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Benefits and costs to China of a climate policy

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ABSTRACT. In future agreements to cut greenhouse gases, a Chinese commitment will probably be essential. Committing for China is easier if the cost is low and the benefit to China is high. Using a new CGE-model of the Chinese economy we discuss the cost and benefit to China of taking on a climate commitment. We argue that a climate commitment gives significant ancillary benefits to China since associated particle and

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NO_x-reductions improve public health and increase agricultural yields. The model of impact on agricultural yields is a novel feature of CGE-models. Comparing benefits to economic costs produces striking results. We find that China may reduce its CO_2-emissions by 17.5 per cent without suffering a welfare loss. Half of the benefit originates in the novel agricultural model. We also discuss the distributional impact of a climate commitment. In general the distributional impact is not averse.

1. Introduction

China has gradually become a vital participant in international efforts to address climate change. China is a signatory to the United Nations Framework Convention on Climate Change (UNFCCC). At the World Summit on Sustainable Development held in Johannesburg, South Africa, in August–September 2002, the Chinese government announced its approval of the Kyoto Protocol. Qualifying as a non-Annex 1 party, China would not be bound in the initial commitment period (2008–2012) to any quantitative restrictions on its greenhouse gas (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 the future. The release meeting of the Initial National Communication of the People's Republic of China on Climate Change was held in Beijing on 9 November 2004.

Next to the United States, China is the largest country source of GHG emissions; in 2002, US carbon dioxide (CO_2) emissions from fuel combustion totalled 5.8 billion tonnes to China's 3.8 billion tonnes, or 23.3 per cent and 15.3 per cent of global emissions, respectively (WRI, 2006). Fossil fuel combustion is not only the main source of local air pollution, but it is the main source of GHG. Measures that alter the amount of different fuels consumed will normally affect emissions of both air pollutants and GHG. For example, more efficient conversion of coal into thermal energy will lower, per unit of power generated, emissions of sulphur dioxide (SO_2), total suspended particulates (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 present paper seeks to delineate that scope. The starting point is an assumption that economic growth takes precedence at this stage of China's development, that efforts to control local and regional air pollution are continuing, while action to control greenhouse gas emissions remains a lower-order but unavoidable concern. Given the economic and environmental baseline to 2010, the study discusses what difference an explicit climate policy – modelled as a tax on carbon – would make to growth on the one hand and to domestic environmental burdens on the other. From the perspective of a Chinese government policymaker, the question the study seeks to answer is:

> Supposing we were to introduce a carbon tax to encourage a less carbon-intensive development path, how much of the economic cost could be offset by ancillary benefits that accrue from lower levels of air pollution and improved human health and agricultural productivity?

In actual fact, a carbon tax is probably not on the table in China today. However, the idea of limiting carbon emissions is discussed in the implicit form of energy efficiency, cleaner production, industrial innovation etc. In 2006, China implemented a Renewable Energy Law that offers subsidies to power and grid access, soft loans, access to a renewable energy development fund etc. A Cleaner Production Law became effective 2003. The recently approved '11th five year programme' states a policy goal to reduce energy consumption per unit GDP by 20 per cent. The regulation and policy goal have in common that they implicitly increase the shadow price of carbon. Their effects will therefore be similar to the effects of an actual carbon tax. In the future, a carbon tax or quota may be explicit. This paper and others may further the debate about a future carbon tax or quota in China.

To analyse the question of how much of the economic cost could be offset by ancillary benefits, the paper makes use of a time-recursive computable general equilibrium (CGE) model with environmental and distributional features. The CGE-model is available at the Development Research Center of China's State Council, and the OECD Development Centre. The environmental features are three: One feature is a description of how emissions of a range of GHGs and air pollutants are affected by perturbations in the economy, in our case a carbon tax. As a second feature, environmental impacts on public health are modelled, starting from emissions generated by the economic model, through impacts on concentration levels and population exposure. The third feature, which is novel in the CGE literature, is a description of impacts on agricultural productivity through changes in tropospheric ozone formation and associated changes in exposure of crops, due to reduced NO_x emissions. Agriculture is still by far the most important employer in China, and avoided crop losses due to reduced levels of surface ozone will affect the incomes of hundreds of millions. To describe the link between emissions, surface ozone, and agricultural productivity we make use of a three-dimensional off-line photochemical tracer/transport model (CTM) that is available at the University of Oslo (Berntsen and Isaksen, 1997).

In addition to its detailed environmental description, our model emphasizes distributional impacts. In China today the most important distributional impacts are related to the urban–rural dimension and the regional dimension. The urban–rural dimension is important since urban dwellers earn three-four times more than rural dwellers, and the gap is increasing (NBS, 2005). The regional dimension is important since people in the coastal provinces earn up to ten times more than people in poor inland provinces. For this reason, we have constructed a two-region, urban–rural model that distinguishes the Guangdong economy from the rest of China, and specifies income generation for urban and rural households in both Guangdong and the rest of China.

2. Previous literature

Nordhaus (1977), in a global model, was among the first to use CGE-models for analysing climate commitment. By that time in the late 1970s national CGE-models had existed for quite some time. The first CGE-model is generally recognized to be that of Johansen (1960). By 1990, the models had been extended to include environmental emissions (Bergman and Lundgren, 1990) and ancillary benefits of lower emissions (Glomsrød *et al.*, 1992). In early models and even many current ones, ancillary benefits are added to the material utility function of consumers, implying no impact on economic behaviour.

Recent models have allowed environmental quality to influence economic behaviour, although not via the agricultural feedback that we propose. Global models have focused on the climate feedback (CO_2-emissions causing lower economic activity) and national models have focused on the feedback from emissions to air to labour supply (e.g., Vennemo, 1997).

CGE-models have been used to study environmental issues in developing countries. The OECD Development Centre, for instance, has promoted CGE-models in India (Bussolo and O'Connor, 2001) and Chile (Dessus and O'Connor, 2003) besides the current paper. The first environmental CGE-models in China for an international audience are Xie (1996) and Zhang (1997). Garbaccio *et al.* (1999) and Zhang (2000) analyse the economic costs of a Chinese CO_2-commitment, while Glomsrød and Wei (2005) analyse the environmental implications of coal washing. These models of China are reasonably close to our model, but lack the associated environmental and distributional dimensions.

A model of China that does include the environmental dimension is Garbaccio *et al.* (2000), who analyse the health benefits of better air quality. Our modelling of health effects is similar to theirs. A difference between the models is the feedback on agriculture. As we shall see, the inclusion of agriculture makes an important difference to the assessment of no regret abatement options. Another difference is our focus on distributional impacts and the inclusion of two provinces and two income categories. In addition there are differences relating to the number of industries (Garbaccio *et al.* have 29, we have 61 in two regions), base-year (1992 versus 1997 in our model) and modelling of economic competition (Garbaccio *et al.* assume profit maximization subject to plan output quotas, while we do not consider the plan output quotas since they are not predominant in the Chinese economy anymore). These latter differences are not decisive for model results, however. We compare our results to those of Garbaccio *et al.*, as well as other papers, in the concluding section.

