4.8 per cent a year to India's 3.7 per cent. Given these

Figure II.1. **Sectoral GDP Shares, China (inner) and India (outer)**, 2000
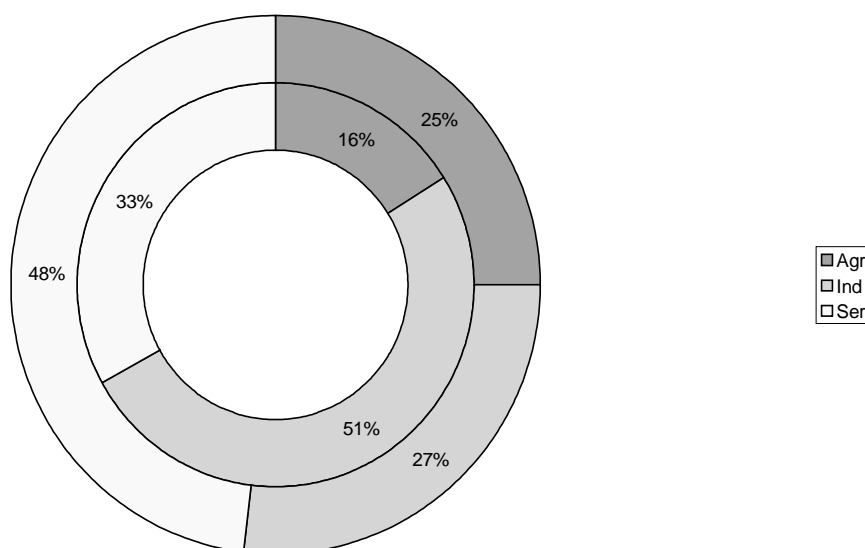
Source: WDI 2002 CD-ROM (World Bank).

trends, the two countries currently differ quite markedly in economic structure, with agriculture remaining a far larger share of GDP in India than in China and services also looming much larger there. Figure II.1 shows the sectoral GDP breakdown for 2000.

II.2. China's Energy Use and Its Implications for the Environment

With rapid GDP growth in China has come rapid urbanisation: the urban share of the population rose from just under one-fifth in 1979 to roughly one-third in 1999. With a growing number of people leading urban lifestyles, commercial energy consumption per capita increased by 35 per cent over this period and electricity consumption per capita trebled (WDI, 2001).

China is the world's largest coal producer (1999 production was more than quadruple India's), with much of that coal having a high sulphur[2] and ash content — and with only about 22 per cent being washed before burning (IEA, 1999). In 1999, coal accounted for 57 per cent of China's total primary energy supply and 77 per cent of its electricity generation (see Figure II.2). About 80 per cent of coal deposits are in the north and west, far from the major growth centres of demand and also far from water supplies for coal-washing at the pithead (EIU, 2001).

Having been a sizeable oil exporter in the 1980s, China became a net oil importer in 1993, and by 1999 its net imports were 47 million tonnes of oil equivalent (Mtoe) (60 Mtoe including Hong Kong, China) (IEA, 2001*b*).

Since 1998, the Chinese government has become more proactive in the development of the country's gas reserves and the promotion of gas as an alternative fuel in domestic heating, transport and electricity generation. This shift has been precipitated in large part by the country's serious air pollution problem. Greater reliance on gas is expected to have the additional benefits of pouring resources into the underdeveloped northwest interior of China and freeing up railway freight capacity which is currently stretched in meeting the needs of coal transport[3]. Even so, the contribution of gas to TPES in 1999 was a nugatory 2.4 per cent. The State Development Planning Commission projects that 6 per cent of the country's primary energy consumption will come from gas in 2010. Clearly, then, in the near to medium term, gas presents a rather limited alternative to coal and oil.

China also has considerable hydroelectric potential. In 1999, some 203.8 TWh of electricity were generated from hydro, representing 16.4 per cent of the total (IEA, 2001*b*). Substantial additional capacity is expected to come online in the next two decades. For example, by 2020, China will have completed a series of large hydropower stations along the Honghe and Lancang rivers, and at the upper reaches of the Yellow and the Yangtze, the last being the Three Gorges Dam[4]. Once all planned projects come onstream, hydropower capacity is expected to increase almost fourfold. According to EIA (2002) projections, by 2010 China's consumption of hydropower and other renewables is expected to almost double over the 1999 level.

Figure II.2. **China: Electricity Generated by Source**, 1999 shares
Total: 1 239 TWh
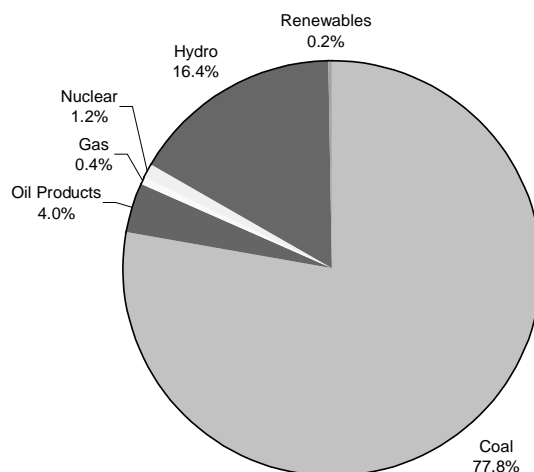
Source: based on IEA (2001*b*).

One of the most dramatic changes in China's energy demand structure over the past decade has been the rapid increase in the share of transport relative to industry. From 1989 to 1999, energy consumption in transport grew by 5.8 per cent per annum, whereas in industry it rose by only 0.7 per cent per annum (IEA, 2001*b*). Consequently, the transport sector's share of total final energy consumption rose over this period from 9 per cent to 14 per cent (industry still dominates with a 57 per cent share). Most significantly for urban air pollution, oil consumption in transport rose by 8 per cent per annum.

II.2.1. Trends in Air Quality

The air quality in China's major cities is poor. Air pollution has become a major health concern in China. In 1999, only one-third of China's 338 monitored cities were in compliance with national residential ambient air quality criteria[5]; indoor air pollution remains a serious health risk as well, with 80 per cent of the population using solid fuels such as coal, wood, and crop residue for cooking and space heating (World Bank, 2001).

Coal burning remains the major source of air pollution in China, but motor vehicle pollution is growing rapidly in major cities and is likely to become a much more serious problem over the next decade.

Since the mid-1990s, the emissions of major air pollutants appear to have been falling, with soot and fugitive dust — major components of total suspended particulates, or TSP — experiencing the steepest decline (38 per cent and 33 per cent respectively from 1995 to 1999) and sulphur dioxide (SO_2) experiencing a much more modest decline of around 10 per cent. Reduced coal consumption explains a part of the decline and stricter regulation a part (with soot and dust having been regulated before SO_2).

The emissions reductions have translated into improved air quality in a number of cities at least, with reported monitoring data for 60 medium and large cities showing TSP concentrations declining in 40 and SO_2 levels falling in 50 between 1991 and 1998 (World Bank, 2001). Despite the fall in TSP levels, they remained high in most urban areas (see Figure II.3): e.g. the median concentration in the 32 largest cities dropped only from 334 to 324 $\mu g/m^3$ over this period, while in the 28 smaller cities it declined from 260 to 215 $\mu g/m^3$. Nitrogen oxide (NO_x) levels rose in many instances, reflecting the rapid growth in vehicular emissions. While NO_x levels remain relatively low, in a few large cities — notably Beijing, Guangzhou and Shanghai — they exceed the Class 2 standard by a factor of 2 or more.

II.2.2. The Health Status of Population and the Effects of Pollution

The average life expectancy at birth in China was 70 years in 1999, up from 67 years a generation ago. Under-five mortality declined dramatically from 65 in 1980 to 37 in 1999, while the overall crude death rate rose slightly from 6.3 per 1 000 population to 7.2 (largely because of an ageing of the population: the proportion of the population 65 and above rose from 4.7 to 6.8 per cent over this period) (WDI, 2001).

In 2000, respiratory disease was the fourth largest cause of death in urban China, accounting for 13.4 per cent of deaths, while it is the largest cause in rural China, accounting for 23 per cent of deaths (NBSC, 2001). There is no more detailed breakdown by type of respiratory disease in this source, but certainly a portion of deaths are caused by acute respiratory infections such as pneumonia[6] that are not specifically attributable to air pollution. (Tuberculosis is listed separately.) WHO (1999) provides a somewhat better indication of the number of Chinese deaths that might be attributed to a combination of chronic exposure to pollution and smoking: an estimated 1 474 000 people died of chronic obstructive pulmonary disease (COPD) in 1998, accounting for roughly 15 per cent of all deaths in China and more than half of all estimated COPD deaths in WHO Member states.

Figure II.3. **Ambient TSP concentrations**

Chinese cities with 1 million or more inhabitants, 1995
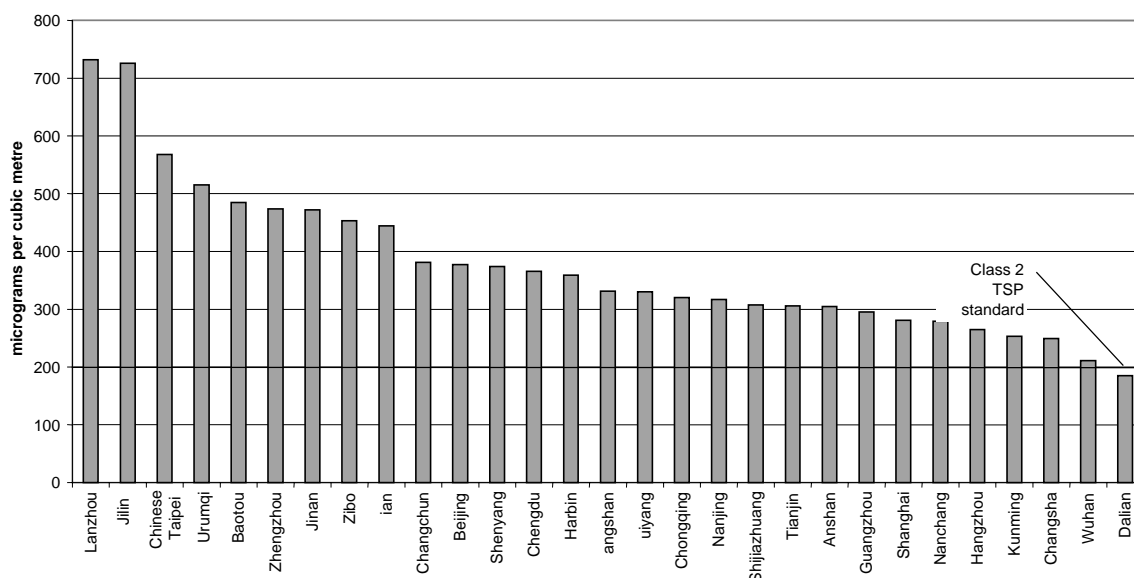
While age-specific mortality figures are not available from this WHO source, an earlier comprehensive study of the global burden of disease undertaken for the WHO by Murray and Lopez (1996) does contain such a breakdown. It shows, for China in 1990, that fully 90 per cent of COPD-related deaths occurred among those 60 years and older.

In the case of respiratory illness (morbidity), the incidence by age is somewhat less skewed towards the elderly. For instance, more than one in four cases of COPD in China (1990) occurred among the working-age population (15-59 years) (Murray and Lopez, 1996). If the same proportion holds in 1998 as in 1990, this suggests that some 4.8 million working-age Chinese suffered from COPD in the latter year (based on WHO, 1999).

Linking environmental conditions to health endpoints, whether illness or death, requires epidemiological studies. Numerous studies exist, but relatively few specifically for China. One such study of air pollution in Beijing (Xu and Dockery, 1994) finds a statistically significant relationship between day-to-day variations in concentrations of SO_2 and deaths from several types of disease. A doubling of SO_2 was found on average to increase deaths from COPD by 29 per cent, from pulmonary heart disease by 19 per cent, and from cardiovascular disease by 11 per cent. In the case of a TSP doubling, the result was found to be significant only for COPD-related mortality, which increased an average 38 per cent.

With respect to other health endpoints, Xu, Li and Huang (1995) have conducted a study of the incidence of air pollution on unscheduled hospital visits at a major Beijing hospital. They find that daily SO_2 levels were a significant predictor of internal medicine, pediatric, and emergency room visits, while daily TSP levels were a significant predictor of total outpatient visits and paediatric visits. The authors find a linear association between daily hospital outpatient visits and both SO_2 and TSP, controlling for other covariates.

Based on the epidemiological studies for China and — where appropriate — other locations, dose-response functions for major health endpoints are selected for use in the ancillary benefits/costs analysis (see Table IV.1 below, which reports not only the central slope estimate of the dose-response function but a confidence interval of ± 1 S.D. around the mean).

II.2.3. Agricultural Productivity

China's agricultural sector accounts for just under one-fifth of GDP and almost half of employment. China's agriculture differs from that of India in that the arable land area in China is a much smaller fraction (13 per cent) of total land area than in India (55 per cent). Even when one accounts for the fact that China's population density is less than half India's, this difference in arable land availability implies either that China's agriculture must be far more productive than India's or that China must depend far more than India on food imports. In short, China must feed a population 25 per cent larger than India's from an arable land area only three-quarters as large. Indeed, Chinese agriculture is more productive: in 1999, China's cereal yields per hectare were a little more than twice India's; the former's pulse yields were three times the latter's; in the case of roots and tubers, the difference in yields was only slight (with China's 7 per cent higher than India's (FAO, 1999).

Here the interest is in the nature and extent of the impact from China's air pollution on crop yields and, by implication, how measures to clean the air could affect agricultural productivity. The effects of tropospheric ozone exposure on plant tissues and crop yields[7] are well established, and the scientific literature is reviewed in USEPA (1996) and EC (1999: section 13.4). Ozone (O_3) is formed in the lower atmosphere as the result of a chemical reaction between various volatile organic compounds (VOCs) and nitrogen oxides (NO_x) in the presence of sunlight (Colls, 1997). Surface ozone levels are thus sensitive to pollution especially from fossil-fuel-burning power plants, industrial boilers, motor vehicle exhaust, gasoline retail outlets, and N-fertiliser induced soil emission of NO_x. Chameides et al. (1994) estimate that 10-35 per cent of the world's grain production occurs in locations where O_3 exposure may reduce crop yields, projecting that O_3 crop exposure could treble by 2025. Aunan, Berntsen and Seip (2000) use an ozone model for China (calibrated to 1990 data) and exposure-response functions from the literature[8] to estimate crop losses in 1990 and project them to 2020. The losses vary by ozone measure used, but in general the biggest predicted losses are for spring wheat and soybeans. For instance, according to the commonly used AOT40 measure[9], spring wheat production in 1990 suffered a shortfall of almost 10 per cent as a result of elevated ozone levels.

Besides ozone, other pollutants potentially affecting crop yields in China are SO_2 (Feng et al., 2002) and atmospheric aerosols, a complex chemical mixture of solid and liquid particles suspended in air and contributing to regional haze. Chameides et al. (1999) estimate the magnitude of the crop losses resulting from the effects of haze on the amount of sunlight (solar irradiance) reaching the earth's surface. Their research focuses on rice and winter wheat in Nanjing province, suggesting that in some of the most fertile eastern provinces of China current regional haze levels may be reducing optimal crop yields by between 5 and 30 per cent.

The Chameides *et al.* findings, however suggestive, are not incorporated in the present analysis and form a ripe topic for future research. One possible complication of their inclusion in an analysis that also considers ozone formation and crop yields is the possible non-additivity of effects. There are likely to be interactions between haze and ozone formation insofar as the latter involves a chemical reaction catalysed by sunlight whereas the former tends to diminish surface sunlight. Neither are the Feng *et al.* findings incorporated in the present analysis.

Figure II.4. **Agricultural Productivity in China**
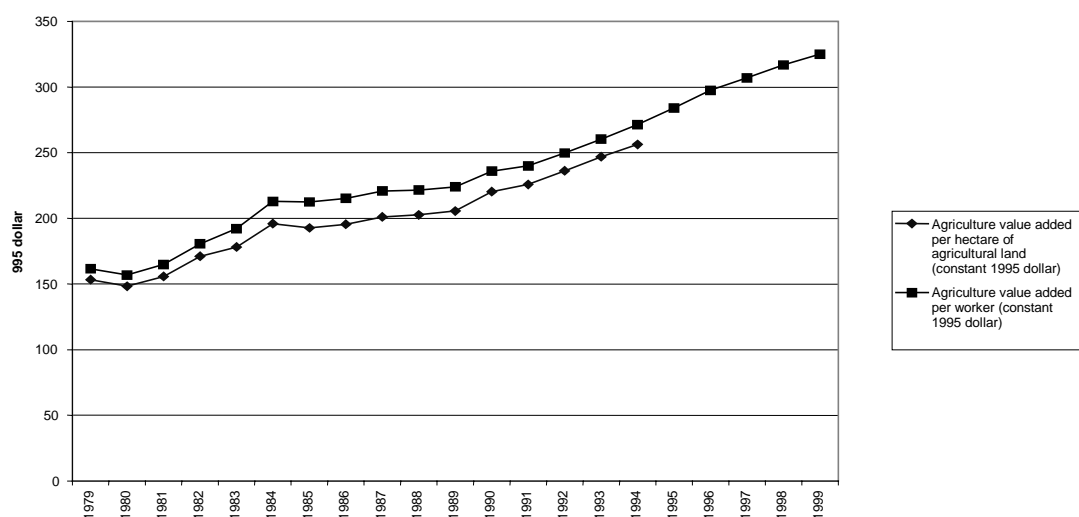
II.3. Why Guangdong?

Guangdong is located on China's prosperous, fast-growing south-eastern seaboard. The province accounted for approximately 10 per cent of China's GDP in 1997, while its population of 70.5 million (the fifth largest province) amounted to 5.7 per cent of the Chinese population. Clearly, then, Guangdong is one of the wealthier provinces, with a year 2000 per capita GDP 60 per cent higher than the national average. For the purposes of this study, Guangdong is particularly interesting because it is not only one of the most important emerging industrial centres of China, but it remains an important agricultural region in the heart of the Pearl River delta. In 1995, the province accounted for 7 per cent of total agricultural output value in China, 5.8 per cent in 2000 (*China Agricultural Yearbook 1996*; *China Statistical Yearbook 2001*). (Regarding specific crops, during 1995-97 the province contributed roughly 8 per cent of China's rice production, 7 per cent of peanuts, 6.5 per cent of tubers, and 6 per cent of vegetables; USDA 2001). Thus, if anywhere, one would expect to observe here noticeable spillover effects from urban, industrial pollution to peri-urban and rural agriculture.

Guangzhou, the capital city of Guangdong province, is an important urban centre, with a 1996 population of almost 4 million and a population density averaging 41 095 per km^2 in the core districts and 2 749 city-wide. This compares with Shanghai's 21 755 persons per km^2 in the core and 1 966 in the entire municipality

(Demographia.com). The city and the province are major industrial centres, with Guangdong accounting for roughly 15 per cent of gross industrial output value in 2000, the largest of any province in China[10].

As Figure II.3 shows, Guangzhou's particulate levels, while high, are not among the highest in China. Its average annual TSP concentration is roughly on a par with Shanghai's — i.e. about 50 per cent higher than the national Class 2 standard, and slightly below the median concentration for large cities. On the other hand, NO_x emissions and concentrations are high, largely by virtue of a rapidly expanding stock of motor vehicles on its roads. Indeed, in 1997, Guangzhou had the highest NO_x concentrations of any major city in China — $140\mu/m^3$ compared with $133\mu/m^3$ in Beijing and $105\mu/m^3$ in Shanghai, all twice or more the Class 2 standard. The high NO_x emissions are particularly relevant to an assessment of pollution's effects on crop yields in the province via ozone formation.