There is an increasing literature estimating the ancillary benefits of greenhouse gas mitigation in China from the bottom up. Examples include Vennemo *et al.* (2006) on the ancillary benefits of China's energy-related CDM-potential; Mestl *et al.* (2005) on ancillary benefits of six actual cleaner production investments in Taiyuan City; and Aunan *et al.* (2004) on a range of clean coal technologies in Shanxi Province. This literature builds on a much larger bottom–up literature from the developed world, as well as research on the individual components of the impact pathway.

3. Air pollution and its effects in China

China's economic growth continues to be among the fastest in the world: Between 1978 and 2003 its real per capita GDP rose on average 8.1 per cent annually (NBS, 2005). 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 (Yande *et al.*, 2004).

The air quality of China's major cities is poor. In 1999, only one-third of China's 338 monitored cities were in compliance with national residential ambient air quality criteria (World Bank, 2001). As is documented by many studies (World Bank, 1997, is a much cited example), the breach of air quality standards for particles and SO_2 amounts to a major environmental health risk from outdoor air pollution.

In addition, the rapid growth in vehicular emissions results in rising levels of nitrogen oxide (NO_x) in many cities, e.g. Beijing, Guangzhou, and Shanghai. Rising NO_x levels increase concerns over tropospheric ozone (O_3). Tropospheric ozone is a photochemical reaction involving NO_x and volatile organic compounds (VOCs) (Colls, 1997). The effects of tropospheric ozone exposure on plant tissues and crop yields are well established, and the scientific literature is reviewed in USEPA (1996) and EC (1999: section 13.4). Since NO_x and VOCs are involved, surface ozone levels are sensitive to pollution from thermal power plants, industrial boilers, motor vehicle exhaust, gasoline retail outlets, and N-fertilizer 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. Aunan *et al.* (2000) use an ozone model for China (calibrated to 1990 data) and exposure-response functions from the literature to estimate Chinese crop losses in 1990 and project them to 2020. The losses vary depending on which dose-response function is used, but in general the biggest predicted losses, in per cent of total production of the given crop, are for spring wheat and soybeans due to their sensitivity and the concurrence of peak levels of ozone and the growth season of these crops. In absolute terms (in tons), the crop losses are biggest for wheat and rice.

To summarize, particles and SO_2, with associated health risks constitute a well-documented, important linkage between economic activity and environmental problems. Emissions of NO_x contribute to ozone formation, which reduces agricultural yields and constitute another important linkage. Below we explore how a carbon tax interacts with these two linkages.

4. The CGE-model

4.1. The economic model

In order to study how a carbon tax interacts with the economic–environmental linkages just described, we use a CGE-model. A brief description of some model features is presented below. For a detailed description of the model see O'Connor *et al.* (2003).

The economic model is a standard time-recursive growth model operating under so-called neoclassical closure, i.e. investment is determined by savings. Savings consist of public savings,[1] foreign savings, and corporate savings, all of which are exogenous, in addition to household

[1] The so-called off budget public sector, which is a characteristic of the Chinese system, has been merged with the public sector.

savings. Household savings are determined in an Extended Linear Expenditure System (ELES) model that also determines consumption demands for individual goods. Since urban and rural households are distinguished, the model has four households, two in each region.

There are 61 sectors of production in the model. Production exhibits constant returns to scale. Production input choices are determined in nested constant elasticity of substitution (CES) functions. Energy enters at the lowest nest and faces substitution with capital first, then with labour and finally with intermediate inputs. The model distinguishes between installed and new capital, and substitution is much more difficult (elasticities of substitution are lower) for installed capital. The parameters of demand and production are determined by literature search and mainly taken from the OECD GREEN model (Lee *et al.*, 1994).

The 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.

Welfare measurement

In a time-recursive model, it is not clear how to do welfare assessments. Consideration of the full dynamic path is one option, but the ELES system is not suited for dynamic welfare measurement. We follow several other authors in studying CO_2-emission reductions in a focal year, in our case 2010, and assessing impacts as of that year. Results in a focal year give a snapshot of the impacts of CO_2-commitments. A snapshot in a focal year is a potential restriction compared with the dynamic path. In this model, impacts are similar between years since we assume that foreign savings, public savings, and corporate savings all are fixed between scenarios. That ties up a significant part of the capital stock.

Although not formally correct in a dynamic setting, we find it helpful to couch our discussion of focal year results in a welfare theoretic language. We estimate the focal year welfare change from climate policy as the sum of three parts, as shown in the following equation

$$\Delta W = \underbrace{e(p, u^{**}) - e(p, u)}_{material\ welfare\ loss} + \underbrace{e(p, u^{*}) - e(p, u^{**})}_{ancillary\ benefit\ agriculture} - \underbrace{(D^{*} - D)}_{ancillary\ benefit\ health}, \quad (1)$$

where p is focal year prices, u is utility, e the expenditure function (derived from LES, not ELES), and D the willingness to pay to avoid the health burden from environmental damage. The star exponent denotes the with-policy state. The two-star exponent denotes the with-policy-and-without-ozone-change state. The first part ($e(p,u^{**}) - e(p,u)$) is the standard equivalent variation (EV) measure of *material welfare* change, interpreted – in the case of a material welfare-reducing policy like a carbon tax and given no decrease in ozone formation – as the maximum amount the individual would be willing to pay to forego the change. This part is a welfare loss. The second part ($e(p,u^{*}) - e(p,u^{**})$) is the impact on material welfare of decreased ozone formation. This part is positive in the case of a carbon tax since decreased ozone formation allows higher agricultural yields and higher material welfare. The third part ($-(D^* - D)$) represents the health-related benefit. If public health improves as expected with a carbon tax, the expression is positive. Total ancillary benefits of a climate policy consist of the health-related benefit plus the impact of lower ozone formation on yields.

Equation (1) is our formula for benefits and costs of Chinese climate policy. The CO_2-abatement level where benefits balance costs is called the maximum 'no regret' abatement level of CO_2 and solves equation (2)[2]

$$e(p, u) - e(p, u^{**}) = e(p, u^*) - e(p, u^{**}) + (D^* - D) \qquad (2)$$

Emissions

The linkage between the economy and the environmental burden depends on the modelling of emissions. Emissions in this model are determined by the intermediate or final consumption of polluting inputs, mostly fossil fuels. In addition, certain industries have emissions not linked to inputs, but related instead to their output levels (e.g., fugitive emissions, as with natural gas leakage and volatile organic compounds). Emission factors associated with each type of consumption and production were originally derived from the World Bank's IPPS project, which used toxic release inventory (TRI) data to establish sectoral emission factors for the United States (see Hettige *et al.*, 1995).[3] The model assumes an exogenous reduction in energy-related emission factors over the simulation period (factors are reduced by 2.5 per cent per annum for TSP and 1.5 per cent per annum for other pollutants). A reduction in pollution factors of this magnitude is in accordance with historical evidence and with recent projections from IIASA (Cofala *et al.*, 2005). Without the reductions, model emissions would have grown at historically unprecedented rates, and environmental benefits of a climate commitment would have been unrealistically high. Given the reductions, a reasonable interpretation of model results is that they show the benefit of a carbon policy in a situation of increasingly stringent traditional environmental policy.