Besides being an interesting case — for the reasons just described — for the study of local benefits of climate policy, Guangdong also lends itself to detailed analysis because of the availability of good quality economic data with which to construct a provincial social accounting matrix (SAM), the extended input-output table that forms the basis for computable general equilibrium (CGE) models such as the one employed here.

III. THE ECONOMY-ENERGY-EMISSIONS MODEL

The economic model used here is a newly-developed two-region CGE model of China, Guangdong Province constituting one region and the Rest of China (ROC) the other. In effect, it seeks to marry the advantages of an economy-wide analysis to those of a more detailed "micro" analysis, in this case of a single provincial economy. An economy-wide approach is desirable in assessing climate policy since this is the domain of national policy makers. A carbon tax or a tradable carbon credit scheme — if ever they were to be implemented in China — would be implemented at the national level. Before committing to any such policy, decision makers would presumably like to see how the tax or trading scheme would affect overall national economic welfare, GDP growth, and broad economic sectors like energy-intensive industries. Only a "macro" model can provide this "big picture"; then the choice is between a CGE-type model and a macroeconometric model. For climate policy analysis, where relative price changes are likely to be a crucial part of the story, the former is preferable.

A national CGE model has important drawbacks, however, for the purpose of estimating the ancillary benefits/costs[11] of climate policy in terms of health and agricultural productivity. The main drawback is that these benefits are local, and they are closely linked to changes in local ambient pollution levels that are in turn a function primarily of changes in local air emissions. The mapping of emission changes into changes in ambient concentrations should ideally be done with as much geographic resolution as possible, using local meteorological data to calibrate an air dispersion model for a particular airshed (say a metropolitan area). Clearly, such a task is impossible to accomplish for a single geographical entity as large as China. By focusing on a single province for more detailed analysis, the task is made more manageable. Guangdong is impacted, as is every other province of China, by the national climate policy (in the simulations below, a carbon tax). The model permits calculation of both the provincial welfare loss and that for all China from the carbon tax. On the benefits side, with a relatively refined dispersion model for major air pollutants in Guangdong, one can estimate the provincial welfare gains from improved health and crop productivity associated with the tax and the resultant improvement in local air quality. While one can do the same for the rest of China, the ancillary benefits calculations are cruder than for Guangdong, given the coarser resolution in the modelling of the emissions-concentration-exposure links.

III.1. CGE Model Structure, Specifications and Calibration

The economic model is a two-region model, with Guangdong (GD) Province constituting one region and the rest of China (ROC) the other. There are 61 producing sectors in the most detailed elaboration of the social accounting matrix (SAM) plus the standard final demand sectors (household consumption, government expenditure, business investment and net exports). In the case of households, there is a four-way differentiation: rural/urban and by region (GD, ROC). Neoclassical assumptions hold regarding competitive-price-based market clearing. Because of the Armington assumption (Armington, 1969)[12], however, the law of one price does not hold across China: i.e. there can be two market-clearing prices for the same type of good, one in GD and the other in ROC.

III.1.1. Production and Factor Markets

All sectors are assumed to operate under constant returns to scale and cost optimisation. Production is modelled using nested constant elasticity of substitution (CES) functions. At the first level, output results from two composite goods: a composite of primary factors plus energy inputs, i.e. value-added plus the energy bundle, and aggregate non-energy intermediate input. At the second level, the split of the non-energy intermediate aggregate into intermediate demands is assumed to follow the Leontief specification, i.e. there is no substitution among non-energy intermediate inputs. Value-added plus energy component is decomposed into aggregate labour and energy-capital bundles. The energy-capital bundle is further decomposed into energy and capital-land bundles. Finally, the energy bundle is made up of four base fuel components, and capital-land is split into capital and land in the agricultural sector.

The model distinguishes between old and new capital goods. The distinction between old and new vintage capital allows the substitution elasticities in the production function to vary with vintage (see Appendix Figure A1, which shows the production nesting and the central elasticity values for old and new capital applied at each branch). The model also includes adjustment rigidities in capital market. While new capital is perfectly portable, old capital is not. If a sector is in decline, a part of the installed capital can be supplied to second-hand capital equipment markets. Thus, the end-period supply of old capital is the installed old capital net of any disinvested old capital. The supply curve of disinvested old capital is a constant elasticity function of the rental ratio of old to new capital, with that ratio constrained not to exceed one.

Labour is assumed to be perfectly mobile across sectors, and thus there is a single region-wide equilibrium wage rate. Capital is assumed to be only partially mobile across sectors, reflecting the limited portability of capital goods. Both labour and capital are imperfectly mobile across regions. In the basic policy scenario, zero interregional mobility is assumed. In the sensitivity analysis, this assumption is relaxed and CET functions are utilised to describe the regional movement of labour and capital. The movement of capital is determined by the relative rental rates and the constant elasticity

of transformation, and the movement of labour is determined by the relative wage and the constant elasticity of transformation. The share parameters in the CET function for capital movement, which represent the regional shares of capital stock with zero cross-regional mobility, are determined endogenously by regional saving and a pre-specified exogenous capital inflow trend.

III.1.2. Interregional and Foreign Trade

The rest of the world supplies imports and demands exports. Trade is modelled using the Armington assumption for import demand, and a constant elasticity of transformation (CET) for export supply. The small country assumption is applied for imports, hence world import prices are exogenous in foreign currency (an infinite price-elasticity). Exports are demanded according to constant-elasticity demand curves, the price elasticities of which are high but less than infinite.

Firms allocate their output between export and domestic sales to maximise profits, subject to imperfect transformation between the two alternatives. The domestic sales of firms are further split into local sales and interregional exports using CET functions. The CET elasticities for exports are assumed identical to the Armington elasticities for imports (3.0 in the first instance; 5.0 in the second).

Products are assumed to be differentiated by region of origin, and a nested CES aggregation function is specified for each Armington composite commodity. At the top level, consumers choose an optimal amount of each good; at the next level, they choose an optimal combination of domestically supplied and imported versions of the good, which is determined by relative prices and the degree of substitutability. At the final level of the nest, the domestic good is further split into a local good and an interregional import from rest of China.

III.1.3. Income Distribution and Demands

Factor income is distributed to four major institutions: enterprises, households, the government and the off-budget public sector. There are two types of household, urban and rural. Household income derives from capital, labour and land income. Additionally, households receive distributed enterprise profits, transfers from the government and rest of the world. Rural households earn all the land returns. Capital revenues are distributed among households and enterprises. Enterprise earnings equal a share of gross capital revenue minus corporate income taxes. A part of enterprise earnings is allocated to households as distributed profits based on fixed shares, which are the assumed shares of capital ownership by households. Retained earnings, i.e. corporate savings for new investment and capital replacement, equal after-tax enterprise income minus distributed profits.

Households are assumed to maximise their utility functions, with household demand modelled using the Extended Linear Expenditure System (ELES). Household disposable income is allocated to goods, services and savings, with savings evaluated using the consumer price index. Household savings are a function of disposable income.

The price and income elasticities are derived from the ELES parameters and the demand functions. Income elasticities differ by product, varying in the following ranges:

— for agricultural and food products: from 0.20 for rice to 0.88 for processed foods, 0.9 for beverages, and 1.0 for livestock products;

— for manufactures: from 0.9 for apparel to 1.35 for automobiles, with most manufactures having elasticities of 0.91;

— for services: from 1.01 for rural construction to 1.41 for urban telecommunications; energy services also have an income elasticity of 1.01.

Energy goods are initially grouped in one basket and then disaggregated by fuel type; in other words, in a first stage, consumers maximise utility over all non-energy goods and a single energy bundle. This allows the introduction of a single energy efficiency parameter, or autonomous energy efficiency improvement (AEEI) factor.

The other final demand accounts — government and business — assume a fixed-share expenditure function. Stock change is modelled as a demand for domestic products. The intermediate inputs, household consumption, and other final demands constitute the total demand for the same Armington composite of domestic products and goods imported from the rest of the world.

To determine the regional allocation of saving, there is assumed to be a national pool of domestic saving and foreign saving which is allocated to regions based on relative capital return rates, to maximise the investment return. The foreign saving for the whole country is exogenous in real terms, with net foreign savings (denominated in foreign currency) rising through 2000 before declining progressively to 2010; foreign savings for each region are endogenously adjusted in response to changes in region-specific real exchange rates.

III.1.4. Government Sector

The government collects taxes from producers, households and the foreign sector, transfers money to the household sector, and purchases public goods. Government derives revenues from direct corporate income taxes, household income tax, import tariffs, and various types of indirect taxes. Subsidies enter as negative receipts. The off-budget public sector, which consists of a variety of fees levied at different levels of government, on both households and enterprises, has been merged with the rest of the government sector.

III.1.5. Macro Closures

Macro closure determines the manner in which the following three accounts are brought into balance: *i)* the government budget; *ii)* aggregate savings and investment; and *iii)* the balance of payments. Real government saving is exogenous and depending on the budget closure rule, different taxes are chosen to be endogenous to maintain the budget balance at its base-year value. Aggregate investment is the endogenous sum of the separate saving components — household, corporate, government, and net foreign saving. This specification corresponds to the "neoclassical" macroeconomic closure in CGE literature.

The value of imports, at world prices, must equal the value of exports at border prices, i.e. inclusive of export taxes and subsidies, plus the sum of net transfers and factor payments and net capital inflows. An exchange rate is specified to convert world prices, e.g. in dollars, into domestic prices. In this model, foreign saving is set exogenously, the equilibrium would be achieved through changing the relative price of tradables to nontradables, or the real exchange rate.

III.1.6. Recursive Dynamics

The model has a simple recursive dynamic structure. Agents are myopic, basing decisions on current prices rather than expected future ones. Dynamics in the model originate from accumulation of productive factors and productivity changes. The base year of the data and the model is 1997. The model is solved for subsequent years in 2000, 2003, 2006 and 2010. The growth rate of population, labour force and labour productivity are exogenous. The growth of the capital stock is determined by the saving/investment relation. In the aggregate, the basic capital accumulation function equates the current capital stock to the depreciated stock inherited from the previous period plus gross investment. At the sectoral level, the specific accumulation functions may differ because the demand for (old and new) capital can be less than the depreciated stock of old capital. The producer decides the optimal way to divide production of total output across vintages. If sectoral demand exceeds what can be produced with the sectoral installed old capital, the producer will demand new capital. In the case of excess supply, the producer will disinvest some installed capital, as explained above.

In defining the reference simulation, a single economy-wide capital productivity is determined endogenously to achieve a pre-specified growth rate of real GDP. When alternative scenarios are simulated, the growth rate of capital productivity is exogenous, and the growth rate of real GDP becomes endogenous.

III.1.7. Model Calibration

The China model is calibrated to 1997 input-output data and provincial/national accounts data for Guangdong Province and all of China. The social accounting matrix (SAM) for the rest of China is constructed by differencing the SAMs for all China and Guangdong.

III.1.8. Welfare Measurement

There are two sorts of welfare change that need to be evaluated in the current context: those resulting from changes in prices of goods and services attendant on the introduction of a carbon tax; those resulting from changes in the health status of the population. The first are solved for endogenously by the CGE model; the latter are calculated exogenously, then added to the endogenous welfare change. In the policy scenario with crop-yield-effects, associated relative price changes are included, together with the direct price effects of the carbon tax. The health benefits consist of morbidity-related and mortality-related welfare changes. Their exogenous calculation requires the

imposition of separability conditions on individuals' utility functions, implying that the utility of reduced morbidity and mortality risk is independent of the consumption levels of various commodities (Boyd *et al.*, 1995). Also, it should be noted that changes in the health status of the population and in mortality are assumed not to have a significant effect on the supply of labour (this assumption could of course be relaxed).

The welfare change from a climate policy experiment consists of two parts, as shown in the following equation for period *t*:

$$\Delta W = (e(p^*, u^*) - e(p, u^*)) - (D^* - D) \qquad (1)$$

where *p* is the price system, *u* is utility, *e* the expenditure function, and *D* the monetary value of the changes in morbidity and mortality (*D*amages), and the star exponent denotes the *with-policy* state. The first term is the standard equivalent variation (*EV*) measure of welfare change, calculated endogenously by the model and interpreted — in the case of a welfare-reducing policy like a carbon tax — as the maximum amount the individual would be willing to pay to forego the change. As Freeman (1993:48) puts it, *EV* measures what change in income (given the original prices) would lead to the same utility change as the change in prices[13]. In the present analysis, both the direct welfare costs of a carbon tax and the indirect effects via changes in crop productivity are captured in the *EV* measure. The second term represents the summation over all affected individuals of their willingness to pay for the *ex post* reduction in morbidity and mortality resulting from the policy change. In short, how much is the improved health status worth to them? If, for example, a given climate policy results in 100 premature deaths averted, then those are evaluated at the "value of a statistical life" (*VSL*[14]). If premature mortality and incidence of morbidity go down as expected with the policy, the expression (*D* - D*) term is negative. Whether the overall welfare change of a given policy is positive or negative depends on the relative magnitudes of the first term (the welfare loss from the carbon tax) and the second term (the ancillary health benefits). Thus, the maximum "no regrets" abatement level of CO_2 is that where:

$$(e(p^*, u^*) - e(p, u^*)) = (D^* - D). \qquad (2)$$

Since in the policy scenario with crop effects, the left-hand term incorporates the welfare gains from increased crop yields, the magnitude of these ancillary benefits is given by the expression:

$$|(e(p^*, u^*) - e(p, u^*))_{\text{without crops}} - (e(p^*, u^*) - e(p, u^*))_{\text{with crops}}|.$$

III.2. Emissions

Air emissions are principally determined by intermediate or final consumption of polluting inputs, mostly fossil fuels. In addition, certain industries have emissions not directly linked to input consumption but related instead to their output levels (e.g. fugitive emissions, as with natural gas leakage and volatile organic compounds). Emission coefficients associated with each type of consumption and production are derived originally from the World Bank's IPPS project, which used toxic release inventory (TRI) data to establish sectoral pollution intensities for the United States (see Hettige *et al.*, 1995). Output-based emissions coefficients have one major drawback, viz. that — once fixed — they do not allow a given output to be produced with fewer emissions, e.g. by

using a different input mix. The only way to reduce emissions is to reduce output. In reality, for most air pollutants, there are three main ways of lowering sectoral emissions: in addition to reducing output, one can alter the input mix, e.g. consuming fewer polluting inputs, or capture the pollutants at the end of the stack through an abatement technology. The model effectively incorporates an abatement technology, in that it assumes a certain exogenous rate of reduction in the pollution-intensity of the economy over the simulation period (for another example of this approach, see Garbaccio *et al.*, 2000). The exogenous rate of pollution reduction implies that, up to a point, pollution abatement is costless. The model also allows for a variety of substitutions away from polluting inputs, e.g.: low-carbon fuels for high-carbon fuels; non-fossil fuel energy for fossil fuels; non-energy inputs for energy (e.g. installation of process control equipment); energy-conserving inputs for energy-using inputs; finally, imports of energy-intensive goods for domestic production of same (Jorgenson *et al.*, 2000). These methods of reducing pollution are not costless.

Dessus *et al.* (1994) transform the output-based pollution intensities of the IPPS data into input-based coefficients by regressing sectoral emissions on sectoral input use, with the unallocated portion of emissions (the residual) attributed to pure process emissions. Formally, the total amount of a given polluting emission takes the following form:

$$E = \sum_i \sum_j \alpha_j C_{i,j} + \sum_i \beta_i XP_i + \sum_j \alpha_j XA_j \tag{3}$$

where i is the sector index, j the consumed product index, C intermediate consumption, XP output, XA final consumption, α_j the emission volume associated with one unit consumption of product j and β_i the emission volume associated with one unit production of sector i. Thus, the first two elements of the right-hand side expression represent supply-related emissions, the third one final-demand-related emissions.

There are six primary air pollutants considered in the analysis of climate policy and its ancillary benefits: carbon dioxide (CO_2) — the main greenhouse gas, total suspended particulates (TSP), sulphur dioxide (SO_2), nitrogen oxides (NO_x), volatile organic compounds (VOCs), and carbon monoxide (CO). In addition, ozone formation is modelled as a function of primary emissions of NO_x, VOCs, and CO; given the dose-response relationships between ambient ozone and yields of various crops, it is possible to estimate approximate dose-response functions linking NO_x emissions directly to crop yields for each of the major crops grown in China — see details and caveats below.

Since the emission coefficients in the model are originally derived from US data, it was necessary to calibrate them to Chinese data by checking model-generated emissions against an emissions inventory contained in the *China Environmental Yearbook*. This was possible only for three major pollutants — TSP, SO_2, and NO_x — while CO_2 emission estimates are available from the IEA's annual *CO$_2$ Emissions from Fuel Combustion*. With respect to VOCs and CO, no independent check was possible.

IV. MODELLING DISPERSION AND IMPACTS

There are two sorts of dispersion model used in this study. The first relates to certain primary air pollutants (TSP, SO_2, and NO_x), while the second relates to secondary ozone formation. The three primary pollutants are implicated — by epidemiological and laboratory evidence — in a variety of human health impacts, including premature mortality (from chronic and acute exposure) and increased incidence of disease, notably various respiratory ailments. Ambient ozone is implicated in crop damage and yield reduction. Following is a description, first, of the modelling of the three health-linked pollutants, their dispersion in the atmosphere, human exposure and resultant impacts; then, of ozone formation and associated agricultural yield effects.

IV.1. The Health Component

Inevitably, in moving from climate policy simulations (for which an economy-wide CGE model poses no particular problems) to estimates of ancillary impacts, there is need for finer resolution in the model. Whereas the geographic location of GHG emissions is indifferent to their effect on climate, the geographic location of particulate emissions is decisive for their impact on human health. Addressing the ancillary effects therefore requires some reasonably reliable way of mapping sectoral emissions (for a given region of the CGE model — say, Guangdong) into impacts on the health of affected populations (in this case, mostly residents of Guangdong and, in particular, of its major city, Guangzhou)[15].

IV.1.1. A Stack-Height-Differentiated Dispersion Model

The approach followed here draws upon a simplified air dispersion model first elaborated in WHO (1989) and recently used in a number of other studies (notably Garbaccio *et al.*, 2000 for China, and Lvovsky *et al.,* 1999 for six major cities in developing and transition economies). The model differentiates emissions into three types, depending on whether they originate from high, low, or medium stacks. Each stack height yields a different dispersion pattern over the land area of a city, with its relative contribution to area-wide average ambient concentration depending on city size (as measured by radius). The general form of the dispersion equation (in this case for TSP) is as follows:

$$cn_{TSP} = b + d_1 (E_{High}) + d_2 (E_{Med}) + d_3 (E_{Low}), \tag{4}$$

where cn_{TSP} refers to measured area-wide concentration, b is a constant reflecting background emissions' contribution to concentrations, and the coefficients, d_i, are the

dispersion coefficients linking emissions, E, from each stack height to measured average concentration. "High" stacks are confined to electricity generation, "medium" stacks encompass most industrial sectors, while "low" stacks encompass construction, transport, other services, and households. The dispersion coefficients are calculated from equations specified in WHO (1989:23), using frequency factors borrowed from Lvovsky *et al.* (1999) and coefficients from the IIASA model reported in Table 3 of WHO (1989).

This type of dispersion model yields the following results: *i)* for low and medium height sources, the concentration/exposure per unit of emissions is strictly inversely related to the city's radius — in other words, the wider the area over which emissions are dispersed, the smaller their effect on average ambient concentration; *ii)* the emissions-exposure relationship for high-stack emissions traces an inverted-U in the city's radius: as high stacks spread more widely than low- or medium-stack emissions, so their contribution to area-average concentrations and exposure rises at first with city size; and *iii)* high-stack sources yield a concentration/exposure per unit of emissions very far below low-stack emissions for virtually any size of city and significantly below medium-stack emissions until city size approaches a radius of 30 km (in other words, a very large city). This suggests that the magnitude of any ancillary health benefits from changes in emission levels depends importantly on where (in which sectors) those changes occur.