There are four primary air pollutants considered in the analysis of climate policy and its environmental benefits: CO_2 – the main greenhouse gas, TSP, SO_2, and NO_x. In addition, the model reports emissions of VOCs and carbon monoxide (CO).

[2] We use the term maximum 'no regret' for a policy that comes at zero cost compared with the pre-policy status quo. The maximum no regret policy is not the welfare maximizing policy, see figure 3. In other words the maximum no regret policy is a policy to induce all no regret options, after which we have reached the welfare maximum; plus inducing additional options that wear down the welfare improvement that was accumulated by the first options.

[3] Since the emission coefficients in the model are originally derived from US data, it was necessary to calibrate them to Chinese data. Scaling all emission factors, model-generated emissions were calibrated to an emissions inventory contained in the *China Environmental Yearbook*, and the IEA's annual *CO_2 Emissions from Fuel Combustion* (IEA, 2000 and 2002). For VOCs and CO no independent check was possible. However, VOCs and CO play no role in determining ancillary benefits.

4.2. Modelling dispersion and impacts

The health component

Addressing the environmental health effects requires a reasonably reliable way of mapping sectoral emissions into impacts on the health of affected populations.

First a *Stack–Height–Differentiated Dispersion Model* is employed, and the approach pursued 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). Application of this dispersion model in conjunction with the CGE-model requires some simplifying assumptions. First, we focus on dispersion to population centres. In the case of Guangdong, this means specifically the capital city, Guangzhou. For the 'rest of China' region, the concentrations used to calibrate the model 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. Since we emphasize population centers we do not include the rural populations affected by changes in air pollution, and our health impact estimates will be conservative. The major impact on rural areas occurs through formation of secondary particles, which together with fine fraction primary particles have a lifetime in the atmosphere long enough to travel across large distances and contribute to regional pollution. More detail on the modelling choices is given in O'Connor *et al.* (2003).

The health damage assessment is mainly based on Chinese epidemiological studies of dose-response relationships between air pollutants and health effects. Table 1 reports the dose-response coefficients for various health endpoints, including an uncertainty interval. The coefficients are based on a limited number of studies, but are generally in line with other studies. For SO_2 and premature deaths, coefficients from Chinese studies vary between 0.02 per cent and 0.19 per cent increase per $\mu g/m^3$, while the coefficient suggested in Aunan and Li (1999) is 0.12 per cent. In table 1, the impacts are given in absolute terms by combining relative functions with the observed or estimated frequency of the health effect end-points, yielding the absolute annual increase in cases per million inhabitants per $\mu g/m^3$.

Valuation of health impacts

Valuing environmental benefits of a carbon policy requires a valuation of environmental health end-points. Valuation of mortality risk is of particular interest. Economists tend to consider mortality risk similar to any good whose value has an opportunity cost interpretation. The problem is to estimate the value of mortality risk precisely. That problem is especially acute in China, where morality risk valuation studies are just beginning to appear. We use an estimate of the value of a statistical life (VSL) from the Taiwanese hedonic wage study of Liu and Hammitt (1999), and adjust it to mainland Chinese 1997 income levels assuming an income elasticity of willingness to pay equal to one. The VSL estimates are, for all China, RMB 355,000, for Guangdong, RMB 572,000, and for the rest

Table 1. *Dose-response relationships[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 day loss (WDL)	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)
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)

Notes: [a]Based on observed frequencies from 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.
Source: Aunan and Li (1999).

of China, RMB 347,000.[4] We could argue that the VSL-estimates should increase over time in line with income growth. We have chosen not to do that since VSL-transfer and application in a new context is an uncertain science indeed. In effect we are assuming an income elasticity greater than one when comparing Taiwan 1999 to mainland China 2010. An income elasticity greater than one has support in the data on between-country

[4] One RMB approximately equals 1/8 USD or EURO.

Table 2. *Estimated monetary values of unit changes in various health endpoints (RMB, 1997 price)*

Health endpoint	*Guangdong*	*Rest of China*
Deaths	572,000	347,000
Infant deaths	572,000	347,000
Outpatient visits	118	72
Emergency room visits	118	72
Respiratory hospital admissions	5,260	3,190
Work day losses	24	15
Acute respiratory symptoms in children	9	5
Acute respiratory symptoms in adults	9	5
Chronic respiratory symptoms, cases	70,800	43,000
Asthma attacks	42	25

Sources: Guangzhou Action Plan (2000) for morbidity. See text for mortality.

health expenditures and may be a reasonable if conservative assumption. By consequence, the estimated health benefit is more conservative than would have been the case otherwise, both compared with agricultural benefits and compared with costs of mitigation. Central values for the various health endpoints considered in the ancillary benefits analysis are given in table 2.

The agriculture component

To model impacts on agriculture we proceed in two steps. First, we run the photochemical tracer/transport model on exogenous pollution scenarios. From results of the simulations we estimate a reduced-form relation between agricultural yields and NO_x. Finally, we insert the reduced-form relation in the economic model. The result is a reduced-form version of the tracer model that is fully integrated in the economic model. While crude, we believe that our procedure produces crop loss estimates of the right order of magnitude and is relevant for discussing man-made pollution through the ozone chain. Since this is a novel feature, we discuss it in some detail, and refer to O'Connor *et al.* (2003) for even more detail.

Pollution scenarios and crop losses using the tracer model

To assemble the reduced-form equations we start out by surveying crop production. The annual production of crops for which dose-response functions are available is shown in table 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. A static crop profile is assumed, and average figures are used for the period 1995–1997.

We apply dose-response functions to the crops. The dose-response functions (Weibull functions) are from EC (1999) and, for rice, from Adams

Table 3. *Crops grown in China (total) and in Guangdong*

Crop	*Average prod. in China 1995–1997 (1000 t)*	*Average prod. in Guangdong 1995–1997 (1000 t)*	*Guangdong share of Chinese prod. (%)*
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[1]	0.1
Sorghum	4,690	1.3	0.03
Corn	114,590	–	
Cotton	4,524	–	

Note: [1] Winter wheat only.
Source: USDA (2001).

et al. (1989).[5] The functions, which are often referred to as *M7*-functions,[6] are presented in figure 1. We assume that tubers (mainly potato) have about the same sensitivity to ozone as wheat (see O'Connor *et al.*, 2003, and references therein).[7] Moreover, we assume that peanuts have approximately the same sensitivity as cotton,[8] and the *M7*-function for cotton given in EC (1999) is applied to this crop. There is no dose-response function in the literature for vegetables as such. The sensitivity to ozone of different vegetables varies significantly. Although very uncertain, we calculate an average function for lettuces, tomatoes, and spinach, denoted 'Vegetables'. The functions in figure 1 are similar to those of Aunan *et al.* (2000), except for cotton, which they did not include.