To apply this dispersion model in conjunction with the CGE model requires some simplifying assumptions. First, it is assumed that all emissions generated by the economic model are concentrated in or around population centres, so the relevant radius to be used in the calculation of d_i is that of the major city (or cities) in the region. In the case of Guangdong, this means specifically the capital city, Guangzhou. For the "rest of China" region, the concentrations used are weighted averages of those in several major cities, where the weights used are population shares. This is meant to take account of actual human exposure to various concentrations. Thus, the average concentration used in calibrating the dispersion model is given by:

$$\overline{CONC} = \frac{1}{n}\sum w_i CONC_i,$$
(5)

where w_i are the population shares of each city in the total sample population ($\sum w_i = 1$).

The dispersion equation for "rest of China" has another peculiarity, viz. that the radius of the "city" is rather large, since the areas of all the cities in the sample are summed as though they constituted a single "mega-metropolis"[16]. This tends to assign relatively more weight to high-stack emissions than would be warranted if a separate dispersion function were calculated for each of the cities. For example, while the radius of Guangzhou is measured at 9 km, that for the "rest of China" composite urban area is 30 km.

Figure IV.1 shows the breakdown of TSP emissions by stack height for Guangdong and ROC in the base year, 1997. There is a striking difference in the relative contributions of low-stack and high-stack sources, with Guangdong showing a significantly higher share of TSP originating from high stacks than elsewhere. Partly this is because of the large weight of northern cities in TSP emissions for the rest of China,

and the importance of low-source emissions from household cooking and heating in those areas. Thus, even though high-stack sources are weighted rather heavily in the dispersion function for ROC, their low estimated share of total emissions reduces the significance of those heavy weights.

Figure IV.1.
TSP Emissions by Stack Height, Guangdong (inner) and Rest of China (outer)
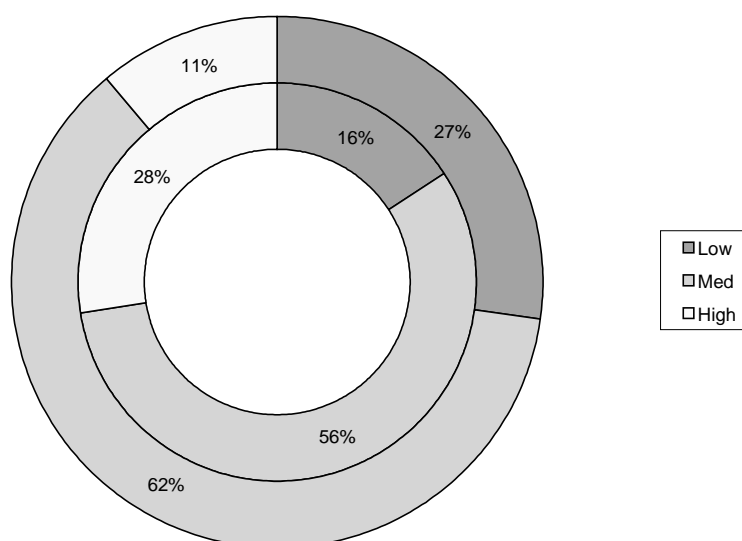
IV.1.2. The Dose-Response Relationships for Health Endpoints

The health damage assessment is mainly based on Chinese epidemiological studies on exposure-response relationships between air pollutants and health effects. Some health end-points not covered by Chinese studies were analysed by using studies carried out in Europe and the United States (e.g. for infant mortality, and some respiratory diseases). Most of the functions derived from these studies are rendered as so-called relative functions. This means that they give the percentage increase in the frequency of a given health effect per $\mu g/m^3$ increase of a given air pollutant indicator. Table IV.1 reports the dose-response relationships for various health endpoints, including an uncertainty interval. In this table, the impacts are given in absolute terms by combining the relative functions with the observed or estimated frequency of the health effect end-points in Guangzhou, yielding the absolute annual increase in cases per million inhabitants per $\mu g/m^3$. Note that the major health effects identified are caused by one or both of two pollutants — TSP and SO_2, with the only other one considered being respiratory problems associated with NO_2. While health effects are also known to be associated with CO and VOCs, the Chinese ambient air quality data do not cover these two pollutants, so it was not possible to incorporate their health impacts in the analysis. Xu (1998) reports results of several epidemiological studies of air pollution and health in urban areas of China that underlie the exposure-response parameter estimates used in the model, notably for TSP and SO_2.

Table IV.1. **Exposure-Response Functions**[a]

Change in annual number of cases per million people (all ages) per 1 $\mu g/m^3$ change
in ambient concentration (uncertainty intervals represent ±1 s.d.)

End-point	Pollutant	Period per case	Coefficient (uncertainty interval)
Deaths	PM_{10}		2.2 (0 – 4.1)
	SO_2		12 (9 – 15)
Infant deaths	PM_{10}		0.7 (0.4 – 0.9)
	SO_2		0.2 (-0.2 – 0.6)
Outpatient visits (OPV)	PM_{10}		4 670 (1 980 – 7 360)
	SO_2		1 800 (1 510 – 2 100)
Emergency room visits (ERV)	PM_{10}		55 (15 – 95)
	SO_2		186 (112 – 260)
Hospital admissions (HA)	PM_{10}	21 days	97 (65 – 121)
	SO_2	"	186 (89 – 302)
Respiratory hospital admission (RHA)	PM_{10}	14 days	56 (28 – 84)
(subgroup of HA)	SO_2	"	56 (28 – 84)
Hospital admission for COPD (HA-COPD)	PM_{10}	18 days	5 (0 – 9)
(subgroup of RHA)	SO_2	"	3 (0 – 5)
Work loss days (WLD)	PM_{10}		18 400 (9 200 – 27 600)
Acute respiratory symptoms in children[b]	PM_{10}	1 day	21 500 (14 190 – 32 470)
	SO_2	"	2 830 (2 690 – 2 970)
Acute respiratory symptoms in adults[b]	PM_{10}	"	28 320 (21 130 – 35 520)
	SO_2	"	7 650 (7 270 – 8 030)
	NO_2	"	1 260[c]
Chronic respiratory symptoms in children[b]	PM_{10}	~1 year	15 (13 – 18)
Chronic respiratory symptoms in adults[b]	PM_{10}	"	34 (29 – 39)
Asthma attacks	PM_{10}	1 day	1 770 (990 – 5 850)

a) Based on estimates for Guangzhou, the capital of Guangdong province.

b) The share of the population that is > or <14 yrs is incorporated in the functions that apply specifically to adults or children. Thus, the functions can be applied to the total population in an area.

c) The unit for change in ambient concentration is 1 pphm; no uncertainty interval given.

Source: Guangzhou Air Quality Action Plan 2001, February 2000, processed.

There is another set of health impacts that would ensue from a significant change in China's energy mix, *viz.* reduced occupational risks to coal miners. Annual deaths of miners have remained in the range of 5 000-6 000, despite official figures suggesting that the number of illegal mines was reduced by 70 per cent (to 23 000) from 1997 to 2002 (*Financial Times*, 17 July 2002, p. 4). In any event, illegal private mines are not the only ones where miner safety protections are lax; state-owned mines are responsible for a sizeable portion of accidents and resultant fatalities. While increased reliance on natural gas as an alternative to coal could create other safety concerns, it is doubtful that the risks — for example, of gas explosions — would compare in magnitude to those currently facing coal miners. Of course, the other side of the reduced mortality risk to coal miners is the loss of their jobs. While in the model this gets reflected in labour reallocation to other sectors, with some downward pressure on wages, in reality there could be significant localised unemployment in major coal-producing regions.

IV.1.3. *Valuation of Health Impacts*

Three sorts of health impacts must be valued: mortality risk, direct costs of illness, and productivity losses from days of missed work or reduced activity due to pollution-related illness. Valuing the first poses the biggest challenge, while the latter two can be valued in a rather straightforward manner — though, as will be explained, the simplest method of valuation is not necessarily the most satisfactory.

Beginning with the costs of illness, it is possible to assign an average value to various health services rendered to pollution victims, whether a hospital admission, an outpatient visit, or an emergency room visit. To the extent that a policy indirectly reduces the frequency of such events, one can determine the cost savings by multiplying unit costs by number of visits/admissions avoided. The loss of productive time from such visits can also be valued at the average productivity of labour. The value of savings from reduced pollution-related illness can then be directly incorporated into an economic model like the one used here — as reduced health-related expenditures and augmented labour supply, respectively. This approach misses at least one potentially important element of reduced morbidity, viz. the effect on psychological well-being of those whose health is improved. For this reason, it provides a lower-bound estimate on individuals' willingness to pay (WTP) to avoid illness. A more complete picture would incorporate this other, psychological dimension, which may itself have productivity consequences not captured by work loss days (WLD). To arrive at such a picture, one needs to use either indirect estimation methods (e.g. hedonic wage studies) or direct elicitation of *WTP* through surveys (e.g. contingent valuation). The former estimate from wage data the premium that must be paid to workers to assume additional risk of occupational illness or injury. The latter use statistical methods to derive from multiple survey responses a robust relationship between morbidity risk changes and *WTP*, controlling for other factors that may affect *WTP*.

To value the reduced risk of premature death from pollution exposure, there are few direct observable costs comparable to the cost of a hospital visit. The human-capital approach to valuing mortality risk is analogous to valuing earnings foregone from WLDs of illness. In this approach, one would value the human capital embodied in an average individual by calculating the discounted present value of expected future earnings over the years of life foreshortened by premature death. An obvious drawback is that zero value is assigned to years foregone of those who are no longer economically active, which includes those elderly individuals most vulnerable to pollution-related mortality.

For this and other reasons, the human-capital approach yields absolute minimum estimates of the so-called "value of a statistical life", or *VSL*. The two methods described briefly above — hedonic wage studies and contingent valuation — can also be used to estimate *VSL*. Indeed, hedonic wage studies are employed at least as commonly to evaluate this as to estimate *WTP* for reduced illness. The same caveats apply in both cases: *i)* that wage earners may not be a representative sample of those most at risk from pollution, viz., the elderly and infirm; *ii)* that pollution-related health risks are largely involuntary while job-related ones are largely voluntary (assuming workers are well-informed); *iii)* that there may be some bias in the *WTP* estimates, if generalised to the whole population, arising from the self-selection of risk-averse individuals into low-risk jobs.

Contingent valuation and other hypothetical approaches to estimation of *VSL* have their own shortcomings which have been extensively discussed in the literature (see Whittington, 1996, for a review of many issues in a developing country context) and much methodological refinement has gone into reducing risks of various biases.

The principal problem with valuing *VSL* and *WTP* more generally is that, for many developing countries, few if any country-specific valuation studies have been done. In that event, there is no choice but to transfer estimates from elsewhere, in which the manner of "benefits transfer" raises its own methodological problems. Probably the most difficult concerns how to adjust for differences in ability to pay across countries, as reflected in per capita income differences. Many studies simply apply as an adjustment factor the ratio of per capita income in target country to that in source country. This implicitly assumes that the income elasticity of *VSL* or *WTP* is equal to unity. Empirical estimates of this elasticity vary widely. Until recently, the few studies available suggested that the income elasticity of *VSL* is positive but less than one: in other words, survival is a necessity. In their benefits transfer study of air pollution in Central and Eastern Europe, Krupnick *et al.* (1996) assume an elasticity of 0.35 (based on contingent valuation studies reported in Mitchell and Carson 1986). Studies estimating *WTP* for pollution-related morbidity benefits also find an income elasticity less than unity: Loehman and De (1982) estimating a range from 0.26 to 0.60 for reduced respiratory symptoms from cleaner air in Tampa, Florida; Alberini *et al.* (1997) an elasticity of 0.45 for similar benefits in Chinese Taipei; Liu *et al.* (2000) an elasticity of a mother's *WTP* to prevent a cold of 0.3 (for her child) to 0.4 (for herself). On the other hand, Hammitt *et al.* (2000) find, from a longitudinal (16-year) compensating wage differential study for Chinese Taipei, that "survival is a luxury good", estimating an income elasticity of *VSL* of between 2 and 3. A meta-analysis of *VSL* studies by Bowland and Beghin (1998) yields a similar elasticity estimate.

China is a case of a country with very few studies of *VSL* or *WTP* for reduced mortality or morbidity risk. Moreover, the one study — Hammitt and Zhou (2000) — estimating *VSL* using a contingent valuation survey finds very low estimates — ones that are close to those generated by a simple human-capital approach (assumed to represent the lower bound of *VSL* estimates). While respondents placed considerably higher value on averting premature death than on averting chronic bronchitis and more in turn on the latter than on avoiding a cold, their *WTP* to avert mortality and chronic bronchitis risk was insensitive to the magnitude of the relative risk reduction, suggesting that "the respondents did not clearly understand the risk reductions they are asked to value, and cast(ing) some doubt on the results" (p. 7).

In the absence of any single "best" set of *VSL* estimates for China, a range of values is considered, bounded by the Hammitt and Zhou (2000) figure of $8 650 on the low end and by the World Bank (1997) estimate of $118 150 on the high end (both at 1997 prices and exchange rate). The central *VSL* values used in the basic policy scenario below are transferred from the Chinese Taipei hedonic wage study of Liu and Hammitt (1999) (adjusted to mainland Chinese income levels assuming an income elasticity of *WTP* equal to one). They are, for all China, $43 275, for Guangdong, $68 973, and for the rest of China, $41 851. In addition, central values for the various health endpoints considered in the ancillary benefits analysis are given in Table IV.2.

Table IV.2. **Estimated Monetary Values of Unit Changes in Various Health Endpoints**
(RBM, 1997 price)

Health Endpoint	Guangdong	Rest of China
Mortality	571 789	346 942
Infant mortality	571 789	346 942
Outpatient visits (cases) (OPV)	117.8	71.5
Emergency room visits (cases) (ERV)	117.8	71.5
Respiratory hospital admissions (RHA)	5 260	3 192
Work loss day (WLD)	24.3	14.8
Respiratory symptoms in children (ARS-Ch)	8.6	5.2
Respiratory symptoms in adults (ARS-Ad)	8.6	5.2
Chronic respiratory symptoms in adults (cases) (CRS)	70 808	42 964
Asthma attacks (ASM)	41.5	25.2

Sources: "Guangzhou Air Quality Action Plan 2001"; Liu and Hammitt (1999); authors' estimation.

IV.2. The Agriculture Component

This study considers only the impacts of regional ozone on agricultural productivity. As noted above, particulate haze and SO_2 have also been implicated in crop productivity losses, but those relationships are not modelled here. Tropospheric ozone (O_3) formation is the product of a photochemical reaction involving oxides of nitrogen (NO_x) and volatile organic compounds (VOCs), both of which are projected based on sectoral emission coefficients contained in the economic model. To simplify, O_3 levels are estimated as a linear function of NO_x only, and crop yields are then related to NO_x emissions rather than O_3 concentrations. An important caveat is that actual O_3 measurements from monitoring stations are not available for China.

IV.2.1. Linking Emissions to Ambient Concentrations: Modelling Surface Ozone

A global three-dimensional off-line photochemical tracer/transport model (OSLO-CTM2) of the troposphere was used to estimate surface ozone levels on a regional scale in China (see Appendix for a description of the model). The CTM2 can be run variably with resolution up to $1.87°\times1.87°$, however, in this study a horizontal resolution of $5.6°\times5.6°$ is used to limit the amount of CPU-time needed. Estimates of the surface ozone level in each province were made by means of a GIS (Geographic Information System) tool by overlaying the province borders and the grid cells in CTM2.

A reference simulation with CTM2 was carried out using the emissions given in Table A1 in the Appendix (equal to the estimated 2000 emissions used for the OxComp experiments described in IPCC (2001)). The NO_x emissions in Guangdong and all of China are 0.46 and 3.10 Tg(N)/yr, respectively, in the reference case (Guangdong was defined as the two grid cells covering 22-28°N and 107-118°E).

To study the effects on surface ozone of changes in NO_x emissions within Guangdong and in China as a whole, respectively, two sets of perturbation simulations were performed with CTM2 in which emissions were kept constant outside the target region. For both regions, simulations with reduced (-50 per cent) and enhanced (+100 per cent) NO_x emissions were performed.

To establish the NO_x-ozone elasticity in Guangdong, the following emission scenarios (emissions outside Guangdong are kept constant) are run:

— reference scenario, where the total NO_x emission in Guangdong is 0.460 Mt(N)/yr;

— reference x 0.5 scenario, where the total NO_x emission in Guangdong is 0.230 Mt(N)/yr;

— reference x 2 scenario, where the total NO_x emission in Guangdong is 0.920 Mt(N)/yr.

The increase in ozone from increasing NO_x emissions is estimated to be slightly higher in eastern Guangdong than in western Guangdong. The estimated linear function for the relationship between NO_x emission and the surface ozone metric *M7* (seasonal average 7 hrs d^{-1} for the period March-October, given in ppbv) for Guangdong is (*SE* in parentheses):

$$M7 = 44.49 + 16.97 \cdot E \qquad\qquad R^2 = 0.98 \qquad\qquad (6)$$
$$\quad\; (1.46) \quad (2.40)$$

where *E* is the annual NO_x emission given in Mt(N)/yr.

The hourly mean levels of surface ozone estimated for the three scenarios are shown in Figure IV.2.

Moreover, the ozone model is run for a corresponding set of scenarios for all of China (emissions outside China are kept constant):

— reference scenario, where the total NO_x emissions in China are 3.095 Mt(N)/yr;

— reference x 0.5 scenario, where the total NO_x emissions in China are 1.548 Mt(N)/yr;

— reference x 2 scenario, where the total NO_x emissions in China are 6.190 Mt(N)/yr.

The estimated function for the relationship between NO_x emissions in all of China (*E* given in Mt(N)/yr) and the surface ozone metric *M7* in Guangdong (given in ppbv) is (*SE* in parentheses):

$$M7 = 40.91 + 2.47 \cdot E \qquad\qquad R^2 = 0.97 \qquad\qquad (7)$$
$$\quad\; (1.90) \quad (0.46)$$

Whereas this relationship turned out to be nearly linear for most provinces, Guangdong included (cf. R^2), this was not the case for four provinces on the eastern coast, adjacent to the East China Sea and the Gulf of Chihli, especially Shandong, but also Jiangsu, Hebei and Tianjin. In the "reference x 2" scenario, surface ozone in Shandong was even estimated to be lower than in the reference scenario. This is a result of the atmospheric chemistry in high ozone areas; increasing NO_x emissions here may *reduce* surface ozone levels (ozone scavenging) (see Figure IV.3).

Figure IV.2. **Hourly Mean Surface Ozone during the Day**
(average for the period March-October) estimated for Guangdong for three emission scenarios:
(reference scenario, x 0.5 and x 2)
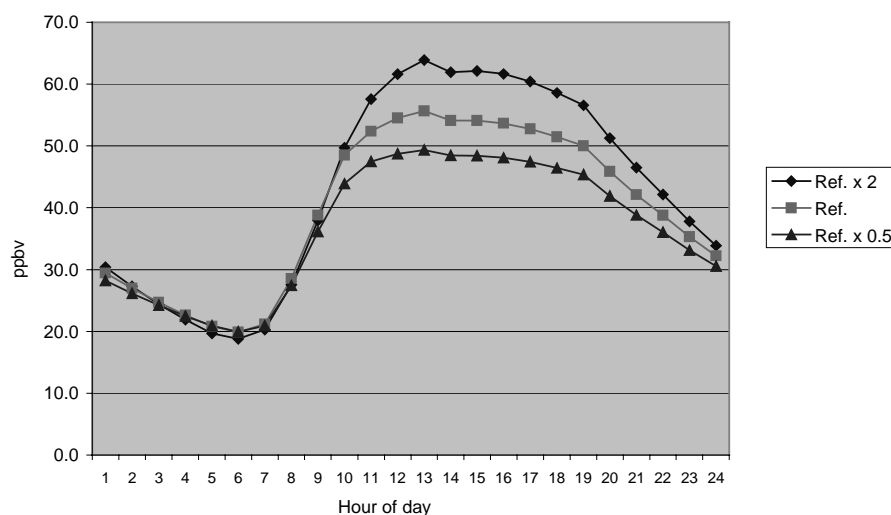
IV.2.2. Dose-Response Functions and Crops Grown in China

The annual production of crops for which dose-response functions are available is shown in Table IV.3. The share of the total Chinese production grown in Guangdong is also given, showing that Guangdong is a core area for rice, groundnuts (peanut), vegetables and tubers. Cotton and corn, two important crops in China, are not grown in Guangdong. In the methodology described here a static crop profile is assumed, and average figures are used for the period 1995-97. According to the data, grain production is presently declining in Guangdong, whereas the production of vegetables and edible oils is increasing.

The dose-response functions are given in Table IV.4 and Figure IV.4. These are taken from EC (1999) and Adams *et al.* (1989)[17]. Based on Mortensen (1995), it is assumed that tubers (mainly potato) have about the same sensitivity to ozone as wheat. Moreover, it is assumed that peanuts have approximately the same sensitivity as cotton, and the M7-function for cotton given in EC (1999) is applied to this crop. This is based on a comparison between the SUM06-functions[18] for peanut (only one cultivar) and several cotton cultivars rendered in USEPA (1996). Peanut has approximately the same shape of dose-response function as the average function for several cotton cultivars.

The dose-response functions are Weibull functions that take the following form:

$$y = a \cdot e^{-(x/s)^c} \qquad\qquad (8)$$

where *s* and *c* are the Weibull parameters,

y= crop yield,
a= hypothetical yield at 0 ppm ozone, usually normalised to 1, and
x= a measure of ozone concentration, here M7 (mean seasonal 7 hrs d^{-1}, in ppbv).

Since there is no data on the production of different types of vegetables in Guangdong, the approach used is to calculate a damage function for "vegetables" in general by averaging the functions for lettuce, tomato and spinach (denoted Vegetables-low). As the sensitivity to ozone of different vegetables seems to vary significantly, it is desirable to calculate an average function for the two more sensitive crops, tomato and spinach, denoted "Vegetables-high".

When applying the M7-functions, which have no threshold level for response, it is necessary to subtract the crop loss that can be attributed to the regional background level of ozone. One thereby obtains an estimate of the net loss attributable to ozone above regional background, denoted $NL_{>Reg}$. For Guangdong, for instance, as the constant in the linear function Eq. 7 is 40.9 ppbv, one may interpret this as the regional background M7 level, i.e. even if all NO_x emissions in Guangdong and the rest of China were shut off, the Guangdong region would still have a M7 (Mar.-Oct.) of about 41 ppbv. There is also a natural (pre-industrial) background level. This is estimated at about 16 ppbv. Correspondingly, one may estimate the net loss attributable to ozone above natural background, denoted $NL_{>Nat}$. Only $\Delta NL_{>Reg}$, however, is likely to be affected by emission reductions in China, because emissions outside the country affect tropospheric ozone in China.

Figure IV.3. **Modelled Surface Ozone in Three Scenarios**
(seasonal average 7 hrs d^{-1} for the period March-October (M7), given in ppbv)
(top: reference x 0.5 scenario; middle: reference scenario; bottom: reference x 2 scenario)
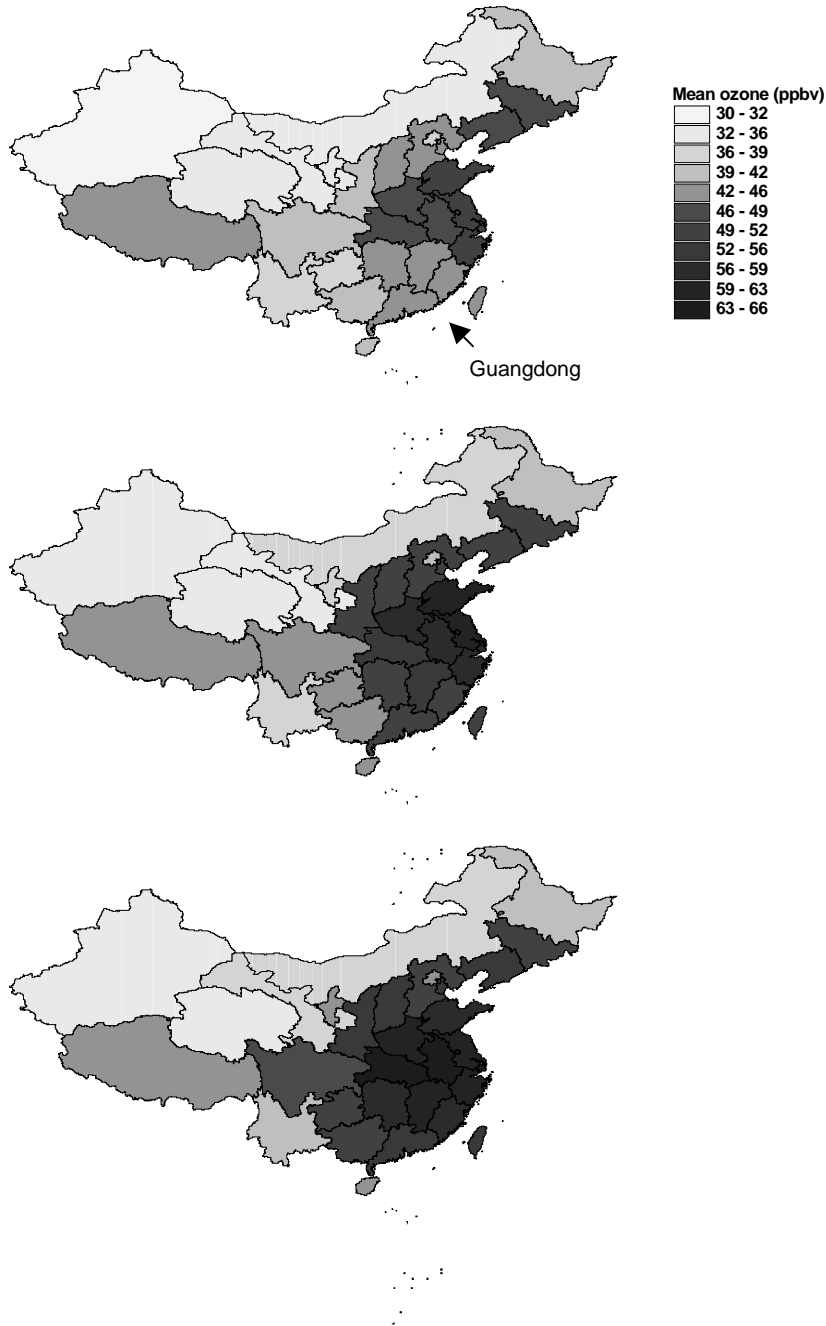
Table IV.3. **Crops Grown in China (total) and in Guangdong**

Crop	Average production in China 1995-97 (1 000 t)	Average production in Guangdong 1995-97 (1 000 t)	Guangdong share of Chinese Production (%)
Soybean	13 818	171.3	1.2
Peanut	10 007	723.7	7.2
Potato	33 301	2 149.3	6.5
Rice	193 689	15 344.3	7.9
Vegetables	337 928	20 079.6	5.9
Wheat	112 022	61.0[a]	0.1
Sorghum	4 690	1.3	0.03
Corn	114 590	-	
Cotton	4 524	-	

a) Winter wheat only.
Source: USDA, 2001.

Table IV.4. **Weibull Parameters in Dose-Response Functions**

Crop	s	c
Soybean	107	1.58
Corn	124	2.83
Wheat	136	2.56
Potato[a]	136	2.56
Rice	202	2.47
Cotton	111	2.06
Peanut[b]	111	2.06
Sorghum	314	2.07
Spinach	135	2.08
Lettuce	122	8.84
Tomato	142	2.369
"Vegetables" low	133	4.43
"Vegetables" high	138.5	2.23

a) Assumed to have the same sensitivity as wheat (Mortensen, 1995).
b) It is assumed that peanut has the same sensitivity as cotton.
Source: EC (1999) for all crops but rice and sorghum (Adams *et al.*, 1989).

Figure IV.4. **Dose-Response Functions for Relative Yield as Functions of M7**
(seasonal 7 hours d^{-1} given in ppbv)
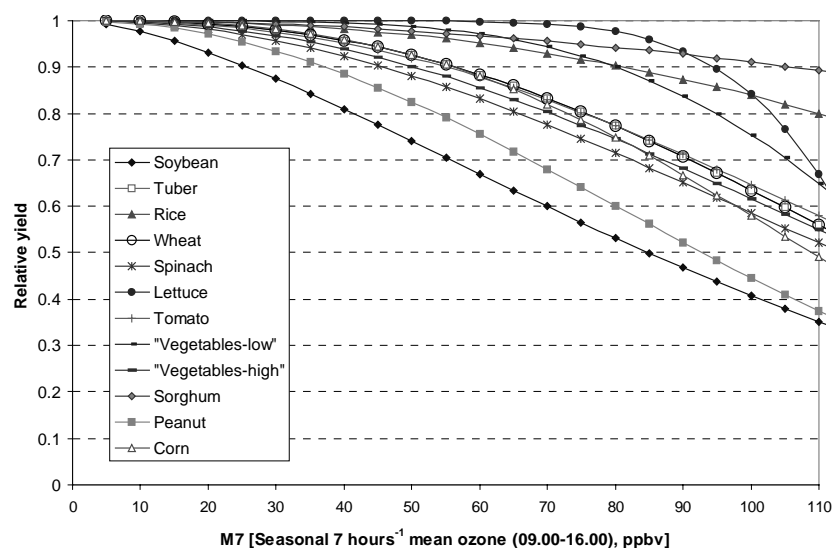
IV.2.3. Crop Losses in China as Functions of Annual Emissions of NO$_x$

Using the scenarios for M7 for each province, data for present crop production on a province level (P$_{pres}$), and the Weibull functions given in Table IV.4 to estimate crop loss ratios at different ozone levels, estimates of the crop loss in a given region and ozone scenario can be derived. For practical reasons, in the following the focus is on crop production as such instead of the crop production loss and functions are derived for P$_{scen}$, the crop production in a given ozone scenario. P$_{scen}$ is calculated as:

$$P_{scen} = P_{pres} [(1- L_{scen})/(1- L_{pres})] \qquad (9)$$

where L$_{scen}$ is the estimated crop loss ratio due to the ozone level in the scenario and L$_{pres}$ is the estimated loss ratio due to the present ozone level (in this case the estimated ozone level in the reference scenario).

To enable easy integration with the macroeconomic model, one can derive a stepwise linear function for P$_{scen}$ (given in 1000 tons) for each crop versus the annual NO$_x$ emission E (as an ozone precursor; given in Mt(N)/y):

$$P_{scen} = P_{reg} (1+b_1 E), \text{ when } E<a_1$$

$$P_{scen} = P_{reg} [1+ b_1 a_1 + b_2 (E-a_1)], \text{ when } a_1<E<a_2 \qquad (10)$$

$$P_{scen} = P_{reg} [1+ b_1 a_1 + b_2 (a_2-a_1) + b_3 (E-a_2)], \text{ when } a_2<E<a_3$$

where P$_{reg}$ is the hypothetical production (1000 tons) at zero emission within the region of interest (i.e. the production at the background ozone level, being the lowest level obtainable from eliminating all NO$_x$ emission *inside* the region of interest). P$_{reg}$ is calculated by the general equation for P$_{scen}$, Eq. 9, using regional background ozone as the scenario condition. Parameter values for Guangdong and ROC, respectively, are given in Table IV.5. "Other crops" consists of tuber, sorghum, soybean, peanut and vegetables.

Table IV.5. **Coefficients in Stepwise Linear Functions for Production of Crops Versus Emissions of NO$_x$ (as a surface ozone precursor) in Rest of China (ROC) and Guangdong**

Region	Parameter	Rice	Wheat	Corn	Cotton	Other crops
ROC	A	181082	117063	117870	4916	389630
	a_1	1.317	1.317	1.317	1.317	1.317
	a_2	2.635	2.635	2.635	2.635	2.635
	a_3	5.270	5.270	5.270	5.270	5.270
	b_1	-3.59E-03	-1.10E-02	-7.70E-03	-2.06E-02	-8.40E-03
	b_2	-7.88E-03	-2.21E-02	-1.58E-02	-3.99E-02	-1.69E-02
	b_3	-3.63E-03	-3.62E-03	-3.02E-03	-9.81E-03	-4.00E-03
Guangdong	A	15550	63	381		23763
	a_1	0.23	0.23	0.23		0.23
	a_2	0.46	0.46	0.46		0.46
	a_3	0.92	0.92	0.92		0.92
	b_1	-1.98E-02	-4.92E-02	-5.27E-02		-4.02E-02
	b_2	-3.78E-02	-9.39E-02	-1.03E-01		-7.65E-02
	b_3	-2.62E-02	-6.47E-02	-7.29E-02		-5.31E-02

Note: A $\equiv$ P$_{reg}$

There is quite good agreement between the original estimates for production in the different scenarios, which were based on applying the Weibull functions, and the estimates that are obtained from using the derived stepwise linear functions. This is illustrated for wheat production in ROC in Figure IV.5. As seen from the figure, the production loss per unit increase in NO$_x$ emission is less at high NO$_x$ levels, which is due to the non-linearities in the NO$_x$-ozone relationship under the high-NO$_x$ scenario in some of the eastern provinces, described above and shown in Figure IV.3. The effect of these non-linearities is enhanced by the fact that the four provinces where this is more pronounced — Shandong, Jiangsu, Hebei and Tianjin — have a high share of the production of many crops (e.g. 56 per cent of the production of wheat and peanut, and 53 per cent of the corn production.)

IV.2.4. Caveats

The methodology described here for estimating crop losses as functions of NO$_x$ emissions is simplified. Due to the coarse scale, it does not capture the geographical variations in distribution of crops and surface ozone. Neither does it take into consideration the effect that other components, such as nmVOC and CO, have on ozone formation. Including these components would likely introduce stronger non-linearities in the relationship between emissions and crop productivity.

Generally, there are large uncertainties connected to the basic assumption that Chinese cultivars have similar sensitivities to ozone as Western cultivars (whose sensitivities are those reported in studies). Moreover, there are especially large uncertainties connected to assumptions about potato having the same sensitivity to ozone as wheat and peanut having the same sensitivity as cotton. Finally, the largest uncertainty derives from the absence of independent tropospheric ozone measurements for China, so that reliance on model-generated estimates is complete.

Figure IV.5. **Wheat Yield as a Function of Annual Emission of NO$_x$**
(as a surface ozone precursor) in "Rest of China"
(top: the stepwise linearised function; bottom: the original Weibull function)

Stepwise linearised

Original (Weibull)
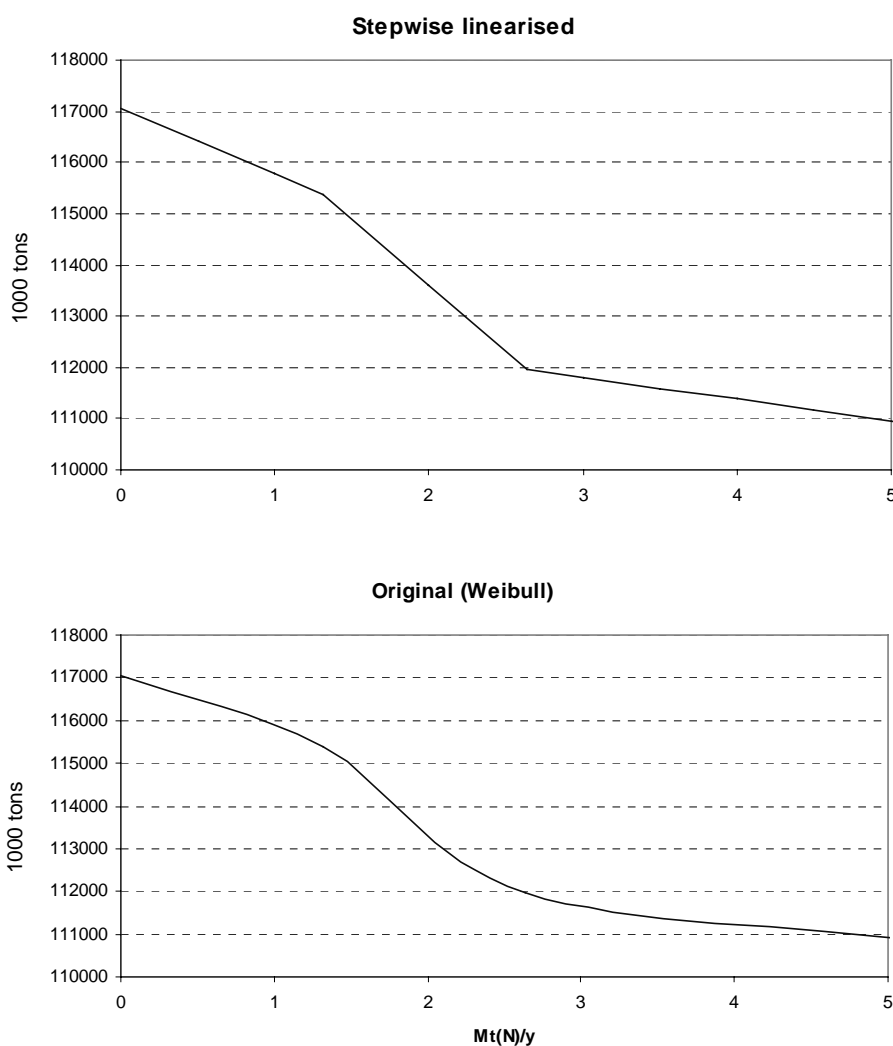
IV.2.5. Economic Analysis of Crop Productivity Effects

The change in agricultural productivity resulting from a change in pollution levels can be directly calculated at market prices. If the change is small enough so as not to affect those prices, the calculation is relatively simple. Where output changes actually alter equilibrium market prices, more information is needed, and a general equilibrium model is a useful tool for taking into account all effects of relative price changes.

The analysis is complicated in this case by feedback effects. Thus, if local air pollution reduces crop yields, a policy that lowers that pollution will improve those yields and, if relative prices and profitability are affected, this will alter resource allocation

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across sectors, resulting in turn in a different pollution profile. The latter in turn will have an effect on crop yields, and so on. The relevant question is whether these feedback dynamics are such that the model converges to a new stable equilibrium.

In effect, the climate policy can be thought of as changing the production technology of farmers, allowing them to produce more output with exactly the same purchased input bundle. In fact, this may not be an attractive option once general equilibrium effects are considered. First, input prices will also be affected by the carbon tax, and this may affect both the mix of inputs employed within agriculture and its relative profitability *vis-à-vis* other activities. Second, even if relative input prices were not affected, whether farmers would choose to apply the same quantities of inputs to achieve higher crop output or to reallocate some productive inputs to other activities depends on: *i)* the magnitude of the relative price and income changes induced by the policy; and *ii)* the relative income and price elasticities of agricultural commodities versus other goods and services.

V. THE BASELINE SIMULATION

The analysis of climate policy's ancillary benefits and costs depends crucially on defining a plausible counterfactual. How would China's economy, energy system and environmental policy evolve over the coming decade in the absence of explicit climate policy measures like a carbon tax or system of tradable carbon credits? The growth of China's economy to 2010 is probably the least uncertain element of the baseline. The average rate of GDP growth could vary within a few percentage points, but when the projected rate of growth is as high as China's, that does not materially alter the analysis. The evolving energy mix of the country in the absence of climate policy is far more difficult to predict. For instance, even though the base-year picture is one of very limited penetration of natural gas in the electricity and household sectors, there is reason to believe that government policy will promote a much more significant uptake by 2010. The assumptions about the energy sector are described below.