Crop losses in China modeled as stepwise linear functions of annual emissions of NO_x

NO_x is the limiting factor in ozone formation in the scenarios. NO_x emissions are available from the economic model simulated without crop-loss feedback. By means of the CTM, we model the impact that the projected NO_x has on the level of surface ozone across China, and derive *M7*

[5] Studies of Pakistan rice cultivars (Maggs and Ashmore, 1998) have found substantially greater impacts of ozone on rice yield than what is found for other crops reported in EC (1999) and for rice reported by Adams *et al.* (1989). Moreover, Zheng *et al.* (1998) show that a rice cultivar commonly grown in Chongqing, China, is more sensitive to O_3 in terms of growth and visible injury than cultivars grown in Pakistan, Japan and the US.

[6] *M7* is a metric of surface ozone; seasonal average 7 hrs d^{-1} for the period March–October, given in ppbv.

[7] Newer studies in Europe indicate that the potato is slightly less sensitive than wheat (Pleijel *et al.*, 2004).

[8] Based on USEPA (1996) where dose-response functions using the ozone metric SUM06 is given.

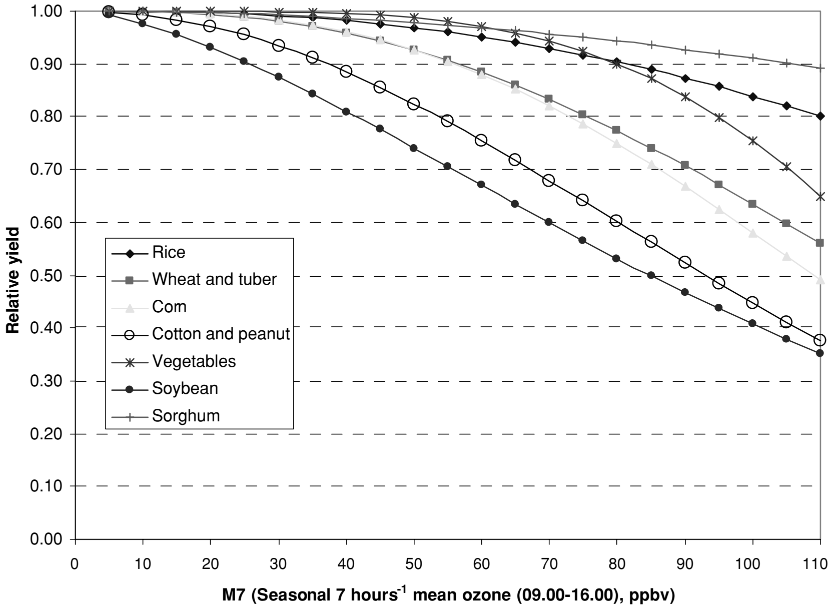

Figure 1. *Dose-response functions for relative yield as functions of M7*

scenarios for each Chinese province. Combining the scenarios with the crop production at a province level and the Weibull functions shown in figure 1, we are able to estimate the production of a given crop in a given ozone scenario, P_{scen}.

We then derive reduced-form functions for crop loss. The reduced-form functions are stepwise linear. They describe the implicit relationship between NO_x emissions and crop production, and take the form

$$P_{scen} = P_{reg}\left[1 + \sum_{i=1} b_i(a_i - a_{i-1})\right], a_i \in \{0, \ldots, E\}$$

where P_{scen} is crop production (given in 1000 tons) at the province level (aggregated in the case of Rest of China), P_{reg} is hypothetical production at zero emissions and E is the annual NO_x emission (as an ozone precursor; given in Mt(N)/y). The parameters a and b of the function for P_{scen} for each crop and region are given in O'Connor *et al.* (2003). Actual emissions E cut off the string of a_i in the summation.

The methodology described here does not 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 to the relationship between emissions and crop productivity. In addition, it is not self-evident that Chinese cultivars have similar sensitivities to ozone as Western cultivars. We regard the estimate for rice as particularly uncertain. Since rice is a major crop, this uncertainty may be important. Two factors indicate that rice yields may be more affected by ozone than what we have assumed: As noted above, studies of Pakistan rice cultivars have found substantially greater impacts of ozone on rice yield than what was reported by Adams *et al.* (1989) and at least one Chinese cultivar has been shown to have a high sensitivity to ozone (impacts on yield was not studied, however). Moreover, concerning wheat cultivars, it has been shown that newer cultivars tend to be more sensitive to ozone than older. We do not know whether this applies to rice too. If it does, it may be important as the share of modern rice cultivars in China is among the highest in Asia (Aunan *et al.*, 2000). A higher impact on rice would increase the estimated yield reduction associated with NO_x-emissions. Finally, perhaps the largest uncertainty derives from the absence of independent tropospheric ozone measurements for China, which forces us to rely on model-generated estimates. If model estimates are lower than the true level, yield reduction is again too low. If estimates are higher, yield reduction is too high.

5. Results of the simulations

5.1. Defining the counterfactual baseline

The analysis of the ancillary benefits and costs of climate policy depends crucially on a plausible counterfactual. The counterfactual describes how China's economy and energy system evolve over the coming decade in the absence of explicit climate policy measures such as a carbon tax or system of tradable carbon credits. The growth of China's economy to 2010 and beyond is uncertain. Equally important, the energy mix evolves even in the absence of climate policy. The energy mix depends, for instance, on relative prices of coal and oil, on considerations of energy security of supply, on how fast China brings on line the country's oil and gas fields, on growth in transport demand, and on construction of gas distribution networks. We assume that the country has the will and the investment resources necessary to import oil, and to bring on line new oil and gas fields with distribution networks in the baseline scenario. Thus, we foresee that oil and natural gas production increase in the baseline scenario, and that there is flexibility to increase production as well as imports of oil in the climate scenarios. A reliance on imports implies that security of supply is not an overriding concern. In actual fact, both production and imports of oil and gas have increased since 1997, and major additions to the market are expected in the period 2005–2010 (e.g., Asian Pacific Energy Research Center, 2004).