Finally, the biggest uncertainty may be in the area of environmental policy and its impacts on emissions. As Anderson and Cavendish (2001) point out, reductions in the pollution intensity of production do not normally occur gradually; rather, when a new control technology is introduced, there is quantum reduction in emissions — often on the order of 90-95 per cent. Thus, if Chinese environmental policy were to mandate or induce widespread adoption between now and 2010 of certain air emissions control technologies, the result could be dramatic improvements in air quality. What are the chances of that happening? One event that could catalyse stronger air pollution control measures in the capital city are the Beijing Summer Olympic Games scheduled for 2008. One can only speculate on how far any clean-up efforts might extend beyond the capital to other heavily polluted cities. In any case, the baseline environmental assumptions are fairly generous in projecting how large a reduction in emissions intensity will occur from now to 2010 in the absence of an explicit climate policy.

V.1. Defining the Economic, Energy and Environmental Baseline

How is China's economy and energy structure likely to evolve to 2010, assuming no explicit action is taken to curtail greenhouse gas emissions? What about emissions of various local air pollutants and ambient environmental conditions? This depends on a variety of factors, some of which are not known with a high degree of certainty. For instance, what will happen to world energy prices over the next decade remains highly uncertain. For energy prices, the latest IEA oil price projections to 2010 are used.

The base-year (1997) input-output tables are thought to be a poor starting-point for projecting future penetration rates of natural gas. They show zero imports of natural gas for Guangdong. Yet, plans for China's first liquified natural gas (LNG) terminal, to be

located in the province, are progressing and it could be operational by 2005[19]. To reflect the fact that China is expected to import significant quantities of natural gas in next 10 years, the Armington import share of natural gas is gradually increased, to 30 per cent in Guangdong and 5 per cent in ROC by 2010. Also, it is arbitrarily assumed that the share parameter of natural gas in households' ELES will gradually increase from 0 to 7.5 per cent of total energy consumption in Guangdong from 1997-2010. This combination of assumptions yields a relatively high natural gas baseline demand trend (see Figure V.2), but given the low starting point, the aggregate effects are relatively small by 2010.

V.2. Baseline Simulation Results

Table V.1 presents the main baseline assumptions and Figure V.1 shows baseline trends in GDP, energy, and emissions of major air pollutants from 1997-2010. The baseline assumes that there are no climate policy changes in the simulation period, and provides a reference to compare with the climate policy scenarios. In the baseline, the GDP growth rates are exogenously set at 7.6 per cent per annum for all China, 8.5 per cent for Guangdong and 7.5 per cent for the rest of China. Population and labour force growth are also exogenous. Urban population is assumed to grow at a significantly faster rate than the overall rate of population growth. Non-energy sector labour productivity is assumed to grow by 5 per cent per annum over the simulation period, while energy sector labour productivity grows at 3 per cent. Based on recent historical experience in China, an autonomous energy efficiency improvement (AEEI) factor of 2.5 per cent per annum is assumed. Finally, emission factors are exogenously reduced by 2.5 per cent per annum for TSP and 1.5 per cent per annum for other pollutants.

Under these assumptions, Figure V.1 shows that China's energy-GDP ratio will fall steadily to 2010 by about 1.2 per cent per annum. Emissions growth for all pollutants is slower than growth in energy consumption[20]. Of all the pollutants, VOCs grow the fastest, followed by NO_x. Both trends are closely linked to the rapid rise in the motor vehicle stock and the shift towards natural gas in power generation and heating (Figure V.2). It should be recalled that gas's rapid demand growth is from a rather small base, with gas contributing a negligible share of power generating capacity in 1999.

Table V.1. **Exogenous Variable Growth Assumptions**
(% per annum)

	1998-2000	2001-03	2004-06	2007-10
Real GDP				
Guangdong	9.0	8.5	8.2	8.2
Rest of China	8.5	7.5	7.2	7.2
Population				
Guangdong	1.60	1.50	1.40	1.30
Urban	3.20	3.20	3.20	3.20
Rural	0.86	0.66	0.44	0.18
Rest of China	0.90	0.90	0.85	0.82
Urban	2.60	2.60	2.60	2.60
Rural	0.16	0.10	-0.03	-0.17
Labour force				
Guangdong	2.0	2.0	2.0	2.0
Rest of China	1.0	1.0	1.0	1.0
Labour productivity				
Energy sector	3.0	3.0	3.0	3.0
Non-energy sector	5.0	5.0	5.0	5.0
AEEI	2.5	2.5	2.5	2.5

*   The growth rate of population in large cities is same as for urban population as a whole.

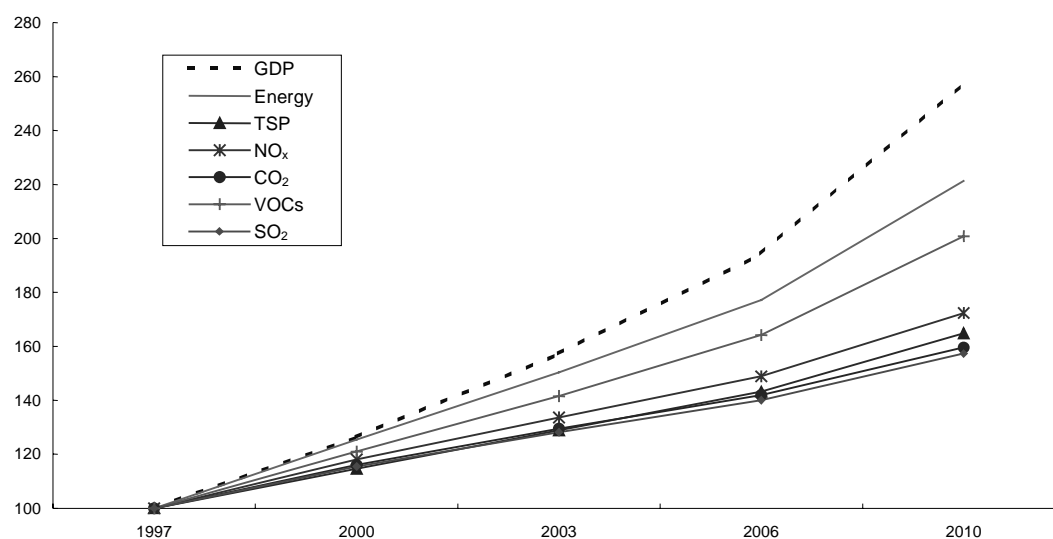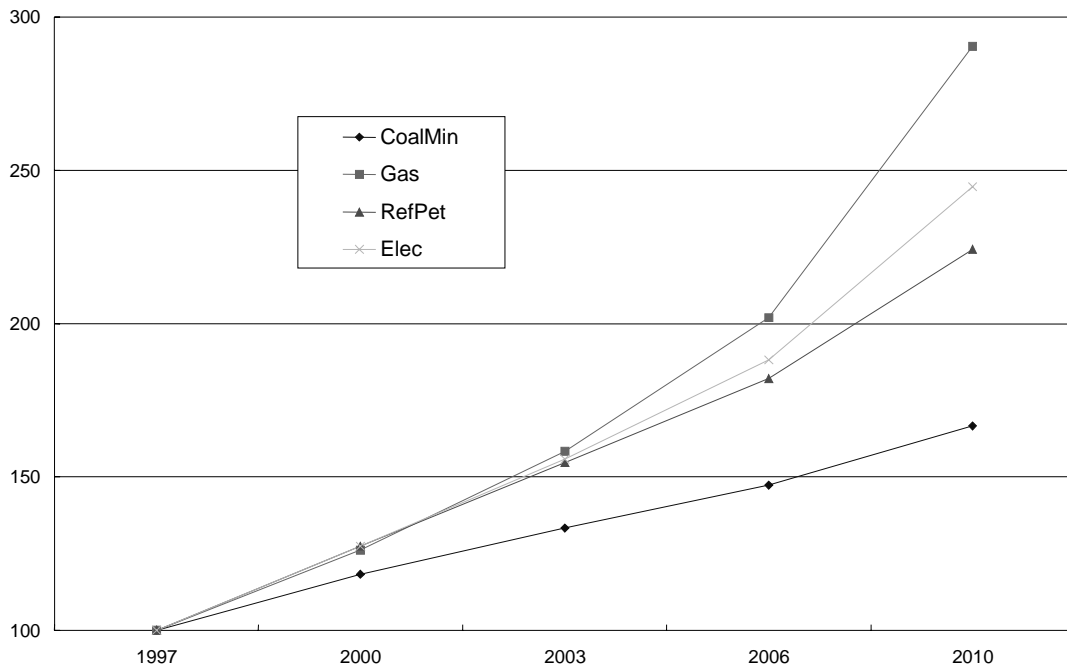Figure V.1. **Growth of GDP, Energy Consumption and Emissions in Baseline, all China**

Figure V.2. **Growth of Energy Demand in Baseline, by type, all China**

VI. CLIMATE POLICY SCENARIOS

Climate policy is modelled as a national carbon tax, i.e. a tax on the carbon content of fossil fuels. The tax is calibrated to achieve targeted percentage reductions in national CO_2 emissions from baseline 2010 levels. CO_2 emission reductions are phased in, from a minimum of 5 per cent to a maximum of 30 per cent in the final simulation year, i.e. 2010. Holding regional real government and corporate savings constant, any additional carbon tax revenues are redistributed back to households. In practice, this means that households receive revenues roughly equal to those returned to corporations. In the "basic" policy scenario results described in the next two sections, complete factor immobility across regions is assumed. (The effects of relaxing this assumption are described later.) First, results are reported for the case where only health effects are valued. Then the agricultural productivity effects of climate policy are included.

VI.1. Scenario Results with Health Effects Only

The basic climate policy simulation results are illustrated in Figures VI.1.a. to VI.1.c. The diagram depicts the *welfare changes* generated by the imposition of the carbon tax, the *ancillary health benefits* associated with the tax, and the *net benefits (costs)* of the policy as given by the vertical difference between the first two curves. Note that the ancillary benefits curve is a straight line from the origin, reflecting: *i)* the absence of a lower threshold for major health effects of air pollution; and *ii)* the constancy of the dose-response relationship irrespective of the initial pollution level. These are simplifications but defensible ones based on the bulk of the epidemiological evidence (notably, particulate-health studies). The slope of the welfare change curve is consistent with the standard result of increasing marginal costs of CO_2 abatement. Where the net benefits curve lies above the *x*-axis, the carbon tax could be interpreted as a *no regrets* one in that the monetised health improvements outweigh the abatement costs, broadly defined. Figure VI.1.a. shows that the maximum "no regrets" abatement rate in 2010 is 5 per cent of baseline emissions for China as a whole.

The "no regrets" abatement rates are slightly different across the two regions of China, being higher in Guangdong (GD) than in Rest of China (ROC). Figures VI.1.b. and 6.1.c. present the welfare analysis for GD and ROC, respectively. Table VI.1 makes plain that the major inter-regional difference is in how terms of trade are affected by the tax. Whereas ROC is a carbon-fuel and energy-intensive good exporting region, GD is the reverse. So, the carbon tax depresses the export prices of ROC, while depressing GD's import prices. Thus, for a 10 per cent carbon reduction, the decline in real GDP for GD is roughly double that for ROC (as a share of regional GDP), while the reverse is true

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for the welfare change (as measured by equivalent variation, EV). This strongly reinforces the result of Bussolo and O'Connor (2001) for India that regional distributive effects of carbon taxes need to be carefully considered where — as is often the case — fossil fuel extraction and energy-intensive industries tend to be heavily regionally concentrated.

Another noteworthy feature of Table VI.1 is that, for a given national carbon abatement rate, GD abates proportionately less than ROC, reflecting a slightly higher marginal abatement cost curve. This is attributable in part to the regional aggregation in the model. The economic size of ROC is roughly nine times that of GD (see Table VI.2). Generally, the intermediate inputs are more diversified in a larger economy; this should be reflected in more zero cells in the input-output matrix for the smaller economy. Indeed, for the five energy inputs to 64 sectors (5*64 matrix, with 320 cells), there are 75 zero cells for ROC to 180 zero cells for GD. So, clearly, it is harder to substitute across energy inputs in GD than in ROC.

Table VI.3 reports the regional welfare losses, ancillary benefits and net benefits/costs as a share of regional GDP for different abatement rates. Because the ROC has higher local air pollution levels than GD, generally relying on dirtier fuels — notably coal (see Table VI.2) — and having higher coal consumption per unit of GDP, its ancillary benefits from carbon abatement are higher than Guangdong's. Even so, it enjoys a lower "no regrets" abatement rate due to the adverse terms of trade effect noted above that adds to the costs of the policy.

Table VI.4 reports the material welfare losses of urban and rural households, respectively, and the net benefits to urban households (rural households are assumed to enjoy no benefits from improved urban air quality), measured as a share of their disposal income. Urban households suffer larger material welfare losses than rural households in ROC, while in GD urban households suffer losses even though rural households experience gains up to an abatement of just under 15 per cent from baseline. One explanation is that urban households spend a larger share of their budgets on energy than rural ones. Also, the urban households lose as suppliers of factors to relatively energy-intensive — and now less profitable — sectors compared to rural households employed mostly in agriculture and light industry. Finally, only rural households own land, and the land rental rate declines less under a carbon tax than the wage rate and profit rate.

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Figure VI.1.a. **Welfare Analysis with Health Impacts only, 2010, All China**

China-wide Welfare Analysis

CO$_2$ reduction from 2010 baseline
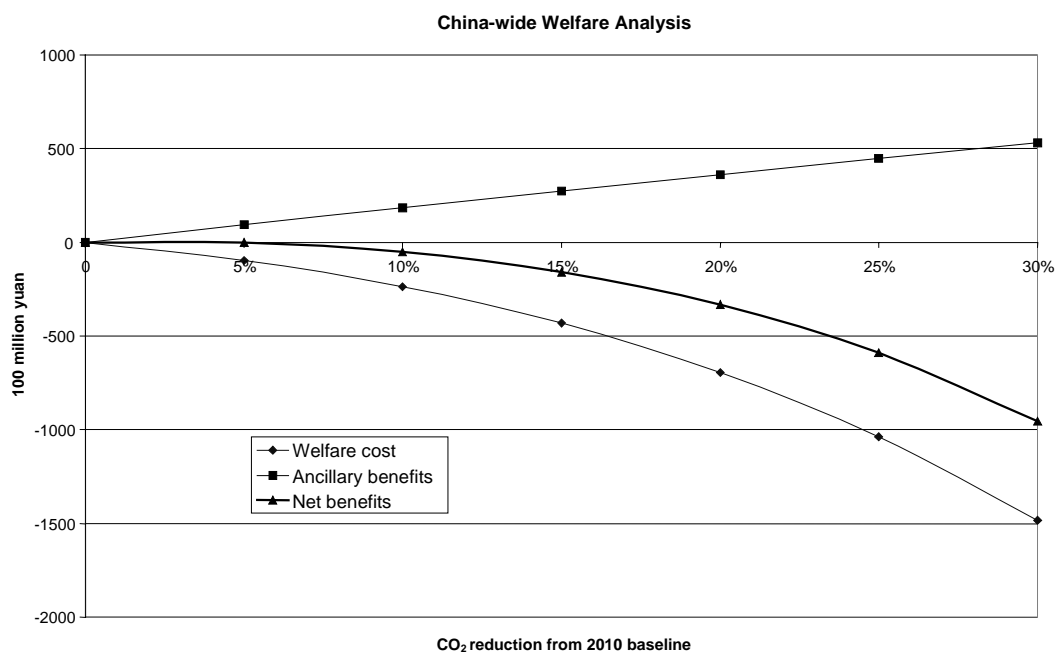
Figure VI.1.b. **Welfare Analysis with Health Impacts only, 2010, Guangdong**

Welfare Analysis for Guangdong

CO$_2$ Reduction from 2010 baseline
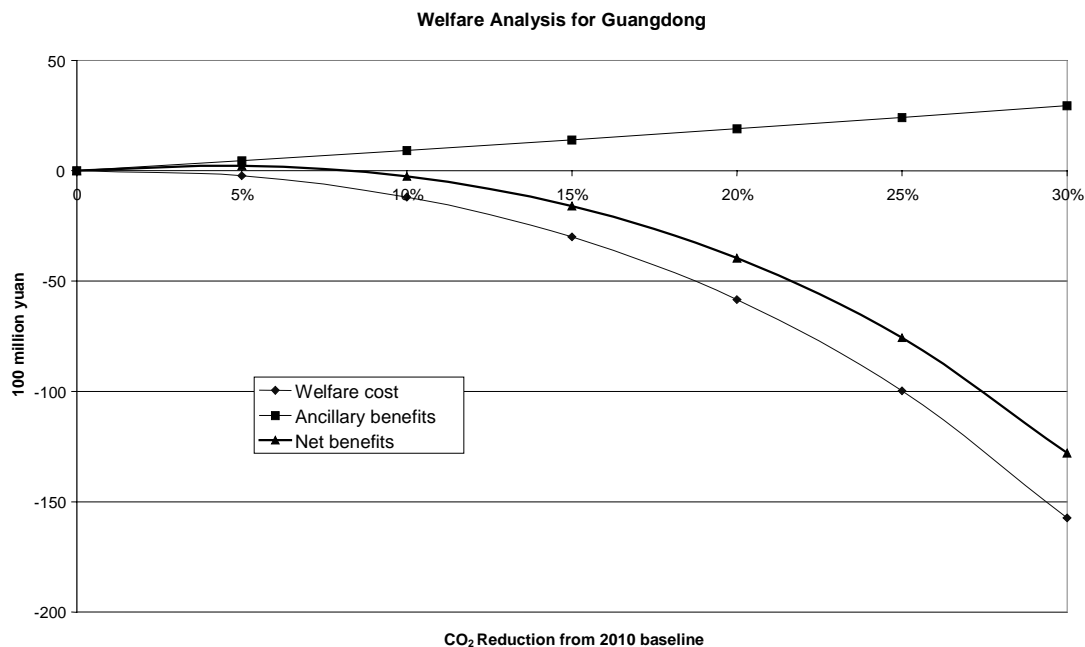
Figure VI.1.c. **Welfare Analysis with Health Impacts only, 2010, Rest of China**

Welfare Analysis, Rest of China

CO₂ reduction from 2010 baseline
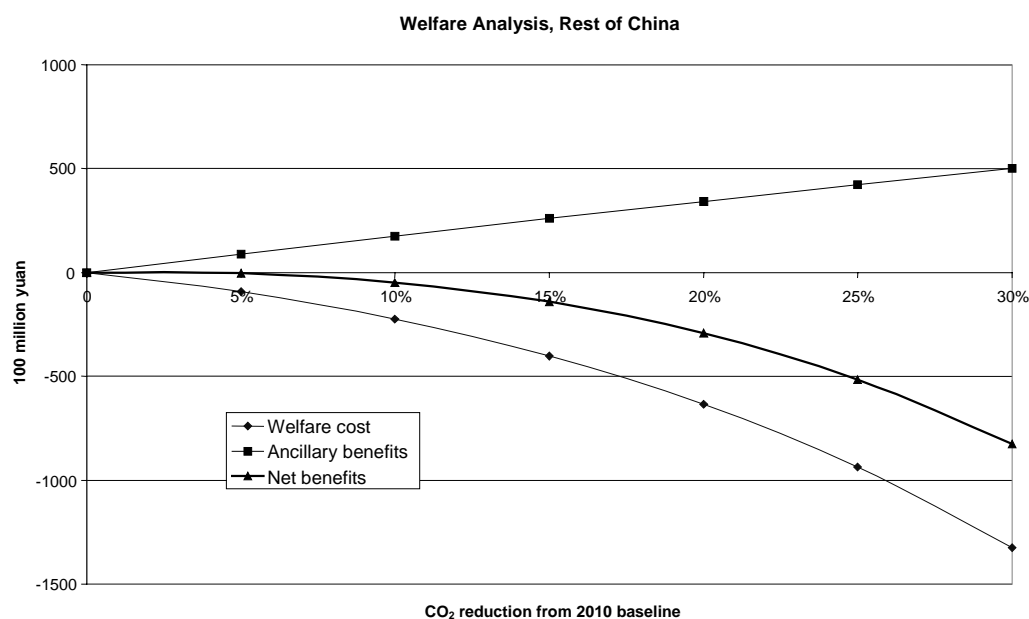
Table VI.1. **Welfare (EV) and GDP Effects of Carbon Reduction, health only, 2010**
(% change relative to baseline)