The baseline assumes that there is no climate policy in the simulation period, and provides a reference for the climate policy scenarios. The guiding reference for the baseline is China's recent history in terms of economic growth and emission growth. In the baseline, the real 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. In policy scenarios, real GDP growth is endogenous. Population and labour force growth are exogenous. The urban population in China has increased from 20 per cent in 1978 to 42 per cent in 2004 (NBS, 2005). Reflecting this long-term trend, the urban population is assumed to grow at a significantly faster rate than the overall population. Labour-augmenting technical change is

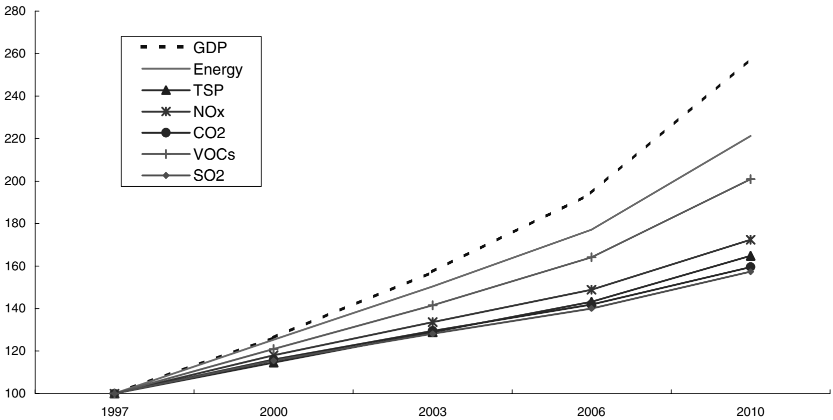

Figure 2. *Growth of GDP, energy consumption and emissions in baseline, all China*

endogenous in the baseline and exogenous in policy scenarios. Baseline labour-augmenting technical change in non-energy sectors is on average 5 per cent per annum over the simulation period, while in energy sectors it is 3 per cent. In policy scenarios, technical change is equal to the baseline. Beside labour-augmenting technical change, an autonomous energy efficiency improvement (AEEI) factor of 2.5 per cent per annum is assumed. The historical growth record of China shows a trend increase in energy efficiency (e.g., Fisher-Vanden *et al.*, 2004). Recently the data show fluctuations in energy efficiency (NBS, 2005). We assume that the trend to higher energy efficiency resumes. If in fact it does not resume, the environmental outcomes will be larger than we are reporting. Besides the AEEI, recall the reduction in emission factors, which in a sense is an emission factor productivity growth. The energy and emission productivity parameters are calibrated to merge China's historical environmental performance in resource inputs with its GDP growth.

5.2. Baseline simulation results

Figure 2 shows that China's ratio of energy to GDP will fall steadily to 2010 by about 1.2 per cent per annum. Emission growth of all pollutants is slower than growth in energy consumption. VOCs grow the fastest, followed by NO_x. The difference between growth trends arises from composition effects. Both VOCs and NO_x are closely linked to the rapid rise in the motor vehicle stock and the shift towards natural gas in power generation and heating. It should be recalled that the growth of the motor vehicle stock is from a rather small base. Gas's rapid growth is also from a small base, with gas contributing a negligible share of power generating capacity in 1999.

CO_2-emissions are projected to grow the slowest among pollutants, and significantly slower than GDP and energy consumption. The main reason is that process emissions grow more slowly than both GDP and energy since the economy restructures along the growth path away from heavy industry, into light industry and service. The fuel switch in the baseline from coal towards gas also contributes to lower growth in CO_2-emissions.

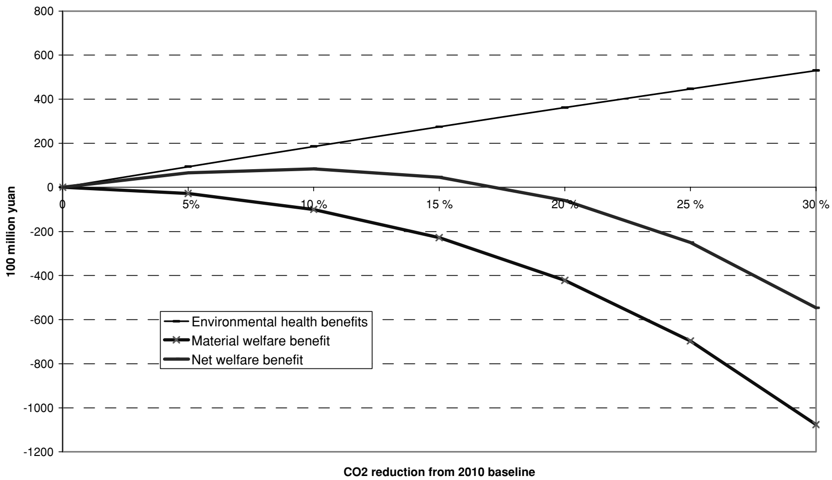

Figure 3. *Net benefit and its determinants, 2010*
Note: Negative material welfare benefit equal to material welfare cost in the text.

5.3. Costs and benefits to China of a climate policy

Exogenous inputs

We study the consequences of imposing targets on Chinese CO_2 emissions in 2010. Since we are interested in how costs and benefits depend on the stringency of targets, we analyse several targets at increasing stringency. The targets start at a five per cent reduction and go to a maximum of a 30 per cent reduction relative to the 2010 baseline.[9] 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 the designated target. Holding regional real government and corporate savings constant, additional carbon tax revenue is distributed back to households. The tax-redistribution scheme that we model can also be described as a particular version of transferable quotas.

Macroeconomic costs and benefits

Our simulations indicate that the cost of a CO_2-commitment is substantial, and, moreover, that it increases with the level of stringency. But we also find that avoided crop loss has a marked impact on the cost curve compared with what it otherwise would have been, and that health benefits are substantial and increase with the level of abatement. Figure 3 illustrates that net benefits (health benefit minus costs including avoided crop loss) are above zero for a fairly long interval of CO_2-reduction and dip below costs around 17.5 per cent CO_2 abatement. In other words, China may reduce its CO_2-emissions

[9] The carbon tax associated with a 5 per cent reduction is estimated at 56 RMB/tC. A 20% reduction requires 290 RMB/tC and 30% requires 540 RMB/tC. For comparison, current prices in the European Trading Scheme for CO_2 amount to 10–30 €/tCO_2, equivalent to 300–1000 RMB/tC.

by 17.5 per cent without a net loss to economic welfare.[10] This estimated no-regrets level of CO_2 abatement equals approximately 600 million tons, which, for instance, corresponds to about 2.5 per cent of global total CO_2 emissions from fossil fuels in 2000 and nearly three quarters of Africa's emissions the same year.

The cost of CO_2 abatement is positive despite the avoided crop loss. The cost curve is non-linear since the marginal cost of CO_2-abatement increases with stringency. The health benefit, however, is a linear function of the abatement level because we assume (i) absence of a lower threshold for health effects of air pollution, and (ii) a linear form of the dose-response relationships in the relevant range of air pollution abatement. The linear form of health benefits is a first-order approximation of the unknown underlying relationship.

The estimated no-regrets point of CO_2 abatement, i.e. the point where the net benefit dips below zero, is of course highly sensitive to the shapes of the cost and benefit curves. If the cost curve is calculated without including the impact of avoided crop loss, its slope will be steeper and we obtain a no-regrets level of CO_2 abatement around 5 per cent. The bend of the cost curve is mainly determined by the degree of substitutability in production and consumption. For example, if there are many substitutes to carbon intensive products and modes of production, increasing the level of CO_2 abatement will not increase the cost much. Hence, high substitutability contributes to a gentle bend in the curve.