	Reduction in CO_2 emissions					
	5%	10%	15%	20%	25%	30%
National						
Carbon emission	-5.00	-10.00	-15.00	-20.00	-25.00	-30.00
EV(as % of GDP)	-0.05	-0.12	-0.22	-0.35	-0.53	-0.75
GDP	-0.05	-0.13	-0.23	-0.37	-0.54	-0.77
Guangdong						
Carbon emission	-4.80	-9.64	-14.53	-19.47	-24.48	-29.57
EV(as % of GDP)	-0.01	-0.06	-0.14	-0.28	-0.47	-0.75
GDP	-0.10	-0.23	-0.40	-0.61	-0.88	-1.21
Terms of trade	0.15	0.28	0.40	0.51	0.60	0.66
Rest of China						
Carbon emission	-5.02	-10.04	-15.06	-20.06	-25.06	-30.05
EV(as % of GDP)	-0.05	-0.13	-0.23	-0.36	-0.53	-0.75
GDP	-0.05	-0.12	-0.21	-0.34	-0.50	-0.71
Terms of trade	-0.15	-0.28	-0.40	-0.51	-0.59	-0.66

Table VI.2. **Energy Structure and Carbon Intensity by Region, 1997**
(% of national total)

	Guangdong	Rest of China
GDP	9.6	90.4
Carbon emission	9.4	90.6
Energy demand	11.0	89.0
Coal	7.6	92.4
Gas	12.7	87.3
RefPet	14.4	85.6
Elec	11.9	88.1

Table VI.3. **Welfare Effects (EV), Ancillary Benefits, and Net Benefits of Carbon Reduction, Health only, 2010** (as % of GDP)

	Reduction in CO_2 emission					
	5%	10%	15%	20%	25%	30%
National						
EV	-0.05	-0.12	-0.22	-0.35	-0.53	-0.75
Ancillary benefit	0.05	0.09	0.14	0.18	0.23	0.27
Net benefit	0.00	-0.03	-0.08	-0.17	-0.30	-0.48
Guangdong						
EV(as % of GDP)	-0.01	-0.06	-0.14	-0.28	-0.47	-0.75
Ancillary benefit	0.02	0.04	0.07	0.09	0.11	0.14
Net benefit	0.01	-0.01	-0.08	-0.19	-0.36	-0.61
Rest of China						
EV(as % of GDP)	-0.05	-0.13	-0.23	-0.36	-0.53	-0.75
Ancillary benefit	0.05	0.10	0.15	0.19	0.24	0.28
Net benefit	0.00	-0.03	-0.08	-0.17	-0.29	-0.47

Table VI.4. **Inter-Household Impact of Carbon Reduction, 2010, Health only**
(as % of disposal income)

			Reduction in CO_2 emissions			
	5%	10%	15%	20%	25%	30%
National						
EV Urban	-0.10	-0.25	-0.43	-0.68	-1.00	-1.40
EV Rural	-0.04	-0.11	-0.22	-0.38	-0.60	-0.89
Net benefits urban	0.03	0.01	-0.05	-0.18	-0.37	-0.66
Guangdong						
EV Urban	-0.06	-0.18	-0.37	-0.64	-1.03	-1.55
EV Rural	0.05	0.05	-0.01	-0.13	-0.34	-0.65
Net benefits urban	0.00	-0.06	-0.19	-0.41	-0.73	-1.18
Rest of China						
EV Urban	-0.11	-0.25	-0.44	-0.68	-0.99	-1.38
EV Rural	-0.05	-0.13	-0.25	-0.41	-0.63	-0.91
Net benefits urban	0.03	0.02	-0.03	-0.15	-0.33	-0.60

VI.2. Carbon Tax Rate, Price Effect, and Fiscal Impact

Table VI.5 reports the carbon tax rates for different abatement rates. It is clear that the carbon tax rises at an increasing rate as the constraint on carbon emissions becomes more stringent, indicating that the marginal efficiency of the carbon tax declines in abatement. At the "no regrets" rate of 5 per cent abatement, the carbon tax approaches $7/tC (1997 prices and exchange rate).

Depending on (direct and indirect) carbon content, the price effect of the tax differs considerably across energy goods. As coal has the highest carbon content among fossil fuels, its price increases most significantly, i.e. by about 7 per cent for a 5 per cent carbon reduction. Price increases for other fuels and energies are generally under 2 per cent. Since the ROC relies more than GD on hydroelectric power in its electricity sector, its electricity price is less affected by the carbon tax.

In terms of fiscal impact, a "no regrets" carbon tax would generate non-negligible but still modest revenues. Table VI.5 also reports the carbon tax revenue as a share of GDP, which stands at just over one-third of one per cent of GDP for a 5 per cent carbon reduction, rising to over 1 per cent for a 15 per cent reduction. Considering that government tax revenue amounted to 14 per cent of GDP in the year 2000, a $7/tC tax could generate perhaps 2.5 per cent of Chinese government revenue. A $15/tC tax would roughly double the percentage carbon reduction and the revenue share. For reference purposes, in 2000 tariff revenues contributed around 6 per cent of total government tax revenue (*China Statistical Yearbook 2001*).

Table VI.5. **Carbon Tax Rate, Price Effects on Energy Goods and Carbon Tax Revenue, 2010**

	Reduction in CO_2 emission				
	5%	10%	15%	20%	25%	30%
Carbon tax (Yuan/tC)	56	121	196	287	398	537
Carbon tax ($/tC)	7	15	24	35	48	65
Price of energy goods (% change from baseline)						
Guangdong						
Coal	7.24	15.92	26.51	39.68	56.48	78.53
Natural Gas	1.66	3.59	5.88	8.66	12.12	16.55
Petroleum	1.96	4.22	6.88	10.08	14.01	18.98
Electricity	1.53	3.27	5.30	7.69	10.57	14.12
Rest of China						
Coal	7.44	16.35	27.21	40.72	57.93	80.46
Natural Gas	2.13	4.60	7.50	10.99	15.27	20.68
Petroleum	1.88	4.05	6.63	9.73	13.56	18.42
Electricity	0.91	1.95	3.17	4.60	6.31	8.41
Carbon tax revenue (as % of GDP)						
National	0.35	0.72	1.10	1.52	1.98	2.49
Guangdong	0.35	0.72	1.11	1.53	2.00	2.53
Rest of China	0.35	0.72	1.10	1.52	1.97	2.49

VI.3. Climate Policy Scenario with NO_x-Crop Yield Linkage

The climate policy scenario is now extended to include agricultural productivity effects. The relationships estimated above between NO_x emissions, ambient ozone and crop yields are incorporated to determine how a carbon tax will affect output of various crops and, in turn, relative prices, incomes of rural households and consumer welfare. As Figure IV.5 illustrates for the case of wheat, a decrease of NO_x emissions improves crop productivity. As Table VI.6 shows, for a 5 per cent carbon reduction, NO_x declines by 3 per cent and the increment of crop productivity ranges between 0.1-0.24 per cent in Guangdong and 0.07-0.25 per cent in rest of China.

Although the direct productivity gains in crop sectors seem modest, there are also second-order productivity effects as productive factors are reallocated from agriculture to higher-productivity industrial activities. In the case of labour, for instance, while most sectors experience a positive employment effect of the "with crops" scenario compared with "without crops", the major crop sectors experience the reverse, i.e. smaller employment increases (under the basic policy scenario, these labour-intensive sectors add workers at the expense of more energy-intensive ones). In total, some 880 0000 agricultural labourers would transfer to other, higher-productivity sectors[21] once crop yield effects are included. Furthermore, the real GDP boost from higher productivity in early years implies higher investment and faster GDP growth in subsequent years[22]. All told, the implications for the welfare costs of carbon abatement are significant. Table VI.7 reports the results for EV and net benefits (ancillary health benefits being unchanged from the previous scenario). The national welfare costs (EV) for a 5 per cent carbon reduction are only one-fifth as large a share of GDP in the with-crops scenario as without NO_x-crop linkages.

Under the with-crop scenario, the national "no regrets" abatement rate increases to between 15-20 per cent reduction from 2010 baseline emissions (Figure VI.2.a to VI.2.c). By taking the difference between the welfare loss (EV) for a given abatement rate (say, 5 per cent) without crops (0.05 per cent of GDP) and the welfare loss with crops (0.01 per cent of GDP), one arrives at an estimate of the ancillary crop benefits of the policy (which amount to 0.04 per cent of GDP, compared with the ancillary health benefits — see Table VI.3 — of 0.05 per cent). In other words, the crop benefits are just slightly smaller than the health benefits at a 5 per cent abatement rate; the same applies at 10 per cent abatement.

The welfare gains (or reduced losses) are proportionately larger for ROC than for GD, primarily because of the difference in their respective sectoral GDP shares: in GD agriculture accounted for only 10 per cent of GDP in 2000, while in ROC it accounted for 17 per cent. Also, the crops that are most sensitive to ozone — e.g. soybeans and corn — are concentrated in ROC, with GD's production negligible (refer to Table IV.3).

Table VI.8 shows the distribution of economic costs (as reflected in *EV*) across urban and rural households for the case with crop effects. Compared with the results in Table VI.4, the costs are significantly lower for both groups of households, but proportionately the improvement is greater for rural households. Indeed, at the national level, rural households actually experience positive welfare gains beyond 10 per cent CO_2 reduction. The higher agricultural productivity translates directly into higher rural incomes. Rural households also benefit disproportionately as consumers from lower food prices because of the larger weight of food items in their household expenditures. The productivity gains in agriculture also raise welfare (or mute its decline) in urban areas via a generalised growth effect. So, whereas a 10 per cent CO_2 reduction *without* crop effects causes a 0.25 per cent decline in real urban disposable income, the same percentage reduction *with* crop effects causes only a 0.15 per cent decline. This translates into higher net benefits for urban households, once ancillary benefits are included.

Because urban households are assumed to capture all the benefits of local air quality improvements, while the costs are distributed across both urban and rural households, the urban population considered alone would favour a somewhat higher rate of carbon reduction than the Chinese population taken as a whole. This applies whether one considers the "with-crops" or "without-crops" scenario. For example, in the "with-crops" case, comparing only urban households' costs to their ancillary benefits yields a "no regrets" abatement rate over 20 per cent.

Equation 11 decomposes total CO_2 emissions into easy-to-interpret components: *i)* the sectoral composition of output; *ii)* the carbon-intensity of energy; *iii)* the energy intensity of the economy; and *iv)* the scale of production, or total output:

$$E = \sum_i \left(\frac{X_i^{Output}}{X_{tot}^{Output}} \frac{E_i}{Ene_i} \frac{Ene_i}{X_i^{Output}} X_{tot}^{Output} \right) \qquad (11)$$

Equation 12 represents a change in emissions as a function of changes in these various components:

$$\partial E = \sum_i \left[\partial\left(\frac{X_i^{Output}}{X_{tot}^{Output}}\right)\frac{E_i}{X_i^{Output}} + \partial\left(\frac{E_i}{Ene_i}\right)Ene_i + \partial\left(\frac{Ene_i}{X_i^{Output}}\right)\frac{E_i}{Ene_i}X_i^{Output} + \partial\left(X_{tot}^{Output}\right)\frac{E_i}{X_i^{Output}} \right] \quad (12)$$

where ∂ is the differential operator, E total emission volume, X_{tot}^{Output} total output (in real terms), E_i the sectoral emission volumes, Ene_i the sectoral fuel (energy) use, and X_i^{Output} the sectoral outputs. Table VI.9 presents the results of solving this identity for the two regions in the China model and for various rates of emission reduction. Changes in the carbon intensity of energy are the single largest source of emission reductions, accounting for 40-60 per cent of the total for a 5 per cent reduction. Changes in the energy intensity of output are the next most important source, accounting for roughly a third of the total for a 5 per cent reduction. The main difference between GD and the ROC is the far more prominent place of sectoral composition changes in the former than the latter. This brings us back to the earlier point about small relative size of the GD economy and the more limited energy substitution possibilities.

Table VI.6. **Impacts on NO_x Emissions, Productivity and Output in Crop Sectors, 2010**
(% change relative to baseline)

| | Reduction in CO_2 emission | | | | | |
	5%	10%	15%	20%	25%	30%
NO_x Emission						
National	-3.00	-6.06	-9.16	-12.33	-15.57	-18.89
Guangdong	-3.87	-7.78	-11.73	-15.73	-19.79	-23.90
Rest of China	-2.90	-5.85	-8.86	-11.93	-15.08	-18.31
Productivity Guangdong						
Rice	0.09	0.19	0.28	0.38	0.47	0.57
Wheat	0.24	0.48	0.72	0.97	1.22	1.47
Other crops	0.19	0.39	0.58	0.78	0.99	1.19
Productivity Rest of China						
Rice	0.08	0.17	0.26	0.34	0.43	0.53
Wheat	0.09	0.17	0.26	0.35	0.45	0.54
Corn	0.07	0.14	0.22	0.29	0.37	0.44
Cotton	0.25	0.51	0.76	1.03	1.30	1.58
Other crops	0.09	0.19	0.29	0.39	0.49	0.59
Output, National						
Rice	0.10	0.20	0.29	0.38	0.46	0.53
Wheat	0.22	0.45	0.68	0.91	1.15	1.39
Corn	0.19	0.37	0.55	0.73	0.91	1.09
Cotton	0.43	0.88	1.34	1.82	2.34	2.90
Other crops	0.14	0.27	0.39	0.51	0.62	0.71
Output, Guangdong						
Rice	-0.03	-0.08	-0.16	-0.27	-0.43	-0.65
Wheat	2.31	4.82	7.56	10.61	14.06	18.03
Other crops	0.26	0.52	0.77	1.01	1.25	1.48
Output, Rest of China						
Rice	0.12	0.23	0.34	0.45	0.56	0.67
Wheat	0.22	0.45	0.67	0.91	1.14	1.39
Corn	0.19	0.37	0.55	0.73	0.91	1.09
Cotton	0.43	0.88	1.34	1.82	2.34	2.90
Other crops	0.13	0.25	0.37	0.47	0.57	0.66

Table VI.7. **Welfare Effects (EV) and Net Benefits of Carbon Reduction, with Crops, 2010**
(as % of GDP)

		Reduction in CO_2 emission				
	5%	10%	15%	20%	25%	30%
National						
EV (as % of GDP)	-0.01	-0.05	-0.12	-0.21	-0.35	-0.55
Ancillary health benefit	0.05	0.09	0.14	0.18	0.23	0.27
Net benefit	0.03	0.04	0.02	-0.03	-0.13	-0.28
Guangdong						
EV (as % of GDP)	0.01	-0.02	-0.09	-0.21	-0.39	-0.65
Ancillary health benefit	0.02	0.04	0.07	0.09	0.11	0.14
Net benefit	0.03	0.02	-0.02	-0.12	-0.27	-0.51
Rest of China						
EV (as % of GDP)	-0.02	-0.05	-0.12	-0.21	-0.35	-0.53
Ancillary health benefit	0.05	0.10	0.15	0.19	0.24	0.28
Net benefit	0.03	0.05	0.03	-0.02	-0.11	-0.25

Figure VI.2.a. **Welfare Gains in 2010, with Crop Effects, all China**

China-wide Welfare Analysis

CO₂ reduction from 2010 baseline
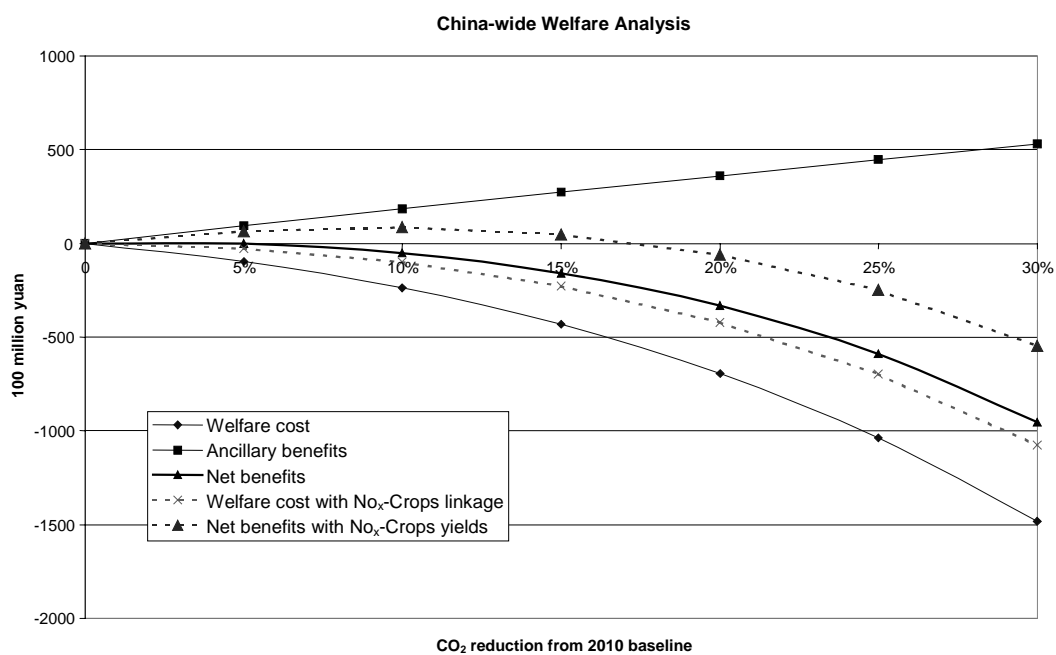
Figure VI.2.b. **Welfare Gains in 2010, with Crop Effects, Guangdong**

Welfare Analysis for Guangdong

CO₂ Reduction from 2010 baseline
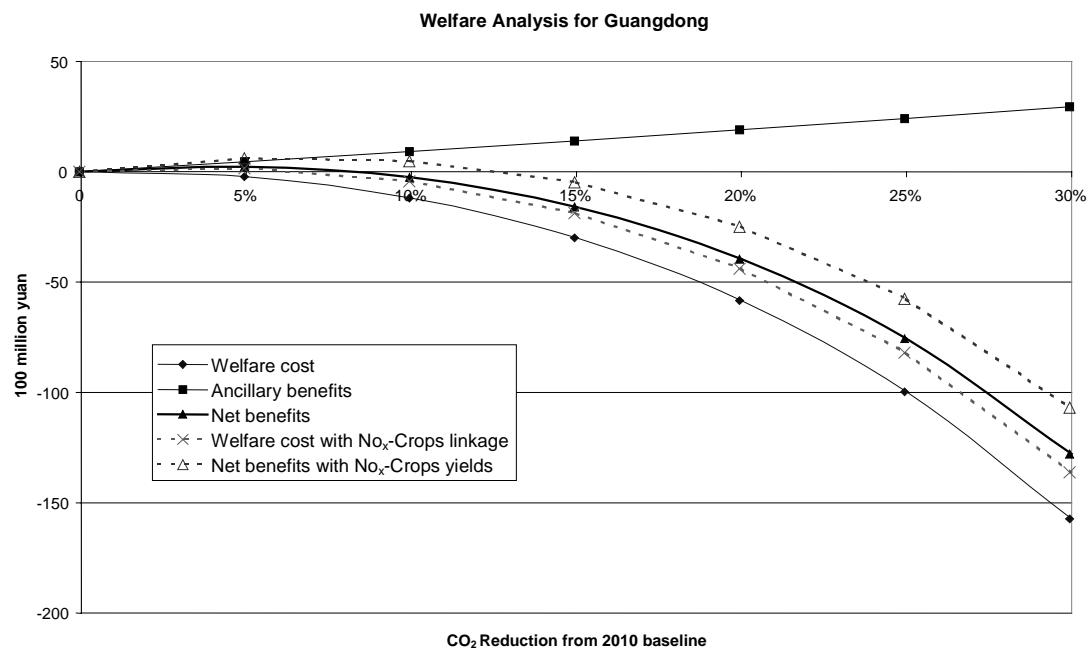
Figure VI.2.c. **Welfare Gains in 2010, with Crop Effects, Rest of China**