In O'Connor *et al.* (2003) we describe in more detail the physical and technical side of the substitution effect. With elasticities of substitution for new capital between 0.2–1.0, except within energy where the elasticity is 2.0, we find that about half of the reduction of CO_2 is due to 'fuel switch' out of coal and into oil and gas within industries. A further one-third of substitution occurs between energy and other inputs at the industry level. Finally, one-sixth of substitution is due to industry shifts. In other words, five-sixths of the reduction of CO_2 is accommodated within industries. The emphasis on within-industry decline in energy intensity is confirmed by econometric studies of historical data using similar definitions of industries (e.g., Fisher-Vanden *et al.*, 2004).

We emphasized above that the crop loss aspect changes the slope of the cost curve from a 5 per cent no regret level to 17.5 per cent. In fact, crop losses also influence the bend of the curve. From the dose-response functions the marginal benefit on agricultural productivity is stronger the more air quality improves. This increasing marginal benefit cushions the cost of restraining carbon use and contributes to a gentler bend in the cost curve. We now describe the impact on agriculture in more detail.

Benefits to agricultural productivity

A carbon tax will affect output of various crops and, in turn, relative prices, incomes of rural households and consumer welfare. As table 4 shows, for a 10 per cent carbon reduction, NO_x declines by 6 per cent and the

[10] Since we simulate at 5 per cent intervals, we reach this estimate by interpolation.

Table 4. *Impacts on productivity in crop sectors, 2010 (per cent change relative to baseline)*

	Reduction in CO_2 emissions	
	10%	*20%*
NO_x emissions	−6.1	−12.3
Productivity		
Rice	0.2	0.3
Wheat	0.2	0.4
Corn	0.15	0.3
Cotton	0.5	1.0
Other crops	0.2	0.4

Table 5. *Distribution of welfare gain/loss by urban/rural or region, 2010*

	Reduction in CO_2 emissions					
	5%	*10%*	*15%*	*20%*	*25%*	*30%*
Net benefit rural	0.02	0.01	−0.04	−0.14	−0.30	−0.53
Net benefit urban	0.08	0.11	0.10	0.02	−0.13	−0.37
Net benefit Guangdong	0.03	0.02	−0.02	−0.12	−0.27	−0.51
Net benefit Rest of China	0.03	0.05	0.03	−0.02	−0.11	−0.25

Note: Equation (2) is used. Urban/rural welfare change as percentage of household disposable income. Guangdong/Rest of China welfare change as percentage of gross product in region.

increment in crop productivity ranges between 0.15 and 0.5 per cent. Cotton experiences the highest increase in yields. The apparent linearity of effects in the 10–20 per cent range originates in the stepwise linear functions used. Rice, the most important cultivar, is an exception.

Although the direct productivity gains in crop sectors seem modest, they are realized on a large scale, since 60 per cent of the population in China live in rural areas. Altogether they represent a significant benefit compared with other costs and benefits of a Chinese climate commitment. Using equation (1) we find that the agricultural benefit has about the same magnitude as the health benefit. This is quite interesting since health benefits, in particular benefits on mortality, have long been thought to be far larger than other environmental benefits. Our results cast doubt on whether this is so in China once other benefits are carefully incorporated.

Distribution of welfare gain and loss

Having studied the macroeconomic costs and benefits of a carbon tax, we now discuss the distribution of these items. The first two rows of table 5 show the distribution of welfare benefits and costs across urban and rural households. The table indicates that up to a climate commitment between 5 and 10 per cent urban and rural households both gain. Urban households

Table 6. *Distribution of welfare gain by urban/rural and region, 2010*

	Reduction in CO_2 emissions					
	5%	*10%*	*15%*	*20%*	*25%*	*30%*
Guangdong						
Net benefits 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						
Net benefits 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

Note: Equation (2) is used. Welfare change as percentage of household disposable income.

gain the most. A commitment between 12.5 per cent and 17.5 per cent implies that urban households gain, while rural households lose (see also figure 3).

Rural households benefit disproportionally from higher agricultural productivity. Higher productivity translates directly into higher rural incomes, although food prices fall to some degree. Rural households also benefit from lower food prices because of the large weight of food items in household expenditures. The productivity gains in agriculture also raise welfare (or mute its decline) in urban areas via the impact on food prices, but the impact is not as strong.

However, in the end, urban households are better off than rural households. The reason is that urban households capture all the health benefits of local air quality improvements, while the costs are distributed across both urban and rural households. So the gain to rural households is to some extent shared with urban households (through lower prices), but the gain to urban households is not shared.

It should be added that in terms of income distribution as conventionally measured (excluding health benefits), it is rural households that win. The reason is that their gain is pecuniary, but the urban gain is primarily non-pecuniary. Rural households are poorer than urban, so the conventional income distribution improves. Since it improves, a casual observer will conclude that a climate commitment particularly benefits rural households. As we have seen, the conclusion is misleading. When pollution is included, the distribution of welfare deteriorates. As an aside, the distribution of welfare is of course much more equitable than the distribution of income, since the distribution of welfare includes a cost element (health cost of urban pollution) that only falls on the urban population.

The last two rows of table 6 show the distribution of welfare gains and losses across provinces. Guangdong Province, which is ten times richer than the poorest provinces, loses compared with the rest of China. Again it is the relief in environmental pollution that drives the result. The regional income distribution actually becomes more skewed in the direction of Guangdong, but the rest of China receives most of the environmental gain.

The rest of China receives most of the environmental gain simply because production in the rest of China is dirtier. It relies more on heavy industry and it is dirtier per unit of output in each industry. Differences in exposure also matter since Guangdong is a coastal province. At the same time, the dirtiness in the rest of China indicates that its production is more energy intensive and explains why it is hit more in terms of conventional income.

Behind the results for urban-rural households and for Guangdong and Rest of China lies an urban-rural-regional matrix of results. Table 6 presents this matrix. It indicates that urban households in Guangdong fare the worst following a climate commitment, while urban households in the rest of China fare the best.

A climate commitment may in other words imply a transfer of welfare from urban households in Guangdong to urban households in the rest of China. Interestingly, for rural households the transfer runs the other way, from the rest of China to Guangdong. The reason for these transfers is the unequal burden of urban pollution, agricultural productivity and reliance on dirty production. However, there are only minor differences in abatement rates between regions. For instance, the 30 per cent reduction consists of 29.6 per cent reduction in Guangdong and 30.05 per cent reduction in the much larger rest of China.

5.4. Sensitivity analysis and caveats

Substitution parameters relative to willingness to pay for health benefits

Above we have emphasized how our results depend on the magnitude of the health benefit curve relative to the cost curve, and they also depend on the shape of the cost curve. The results assume 'best guess' values of willingness to pay (WTP) for morbidity and mortality risk reductions in the Chinese population. Based on the literature, central values of the relevant substitution elasticities are used, in line with those employed in the OECD's GREEN model (Lee *et al.*, 1994). It goes without saying that if WTP were significantly lower than the values assumed here, the health benefits of climate policy would be reduced proportionately. The health benefit curve would rotate downwards in figure 3. 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 and the cost curve would have a sharper bend. Needless to say, it is also possible that the parameter values used here underestimate rather than overestimate actual Chinese elasticities, in which case the reverse is true: the cost curve bends more gently than shown here. A higher value of VSL, e.g. because it grows over time, would mean that the health benefit curve rotates upwards. More climate abatement would be warranted.

There is uncertainty in any simulation exercise, but where the parameter values are based on 'consensus' estimates from a range of studies mostly done outside 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, which are reported in full in

O'Connor *et al.* (2003), prove to be rather sensitive to the choice of elasticity values. Their halving results in welfare losses about three times those in the basic policy scenario. The no-regret abatement level is reached below 5 per cent. A halving of substitution elasticities implies an economy that in an international context is rather inflexible, however. Even with this inflexibility, health benefits amount to between 50 and 60 per cent of the material 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, health benefits amount to 30 per cent of those costs. Thus, even a climate policy maker who subscribes to the idea of an inflexible economy by international standards could expect to recover a substantial fraction of material welfare costs associated with a carbon tax through lower health costs and crop losses from pollution.

Interregional mobility and tax revenue distribution

Other important parameters and exogenous assumptions that may 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. In the 'basic' policy scenario, complete factor immobility across regions is assumed. Sensitivity tests, however, show that raising the elasticities on interregional labour and capital mobility from zero to unity and then to two hardly affects the basic results.

The method of carbon tax revenue distribution may affect the size of welfare gains and losses in several ways, by augmenting or reducing existing tax distortions, but also by altering the size of the pool of savings and 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 is used for reducing the corporate income tax, then the welfare cost of the tax is considerably reduced and the no regret 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 revenues are split roughly 50:50 between the corporate and household sectors in order to maintain corporate savings). Some of the apparent gain from emphasizing lower corporate taxes may be due to the fact that we emphasize long-run results. Increased savings now leave more for consumption in the long run, but less in the short run.

Sensitivity to dose-response function for rice yield

As mentioned in section 4 the dose-response function for impacts of surface ozone on rice yields is particularly uncertain. To test the sensitivity to steeper yield loss functions for rice crop loss estimates, we adjusted two alternative Weibull functions so they pass through the estimated yield loss given for Pakistan rice cultivars in Maggs and Ashmore (1998) (the study gives a point estimate of 37 per cent yield loss at 43 ppb but no function). Applying an adjusted function that is falling relatively steeply between 40 and 60 ppb, the estimated avoided yield loss in a scenario where NO_x emissions are reduced by 12.3 per cent (equal to the scenario where CO_2 emission reduction are reduced by 20 per cent, see table 4) becomes 4.6 per cent higher. Applying an adjusted function with a moderate slope between 40 and 60 ppb, the estimated avoided yield loss for the same scenario becomes 0.9 per cent higher. Finally we applied the function for soybean, which has the steepest function among the crops assessed here (see figure 1). The estimated avoided rice yield loss becomes 1.1 per cent higher. As rice constitutes less that half of the grain production, and other crops, such as vegetables, are of higher economic value than grains per ton, the sensitivity to a steeper rice crop loss function is small.

Sensitivity to environmental policy

It is possible that China in the future will pursue a more aggressive local environmental policy than at present. The analysis above implicitly demonstrates that policies to reduce TSP and SO_2 may be sensible in their own right. A CO_2-tax is a blunt instrument for reducing local pollutants and it is of interest to analyse what scope remains for a CO_2-tax given an aggressive local environmental policy. Here we show the impact that policies to reduce TSP might have on no-regret CO_2-abatement.

The starting point is two policy designs to reduce emissions of TSP by 10 per cent relative to the 2010 baseline. In policy design number one, we exogenously reduce emission factors for TSP across the board. This policy design leaves fossil energy consumption and CO_2-emissions unchanged. In policy design number two, we leave emission factors as they are and reduce TSP emissions via reduced consumption of fossil energy. The instrument for reducing TSP in policy design number two is a tax on TSP-emissions. Policy design number two lowers CO_2-emissions as well as TSP-emissions. An optimal policy to reduce TSP emissions is likely to target both emission factors and energy consumption, similarly to an average of the two policy designs we study.

The result of reducing emission factors such that TSP-emissions fall 10 per cent is illustrated in figure 4. In our model, the impact on no-regret CO_2-emissions is negligible. Nothing of interest happens to material welfare benefits since the only change in the model is a reduction in emission factors. Perhaps a little surprisingly, hardly any change is seen in environmental health benefits either. The reason is that as TSP emissions are reduced, damages from SO_2-emissions take over. Therefore, health benefits are not reduced by roughly 10 per cent as could be expected, but by 4 per cent. Four per cent lower health benefits do not make a significant dent in no-regret CO_2 options.

Turning to option number two for TSP reduction the no-regret potential for CO_2-emission reductions disappears, see figure 5. The reason is that the TSP-tax overlaps greatly with what a CO_2-tax would have yielded. There is no reduction in emission factors, and therefore the only way to lower TSP-emissions is by reducing fossil fuel use. The reduction in fossil fuel use brings along a reduction in combustion-related CO_2-emissions.

Since a tax in TSP is similar to a tax on CO_2, further CO_2 reductions have a higher material welfare cost than the cost indicated in the base-case analysis. Another way to understand this is that the CO_2-instrument takes over

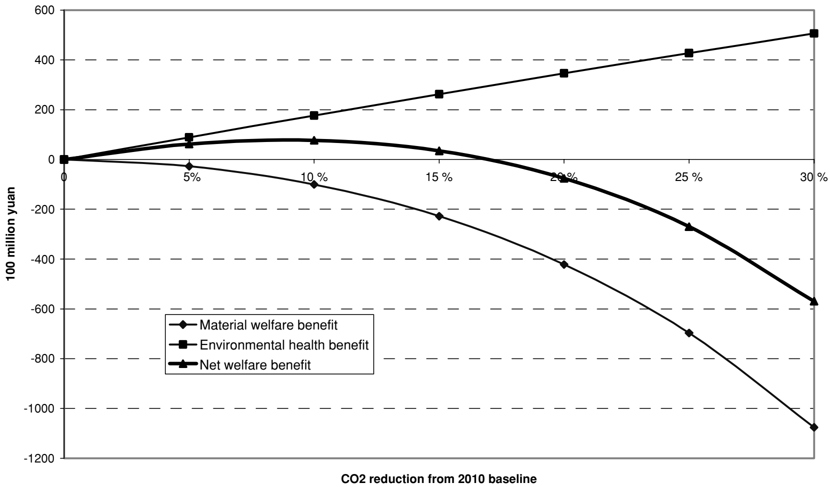

Figure 4. *Sensitivity: emission factors for TSP emissions reduced approx 10 per cent*
Note: Negative material welfare benefit equal to material welfare cost in text.