Welfare Analysis, Rest of China

CO$_2$ reduction from 2010 baseline
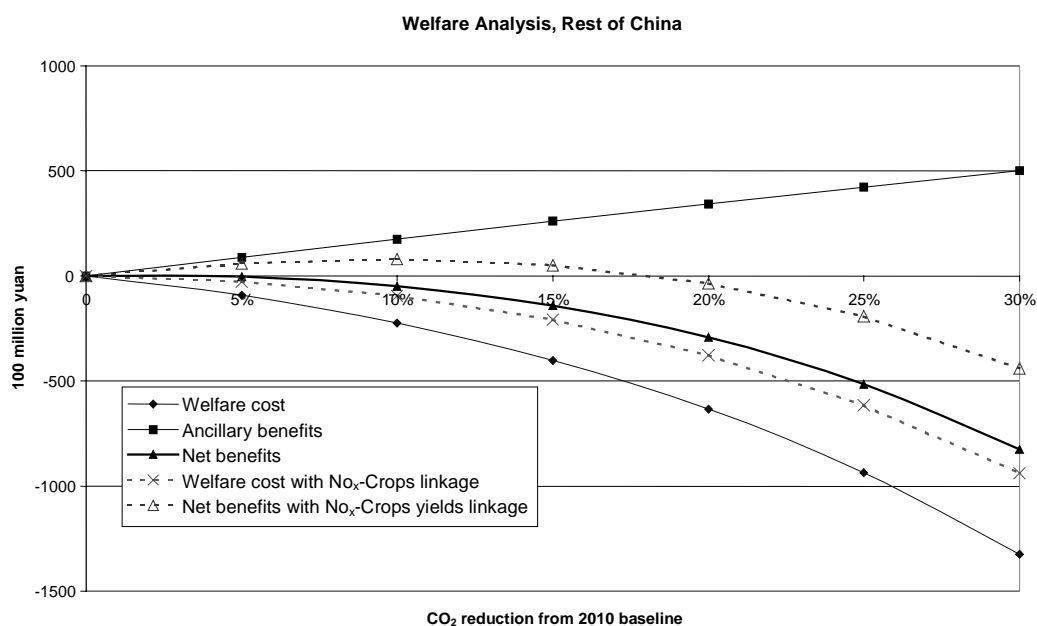
Table VI.8. **Inter-Household Impact of Carbon Reduction, with Crops, 2010**
(as % of disposal income)

	Reduction in CO$_2$ emissions					
	5%	10%	15%	20%	25%	30%
National						
EV Urban	-0.05	-0.15	-0.29	-0.48	-0.75	-1.11
EV Rural	0.02	0.01	-0.04	-0.14	-0.30	-0.53
Net benefits urban	0.08	0.11	0.10	0.02	-0.13	-0.37
Guangdong						
EV Urban	-0.04	-0.13	-0.30	-0.56	-0.93	-1.44
EV Rural	0.09	0.13	0.11	0.03	-0.14	-0.41
Net benefits urban	0.02	-0.02	-0.13	-0.33	-0.63	-1.07
Rest of China						
EV Urban	-0.06	-0.15	-0.28	-0.47	-0.73	-1.07
EV Rural	0.01	0.00	-0.06	-0.16	-0.31	-0.54
Net benefits urban	0.08	0.13	0.13	0.07	-0.06	-0.28

Table VI.9. **Decomposition of CO_2 Emission Change, 2010**

| | Reduction in CO_2 emissions |
	5%	10%	15%	20%	25%	30%
GD						
Sector Composition	23.6	23.6	23.6	23.6	23.8	24.0
Carbon-intensity of energy	38.3	36.9	35.4	33.6	31.5	29.1
Energy-intensity of output	32.0	33.0	34.2	35.4	36.9	38.5
Scale of production	6.1	6.4	6.9	7.3	7.9	8.5
ROC						
Sector Composition	2.3	2.3	2.3	2.3	2.4	2.5
Carbon-intensity of energy	58.4	57.1	55.6	53.9	51.9	49.7
Energy-intensity of output	35.1	36.2	37.4	38.7	40.2	41.8
Scale of production	4.2	4.5	4.8	5.1	5.5	6.0

VII. CAVEATS AND SENSITIVITY ANALYSIS

The basic climate policy scenario assumes "best guess" values of willingness to pay for morbidity and mortality risk reductions in the Chinese population; based on the literature, it also uses central values of the relevant substitution elasticities, in line with those employed in the OECD's GREEN model. It goes without saying that, if WTP were significantly lower than the values assumed here, the ancillary health benefits of climate policy would be reduced proportionately. Likewise, if actual substitution elasticities in China were significantly lower than the values used here, the costs of adjustment to a carbon tax would be significantly higher. Needless to say, it also possible that the parameter values used here underestimate rather than overestimate actual Chinese elasticities, in which case the reverse is true: adjustment costs are actually lower than those shown here.

Inevitably there is a degree of uncertainty in any simulation exercise of this kind, but where the parameter values are based on "consensus" estimates from a range of studies most done elsewhere than in China, the uncertainty is that much greater. For this reason, it is advisable to consider a "low case" scenario, in which substitution elasticities are set to values at the low end of the range of plausible estimates. Ideally, the lower bound values would be based on a probability distribution, but this study uses an *ad hoc* procedure of halving the substitution elasticities used in the basic policy scenario. The results prove to be rather sensitive to the choice of elasticity values, with their halving resulting in welfare losses that converge to about three times those in the basic policy scenario. Even with these very conservative assumptions about substitution elasticities (and holding the *VSL* and *WTP* constant at levels of the basic policy scenario), ancillary benefits amount to between 50-60 per cent of the welfare costs of a carbon tax in the 5-10 per cent abatement range. When abatement increases to the range of 15-20 per cent, ancillary benefits still amount to 30 per cent of those costs. Thus, even a very risk averse climate policy maker could expect to recover a substantial fraction of welfare costs associated with a carbon tax through lower health costs and crop losses from pollution.

Other important parameters and exogenous assumptions that may significantly influence the results include the assumption about interregional factor mobility and the method of recycling the carbon tax revenue. The degree of factor mobility is thought to affect the overall flexibility of the Chinese economy's response to changing cost structures. Sensitivity tests, however, show that raising the elasticities on interregional labour and capital mobility from 0 to 1 and then to 2 hardly affects the basic results.

The method of carbon tax revenue distribution may affect the size of welfare losses in several ways, perhaps most obviously by augmenting or reducing existing tax distortions but also by altering the size of the pool of savings, hence investment. Results are in fact quite sensitive to the recycling assumption used. If, instead of merely holding real corporate savings constant, all carbon tax revenue goes to reducing corporate income tax, the welfare costs of the tax are considerably reduced and the "no regrets" abatement rate substantially increased. On the other hand, if all revenue is redistributed to households (with none bound for the corporate sector), then welfare costs are substantially higher than in the basic policy scenario (where as noted above revenues are split roughly 50:50 between the corporate and household sectors).

In sum, simulation results prove sensitive to two choices: *i)* of substitution elasticity parameters; and *ii)* of revenue recycling assumptions. The first is essentially a modelling choice that should be informed by the best available empirical estimates in the literature. The second is a policy choice that should be informed by considerations of dynamic efficiency but also of equity. As a last sensitivity exercise, one can examine a scenario in which low substitution elasticities are assumed together with full revenue recycling through offsetting reduction in corporate tax. Supposing that risk-averse policy makers were faced with considerable uncertainty about the "true" elasticity values, they might prefer to assume the lowest plausible values in formulating policy. It has already been shown that, with the standard recycling assumption used in the basic scenario and low substitution elasticities, there is no scope for "no regrets". If, however, policy makers were to consider this alternative recycling scheme, how would that affect the results?

Figure VII.1 presents the results for all China (the GD and ROC results, not presented, are broadly similar). The recycling assumption has a marked effect on welfare costs, hence on the net benefits curve and the "no regrets" range of CO_2 reductions. Relative to the basic policy scenario, the "no regrets" reduction in this scenario is almost five percentage points higher. The recycling assumption also has a sizeable impact on the absolute magnitude of net benefits at their maximum (in this case, at roughly a 10 per cent reduction from baseline). Whereas in the basic policy scenario, maximum net benefits are equivalent to roughly 0.04 per cent of GDP, in this sensitivity they treble to 0.12 per cent. In short, then, an appropriately chosen recycling scheme can significantly improve the prospects for a "no regrets" climate policy, even if substitution elasticities should prove in fact to be on the low end of the plausible range.

This result applies, however, only to the terminal year of the scenario period, 2010. The recycling of all revenue to corporations implies, however, that, in the early years, households would actually have to reduce consumption to support the higher investment rate. Thus, a full comparison of the two recycling assumptions would require a discounted present value analysis of the consumption path over the entire simulation period.

Figure VII.1. **Sensitivity: Low Elasticities, Corporate Tax Reduction**

(China-wide Welfare Analysis)
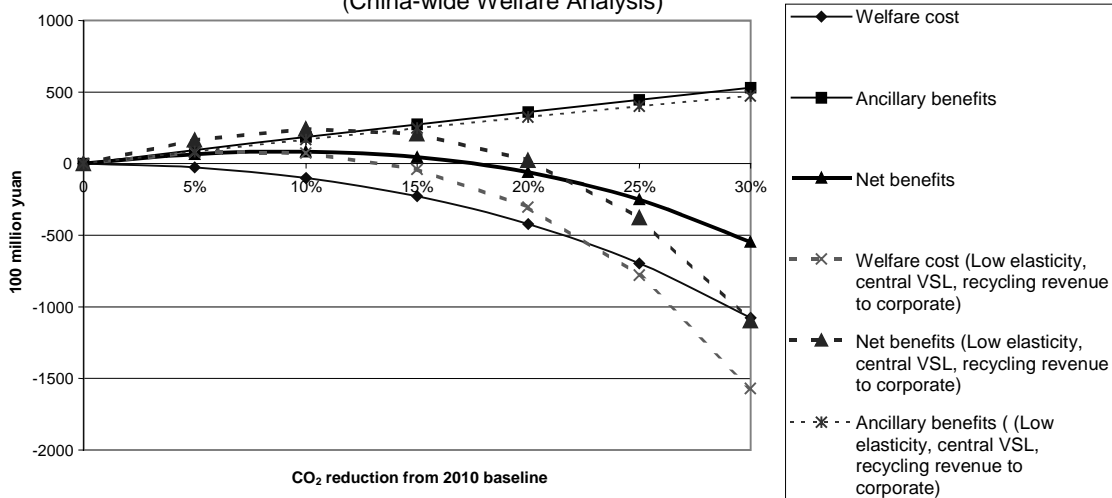
VIII. COMPARATIVE ASSESSMENT OF ABATEMENT COSTS AND ANCILLARY BENEFITS

Briefly the results presented here are compared with those of other studies that have analysed climate policy and its ancillary benefits, whether for China or other developing countries.

VIII.1. Other China Studies

There are two other major CGE modelling exercises for China (Zhang; Garbaccio, Ho and Jorgenson) that have analysed the economic impacts of climate policy and, in the latter case, the health benefits of a carbon tax.

Zhang (2000*a*) analyses the economic effects of China's reducing carbon emissions in 2010 by 20 per cent and 30 per cent below baseline, respectively. The results suggest rather large welfare effects of a carbon tax. In the 20 per cent reduction case, GNP is reduced by 1.5 per cent and welfare diminished by 1.1 per cent. These figures contrast with those in Table VI.1 of this paper, which suggest a 0.37 per cent GDP reduction and a similar welfare reduction for a 20 per cent carbon reduction. Two significant differences between Zhang's model and ours may contribute to the contrasting results: *i)* Zhang uses 1987 data, when oil imports were negligible, so oil does not constitute a major potential substitute for coal, whereas in the 1997 data used here it does; and *ii)* Zhang's model contains only 10 sectors to the 63 sectors here; with so few sectors, substitution possibilities are inherently limited in response to a carbon tax, raising the costs of adjustment as explained above.

Garbaccio *et al.* (1998) examine costs of reducing carbon emissions over 30 years by 5, 10 and 15 per cent below baseline. Given their offsetting of the carbon tax by lowering the income tax on enterprises, they find that in year 30 progressively larger emission reductions actually yield progressively larger increments to GDP, with a 15 per cent reduction adding almost 1 per cent to GDP in the final year (based largely on an increase in the investment rate of 2.35 percentage points). Based on the same revenue recycling assumption and essentially the same model as in their earlier paper, Garbaccio *et al.* (2000) arrive at similar if less dramatic results, with a 10 per cent carbon reduction from baseline yielding a 0.14 per cent increment in GDP by the year 2010.

This latter result is very similar to the one derived here from the revenue recycling sensitivity analysis where all carbon tax revenue is redistributed to corporations. In that case, there is an increment to 2010 GDP (relative to the baseline) of 0.1 per cent in 2010 from a 10 per cent carbon reduction, based on an increase in the investment rate of 0.3-0.5 percentage points over the simulation period.

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The later paper of Garbaccio, Ho and Jorgenson also estimates the reduction in health damages from a 10 per cent carbon reduction. Premature deaths and chronic respiratory illness are estimated to fall by around 7 per cent from baseline levels. In the case of deaths, this amounts to 36 000 premature deaths averted by 2010, or 255 lives saved per million tonnes of carbon reduction. The current results find a slightly lower figure of 210 premature deaths averted per MtC reduced, for a 15 per cent reduction below baseline (see Table VIII.1 below).

There are also a number of studies that use a bottom-up approach to examine options in China for energy-efficiency investments and other measures to control CO_2 emissions. Some use a simple project evaluation approach while others employ more formal optimisation models. In the former mode, London *et al.* (1998) examine 25 cases of energy-saving technology investments across a range of industrial sectors, from metallurgy to textiles to coal processing. They find that 13 out of 24 investments studied would have negative incremental cost per tonne of CO_2 reduction. The largest negative cost was that associated with installation of a reheating furnace for steel rolling, with improved sintering in aluminium production and computer monitoring of energy use in textile production also yielding negative costs per tonne CO_2 reduction. While the study clearly demonstrates some scope for "no regrets" emission reductions — even before considering ancillary benefits — it is impossible to say how large a share of China's CO_2 emissions could potentially be reduced by such investments.

Sathaye *et al.* (1996) find little scope for negative-cost carbon abatement in China, in contrast to Brazil and India where initial energy inefficiencies are found to be greater. This assessment finds corroboration in a study by Zou and Li (2000) of potential investments under the Clean Development Mechanism (CDM) of the Kyoto Protocol. They identify only two abatement options that may entail negative costs: coal-bed methane and afforestation. At the same time, several efficiency improvements in industrial boilers could yield carbon reductions at minimal cost (<$1/tC)[23]. Zhang (2000*b*) provides comparisons of the average energy intensity of certain energy-intensive industrial processes in China (circa 1994) with best practice abroad, suggesting that: *i)* in steel production China's energy use per tonne is three-quarters higher than Italy's; *ii)* in cement clinker production, it is roughly 60 per cent higher than Japan's; *iii)* in coal-fired power plants, it is one-fourth higher than the former Soviet Union; *iv)* in industrial boilers, thermal efficiency in China is between 70-88 per cent of that in the most advanced countries. Zou and Li estimate that, if China's industrial boilers were to reach the efficiency of those in the industrialised world, not only could CO_2 emissions be cut significantly (by perhaps 13 MtC), but so also could SO_2 and particulate emissions (by 400 000 tonnes and 250 000 tonnes respectively). The authors do not quantify the health impacts of these reductions, but the reductions are rather small as a share of estimated 1997 China-wide emissions of these pollutants — between 2 and 3 per cent. Viewed differently, the reduction in particulate emissions is only about a fourth as large as the reduction that would accompany a carbon tax set to achieve a 5 per cent CO_2 reduction by 2010.

VIII.2. Other Ancillary Benefits Studies for Developing Countries

Several studies for other developing countries examine ancillary benefits of climate policy or co-benefits of policies aimed at simultaneously controlling local and global air pollutants. A couple of these (for Chile and India) are predecessors to the current study, conducted under the auspices of the OECD Development Centre. In addition, the US Environmental Protection Agency has sponsored several such studies under its Integrated Co-Control Benefits Analysis Program (ICAP) — for Argentina, Brazil, Chile, China, Republic of Korea, Mexico and South Africa. Most of the studies undertaken by ICAP are "bottom-up" studies of specific technology or other control measures; only for China and Korea are economy-wide models employed. Besides China, the other country for which two studies exist — one from the OECD and one from ICAP — is Chile. As Table VIII.1 suggests, the two studies are broadly consistent in terms of their estimates of lives saved per unit of carbon reduction. The study by Dessus and O'Connor (2003) uses a CGE model, while that of Cifuentes *et al.* (1999) is a "bottom-up" analysis based on consideration of specific abatement technology options for Santiago.

Table VIII.1. **Comparison of Mortality Benefit Estimates from CO_2 Reduction**

Study	Lives saved per MtC reduction	Scenario Assumptions
Current study (2002)	210	China, 2010: 15% CO_2 reduction
Garbaccio *et al.* (2000)	255[a]	China, 2010: 10% CO_2 reduction
Bussolo and O'Connor (2001)	334	India, 2010 15% CO_2 reduction
Dessus and O'Connor (2003)	100	Chile, 2010: 10% CO_2 reduction
Cifuentes *et al.* (1999)	89[a]	Chile, 2020: 13% CO_2 reduction
Abt Associates (1997)	82[a]	USA, 2010: 15% CO_2 reduction

a)   The mortality benefits in these studies are solely attributable to particulate reductions, either PM-10 or PM-2.5.
Sources: O'Connor (2000); Bussolo and O'Connor (2001); authors' calculations.

A brief comparison can also be made of the present results to those of ancillary benefits studies for OECD countries. Burtraw and Toman (1997) report, based on a review of eight US studies, a mean value of ancillary benefits per tonne of carbon reduction of around $24 (virtually identical to Ayres and Walter, 1991), with a lower-bound estimate of $3 and an upper bound of $79. On average, the ancillary benefits amount to around one-third of abatement costs for moderate rates of abatement (up to 10 per cent from baseline). While the basic policy scenario for China shows significantly higher ancillary benefits in relation to welfare costs over a comparable range, the Butraw/Toman mean is consistent with the results of the sensitivity analysis assuming low substitution elasticities. There, the ancillary benefits sum to half of welfare costs at 10 per cent CO_2 abatement, falling to one-third of those costs in the 10-20 per cent abatement range.

IX. TRADE-OFFS BETWEEN LOCAL AND GLOBAL POLLUTION CONTROL?

As noted earlier, the starting point of the current study is the assumption that the Chinese government will take certain measures over the next decade to control local air pollution, independently of climate policy. It then weighs the added local benefits of climate policy against its costs in this context. The question arises: what if one were to focus the analysis on the control of local pollution by the most cost-effective means rather than as a side-benefit of greenhouse gas control? How would the choice of technology and the associated costs differ? One would expect that a carbon tax represents a "second-best" method of controlling local and regional pollutants like particulates and SO_2. What are the efficiency losses of using this second-best instrument *versus* a first-best one like a particulate (or sulphur) tax? This depends on how closely correlated the specific technology options are in terms of the two dimensions: local pollutant abatement cost-effectiveness and greenhouse gas abatement cost-effectiveness. Peng (2000) suggests that there may be some correlation between cost-effectiveness in the two areas. He finds that energy-efficiency improvements, especially in coal use, are a very low-cost means of reducing carbon emissions while also being the lowest-cost method of lowering PM-2.5 and SO_2 emissions (see Table 9 in Peng, 2000). The problem is that these measures reduce carbon emissions by only about 16 per cent and PM-2.5 by only 18 per cent from their baseline 2020 levels. Moreover, the next most cost-effective means of controlling local pollution bring no net reductions in carbon. Recall, however, that the preceding analysis finds a China-wide "no regrets" rate of CO_2 reduction (2010) in the 15-20 per cent range. So, over most of the relevant range of "no regrets" abatement, there would appear to be considerable overlap between the most cost-effective local pollution control measures and the most cost-effective CO_2 control measures. This result is consistent with the findings of Aunan, Fang *et al.* (2000, 2003) for Shanxi Province, China. Figure 4 of Aunan, Fang *et al.* (2000) shows a very strong correlation between the cost-effectiveness of specific measures in reducing PM-10 exposure and cost-effectiveness in reducing CO_2 emissions. For instance, co-generation ranks most cost-effective in both dimensions, while coal washing ranks least cost-effective in reducing PM-10 exposure and second most costly (after briquetting) in reducing CO_2 emissions. Staff Mestl *et al.* (2002) look at a range of "cleaner production" options in Taiyuan City, Shanxi. By far the most promising is combined cycle power production using blast furnace gas at a major iron and steel works. Not only does this investment yield a sizeable annual profit, but it yields by far the largest reductions in both CO_2 of any option (roughly a half million tonnes per year) as well as large indirect reductions in TSP and SO_2. In contrast, installation of an electric arc furnace would have by far the highest annualised cost and would yield among the smallest CO_2 reductions of any option (roughly 27 000 tonnes per year), while achieving only minor SO_2 reductions and modest TSP reductions.

X. POLICY CONCLUSIONS AND AREAS FOR FUTURE RESEARCH

The Chinese government has already approved and indicated its intention to ratify the Kyoto Protocol. While as a non-Annex 1 country it is not required to enforce quantitative restrictions on its greenhouse gas emissions, it is interested in understanding better the economics of slowing emissions growth.

This study seeks to redress an oversight in much climate economic analysis by explicitly incorporating estimates of the value of ancillary benefits. It finds scope for non-negligible "no regrets" abatement from a growth baseline once these are included. While other researchers (e.g. Zhang and Garbaccio *et al.*) have estimated the welfare costs of a carbon constraint in China, and the latter have provided estimates of the health benefits of carbon abatement, this is the first study to integrate an analysis of both the health and agricultural productivity effects of a carbon tax. The inclusion of agriculture makes an important difference to the assessment of "no regrets" abatement options.

Before considering agriculture, this study arrives at an estimated China-wide "no regrets" CO_2 abatement rate of roughly 5 per cent of baseline 2010 emissions. Including agriculture raises the "no regrets" abatement rate to over 15 per cent. As a share of GDP, the agricultural productivity benefits are almost as large as the health benefits. Net benefits of climate policy with agricultural effects included are maximised with a 10 per cent emissions reduction from baseline. The regional decomposition of the results shows that the agricultural productivity benefits are relatively modest in GD but quite large in ROC, reflecting the latter's greater agricultural orientation and specialisation in ozone-sensitive crops. Since the benefits of improved agricultural productivity are captured largely by farm households who are on average poorer than the urban households enjoying benefits of cleaner air, inclusion of agricultural impacts can have an important bearing on the analysis of distributional implications of climate policy. Before considering agricultural effects, rural households in ROC experience welfare losses even at low levels of carbon abatement; once those effects are considered, they enjoy welfare gains up to a 10 per cent abatement rate. In short, while a purely health-based measure of ancillary benefits would tend to show such policy to be urban-biased, a broader definition of benefits alters the picture considerably.

While different pollutants figure in the two major sorts of local/regional pollution damage evaluated here — particulates and SO_2 are the major contributors to health damage, NO_x the major contributor (along with VOCs) to ozone formation and crop damage — emissions of both sets of pollutants move roughly proportionally with CO_2, at least in response to a carbon tax. For a tax achieving a 15 per cent CO_2 reduction by 2010, the percentage reductions in TSP and NO_x are virtually identical, i.e. around 10 per cent each, while the percentage reduction in SO_2 is very close to that for CO_2 itself, at 15.5 per cent.

While the analysis here provides no basis for supposing that climate policy represents a first-best way of addressing local air pollution problems, there is also little reason to suppose that, for modest reductions in local pollution, it imposes significant additional costs relative to a better-targeted instrument. For large-scale reductions in local air pollutants like particulates, SO_2, and NO_x — i.e. on the order of 80-95 per cent — there can be no substitute for the sorts of end-of-stack (or tailpipe) controls that are now widely used in industrialised countries and increasingly in middle-income developing countries.

The analysis shows that the results are not particularly sensitive to assumptions about the mobility of productive factors between GD and the ROC. On the other hand, the choice of substitution elasticities among productive inputs and the rule used for redistributing carbon tax revenue do substantially influence the results. Costs are substantially higher when elasticities are halved, due to much lower flexibility in economic structure. Even with low elasticities, however, a recycling scheme that returns carbon tax revenues to enterprises through reduced corporate tax allows for significant "no regrets" abatement. In short, there would appear to be something like a double dividend associated with the use of the revenue to lower corporate tax (consistent with Garbaccio *et al.* 2000), working through a boost to investment and growth, though the welfare effects of near-term consumption foregone to achieve higher end-period consumption would also need to be considered.

What is not attempted here is to examine alternative specifications of abatement cost dynamics, beyond altering assumptions about substitution elasticities among various productive inputs. This is an area ripe for further research, since targeted investment in R&D on low-carbon alternatives to current fossil-fuel-based energy sources can have important consequences for the long-run economics of climate change mitigation (cf. Papathanasiou and Anderson, 2000). In the timeframe considered here, even if such R&D were to bear fruit in important technical breakthroughs, diffusion of any new technologies throughout the energy system would necessarily be limited, given the slow turnover of the energy-producing capital stock.

A few other limitations of the analysis deserve mention. First, the use of a CGE model with competitive price adjustment can obscure certain real-world adjustment costs of climate policy. For example, rather than experiencing smooth wage adjustment to maintain full employment in the face of structural change, the real economy may be characterised by persistent unemployment in certain sectors (e.g. those that are especially energy-intensive). Unemployment, in turn, may bring with it welfare costs to workers and households beyond those associated with a fall in wages. Furthermore, the costs of adjustment may be geographically localised, with especial hardship experienced by certain cities and provinces. The preceding analysis showed how terms-of-trade movements affect the welfare gains (losses) of ROC *versus* GD; a far more disaggregated model would be needed to identify the most vulnerable regional economies, but a fairly good predictor would be the degree of dependence of a given provincial or municipal economy on energy-intensive industries and, in particular, on coal.
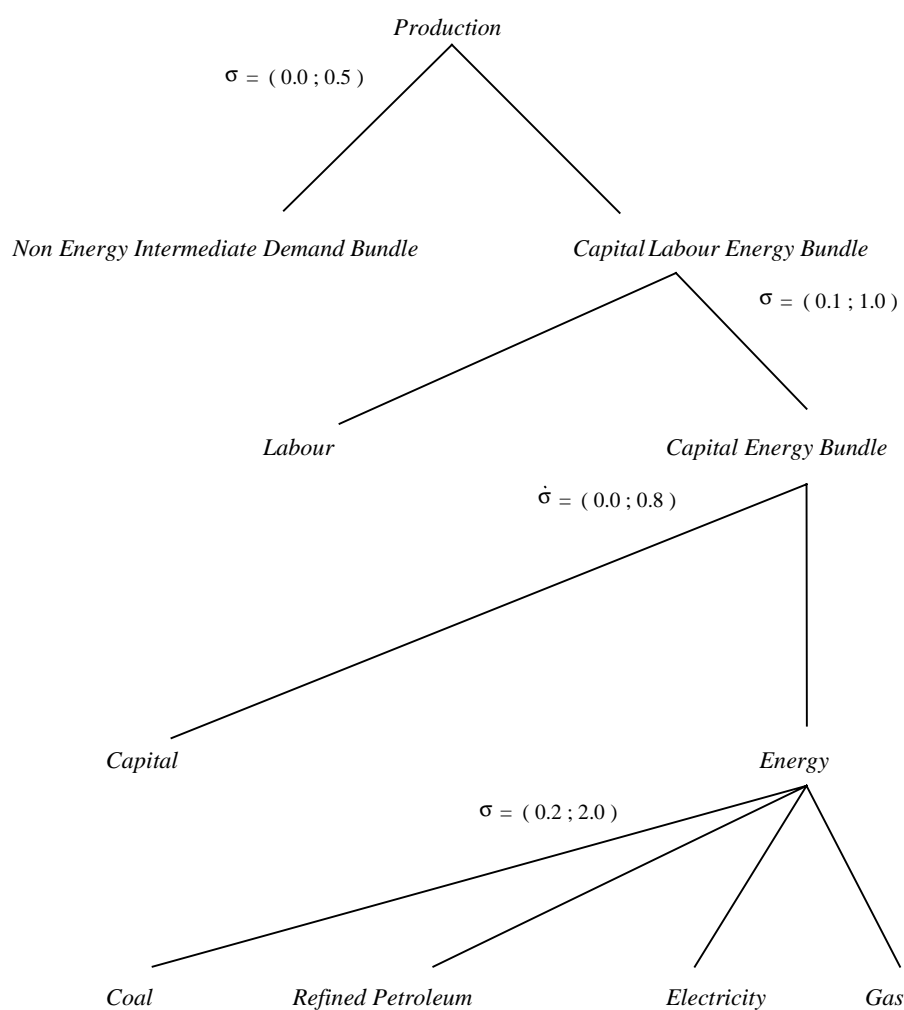
Second, and already noted, an economy-wide model — even one with two separate regions — is difficult to wed to a model of local air quality in major metropolises. It has been necessary to use some fairly gross simplifications in modelling the link from emissions to local air quality, whether to examine health impacts or to consider crop productivity effects. This sort of analysis would benefit from complementary "bottom-up" modelling of technology substitution options and costs in various regional or local economies (in the spirit of the study by Cao, 2002, for Guiyang).

Third, the assessment of health impacts could be further refined by incorporating estimates of age-cohort-specific mortality and morbidity changes. The epidemiological literature points to especially strong effects of air pollution exposure on high-risk groups, both in terms of age and in terms of health status and other complicating factors. This would be worthwhile, however, only if one were able to link these age-specific impacts to age-specific estimates of willingness to pay for risk reduction. (Even in OECD countries, few such studies currently exist, though Krupnick, 2001 reports contingent valuation survey results for a Canadian age-differentiated sample population.)

Fourth, considerable uncertainties remain about the true willingness to pay of Chinese individuals for reductions in pollution-related health risks. The valuation of the health benefits of climate policy is highly sensitive to the value of a statistical life, so the results are bound to be subject to some uncertainty that can only be resolved by careful empirical estimation of China-specific *VSL*s. This work remains to be done. Also, a further refinement would be to endogenise the health impacts of pollution on labour supply and on the utility of consumption.

APPENDIX 1

Appendix Figure A1: Production Nesting

Production

σ = (0.0 ; 0.5)

Non Energy Intermediate Demand Bundle   *Capital Labour Energy Bundle*

σ = (0.1 ; 1.0)

Labour   *Capital Energy Bundle*

$\dot{\sigma}$ = (0.0 ; 0.8)

Capital   *Energy*

σ = (0.2 ; 2.0)

Coal   *Refined Petroleum*   *Electricity*   *Gas*

Note: first elasticity value in parentheses applies to installed capital stock; second to new investment.

APPENDIX 2

The OSLO-CTM2 is a global 3-dimensional off-line chemical tracer model (CTM) that uses pre-calculated fields of winds and other physical parameters to simulate the chemical turnover and distribution of chemical species in the troposphere (Sundet, 1997). The meteorological input data for the model have been made specifically for this model by generating a series of 24-hour forecasts for the year 1996. An extensive set of data is sampled every three hours, including convective mass fluxes. The CTM can be run variable with resolution up to 1.87°x1.87°, however, in this study a horizontal resolution of 5.6°x5.6° is used to limit the amount of CPU-time needed. The vertical resolution is also determined by the input data and the current model version includes 19 levels from the surface up to 10 hPa.

The transport of chemical species by the mean winds is calculated by the very accurate second order moment method (Prather, 1986). Rapid vertical by small-scale convective cells (e.g. thunderstorms) is based on the Tiedtke mass flux scheme (Tiedtke, 1989). The chemical scheme includes 55 chemical compounds and 120 gas phase reactions in order to describe the photochemistry of the troposphere (Berntsen and Isaksen, 1997; Berntsen and Isaksen, 1999). Photodissociation is calculated on-line, following the approach described in Wild *et al.*, (2000). An overview of the global natural and anthropogenic emissions of NO_x, CO and non-methane hydrocarbons (NMHC) is given in Table A1. Methane concentrations are fixed to 1700 ppbv in the Southern Hemisphere and 1790 ppbv in the Northern Hemisphere. The lightning amounts to 5 Tg(N) yr^{-1} and the emissions are coupled on-line to the convection in the model using the parameterisation proposed by Price and Rind (1993) and the procedure given by Price *et al.*, 1997*a,b*). Aircraft emission from the NASA-92 inventory is used (Baughcum *et al.*, 1995) and scaled up to 0.7 Tg(N) yr^{-1} thought to represent emissions for year 2000. Deposition of trace gases is based upon Wesley (1989) and the boundary layer mixing is treated according to the Holtslag K-profile scheme (Holtslag *et al.*, 1990). Influence of stratospheric ozone is estimated using a synthetic ozone approach (McLinden *et al.*, 2000) where the ozone flux from the stratosphere is prescribed, but the model transport generates an ozone distribution that varies with time and space.

Table A1. **Overview of Global Natural and Anthropogenic Emissions of NO_x, CO, and NMHCs in the OSLO-CTM2**

NO_x (Tg(N)/yr)	
Fossil fuels	31.7
Biofuels	1.3
Biomass burning	7.1
Aircraft	0.7
Lightning	5.0
Soils	5.6
Stratosphere	Approx. 0.5
TOTAL	51.9
CO (Tg/yr)	
Fossil fuels	650
Biomass burning	700
Oceans	50
Vegetation	150
TOTAL surface sources	1 550
NMHC Tg(C)/yr	
Isoprene	220
Acetone	30
Fossil fuels	124
Biomass burning	32
TOTAL	406

NOTES

1. That having been said, the period 1996-98 saw a slight decline in China's CO_2 emissions before they began rising again in 1999. By 2000, they had just recovered to their 1996 level; based on reference approach to emissions estimation in IEA (2002).

2. The average sulphur content of Chinese coal is reported to be 1.35 per cent; IEA (1999).

3. See "The New Hot Fuel in China", *Far Eastern Economic Review*, 14 September 2000.

4. China currently has the highest level of hydropower development activity in the world. In addition to the Three Gorges dam (18.2GW), the 3.3GW Ertan and the 1.8GW Xiaolangdi hydroelectric projects are also under construction. In toto, schemes with a combined capacity of 50GW are currently under construction, which will double the existing capacity in the country. The construction of an additional four large-scale projects will commence shortly: Xiluodo (14.4GW), Xiangjiaba (6GW), Longtan (4.2GW), and Xiaowan (4.2GW). A further 80GW of hydropower is planned, including 13 stations along the upper reaches of the Yellow River, and 10 stations along the Hongshui River (International Energy Outlook 1998, "Hydroelectricity and other renewable resources": http://www.eia.doe.gov/oiaf/ieo98/hydro.html).

5. The Class 2 annual average concentration limit for TSP is 200 μ/m^3, while that for PM-10 is 100 μ/m^3; for SO_2 it is 60 μ/m^3, and for NO_x 50 μ/m^3; World Bank (2001).

6. WHO (1999) estimates that 325 000 Chinese deaths in 1998 were caused by acute lower respiratory infections.

7. Inhibited photosynthesis, respiration and nutrient uptake can lead to reduced agricultural crop yields; Aunan, Berntsen, and Seip (2000).

8. A caveat: the studies of ozone effects on crop yields do not include cultivars found in China.

9. Calculated as sum of differences between hourly mean concentration and 40 ppb for hours when that mean is greater than 40 ppb (expressed as ppb-h).

10. This refers to the output of both state and non-state industrial enterprises with annual revenues of over 5 million yuan; *China Statistical Yearbook 2001*.

11. While recognising that in principle there can be ancillary costs as well as benefits associated with climate policy, hereafter "benefits/costs" is abbreviated to "benefits"; one can always think of costs as negative benefits, and *vice versa*.

12. By this assumption, products are assumed to be differentiated by region of origin, with less than perfect substitutability across regions. Thus, the cross-price-elasticity between car imports and domestic cars is not infinite. Neither is that between car imports from different regions.

13. In still another formulation, the EV of a price increase is the quantity of money income which, if taken away from the individual before the price rise, would leave him or her at the same level of utility as if the price rise had occurred (based on Perman *et al.* 1996, p. 257)

14. VSL can be alternately expressed as the value of a premature death avoided and is estimated by individual willingness to pay (WTP) for small reductions in the risk of premature death from specific

causes (whether on-the-job accidents or pollution-related illness). Algebraically, *VSL = Value of Δrisk/ Δrisk*. So, for instance, if the average WTP for a 1/100 000 reduction in the risk of premature death (say, associated with a $10\mu/m^3$ reduction in PM_{10}) is \$50, then the *VSL* implied by that WTP is \$50/(1/100 000), or \$5 million.

15. The analysis abstracts from interregional pollutant transport. In effect, then, all emissions originating from Guangdong sources are assumed to have their impacts only in Guangdong, and the same for the rest of China.

16. This is done because the emissions for "rest of China" are aggregate emissions over all the cities and provinces (except Guangzhou/Guangdong). Thus, one would finish with grossly underestimated dispersion coefficients if one were to calibrate a dispersion model using total "rest of China" emissions and average concentrations over the area of a city of average size.

17. It should be noted that studies of Pakistan rice cultivars have found substantially greater impacts of ozone on rice yield than what is indicated by the studies reported in EC (1999).

18. SUM06 is another ozone metric, applying a threshold of 60 ppbv, see USEPA (1996) for details.

19. See the 2002 study by the University of Petroleum-Beijing and the Pacific Northwest National Laboratory of the United States on China's emerging natural gas market (UPB-PNNL 2002).

20. The estimated reductions in CO_2, SO_2 and black carbon emissions during the second half of the 1990s (Streets *et al.*, 2001) are not fully captured in the baseline, which projects forward emissions from the base-year, 1997, adjusting emission coefficients downward only gradually over the entire simulation period. There remains some question about the durability of the emission reductions since 1996, as they are due in part at least to economic impact on China of the Asian financial crisis of 1997, though domestic reforms have also played a part.

21. The crop sectors are the lowest in terms of marginal labour productivity, as measured by the sectoral wage.

22. The growth rate effect implies that a different carbon tax schedule applies to the "with-crop" scenario, but the actual tax rates differ only very slightly from those in the "without-crop" scenario.

23. Even so, the authors suggest that, given imperfect capital markets, many firms may have difficulty finding the capital for the initial investments required.

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