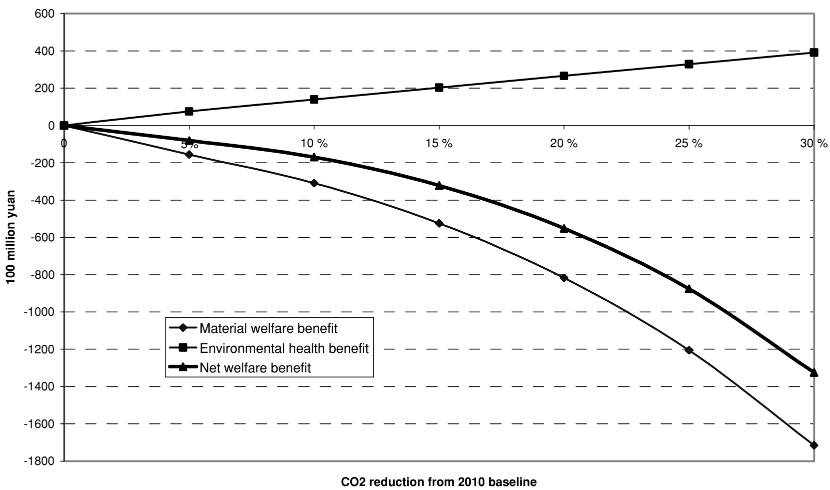

Figure 5. *Sensitivity: fossil energy use for TSP emissions reduced approx 10 per cent*
Note: Negative material welfare benefit equal to material welfare cost in text.

where the TSP-instrument stops. The TSP-tax is in this case an instrument for reducing fossil fuel consumption. The CO_2-tax reduces it further.

In a mixed TSP-policy that combines lower emission factors and fossil fuel reductions, the scope for no-regret CO_2 options will lie between zero and the baseline impact of approximately 17.5 per cent.

Relative importance of sensitivities

In sum, simulation results prove sensitive to three choices: (i) substitution elasticity parameters, (ii) revenue recycling assumptions, and (iii) initial

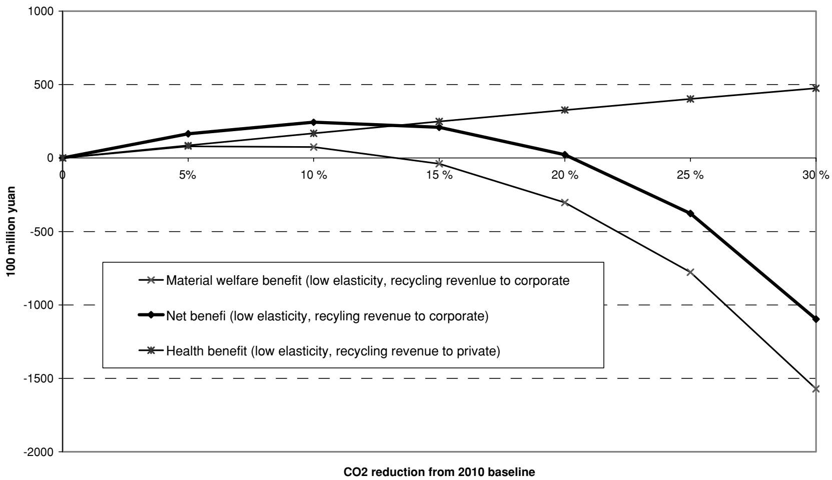

Figure 6. *Sensitivity: low elasticities, corporate tax reduction*
Note: Negative material welfare benefit equal to material welfare cost in text.

environmental policy. 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. The third is a policy choice that should be informed by cost–benefit considerations of local environmental policy. As a last sensitivity exercise, we can examine a scenario in which low substitution elasticities are combined with full revenue recycling through offsetting reduction in corporate tax.

The recycling and substitution assumption has a marked effect on material welfare costs, hence, on the net benefits curve and the no regret range of CO_2 reductions; see figure 6. Relative to the basic policy scenario, the no regret reduction in this scenario is almost 5 percentage points *higher* despite the counteracting impact of lower substitutability. The recycling and substitutability assumptions also have 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 regret climate policy, even if substitution elasticities should prove in fact to be on the low end of the plausible range.

6. Conclusions and comparison with earlier research

The question we posed in the introduction to this study was: Supposing we were to introduce an energy tax to encourage a less carbon-intensive development path, how much of the economic cost would be offset by benefits that accrue from improved human health and agricultural productivity? We conclude that the answer is: A substantial share of costs would be offset. In fact, for 17.5 per cent of CO_2-emissions *all* costs would be offset.

The main methodological innovation of the paper, the model of feedbacks from emissions to agricultural productivity through the ozone chain, is important for the conclusion. The agricultural productivity benefit is almost as large as the health benefit. Had we not included it only 5 per cent of CO_2-emissions could be offset. The marginal impact on the conclusion is therefore very high.

Besides the aggregate economic effect, including the agricultural feedback is important for distributional impacts. Before considering the agricultural feedback, poor rural households experience welfare losses even at low levels of carbon abatement. Once agricultural effects are considered, rural households enjoy welfare gains up to a 10 per cent abatement rate. Thus, while a purely health-based measure of ancillary benefits would tend to show a climate commitment to be urban-biased, a broader definition of benefits alters the picture considerably.

A criticism sometimes voiced to the question we have addressed in this paper is that environmental benefits are better achieved by environmental policy than climate policy. To advocate climate policy based on ancillary environmental benefits is not reasonable, the criticism goes. We have addressed this criticism in a sensitivity analysis above and it all depends on the extent of overlap between, for example, a policy to reduce TSP and a CO_2-control policy. If the overlap in impact is small because the target on TSP by and large is achieved through lower emission factors, then there remains significant scope for a no regret CO_2 policy. On the other hand, if the target on TSP is achieved through lower consumption of fossil fuel, the scope for no regret CO_2-reductions is small. On the other hand, a TSP-policy that lowers fossil fuel consumption will reduce CO_2 as a co-benefit. It does not matter to the environment or to an overall welfare evaluation which is the main beneficiary and which is the co-beneficiary.

As a matter of principle, environmental policy in developing countries should not, in our view, be assumed to be optimal when in fact it is not. According to standard cost–benefit theory (e.g., Drèze and Stern, 1987), we should evaluate a policy instrument (in this case climate policy) based on other policy instruments being what they are, not what we would have liked them to be. In China as in other developing countries, it is likely that environmental policy becomes more stringent as the economy develops (e.g., Copeland and Taylor, 2004). As discussed above we assume an increasingly stringent environmental policy in terms of emission factors, since that is a realistic expectation. Our model of baseline fossil fuel consumption allows autonomous energy efficiency improvements.

Our analysis shows that the ancillary benefits, economic costs, and 17.5 per cent no regret CO_2-reduction is 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 influence the results. Costs are higher when elasticities are halved, due to much lower flexibility in the 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 regret abatement. In short, there would appear to be a double dividend associated with the use of the revenue to lower corporate tax, 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.

It is of interest to compare our analysis with those of others. Zhang (2000) analyses the economic effects of China reducing its carbon emissions in 2010 by 20 per cent and 30 per cent below baseline, respectively. In the 20 per cent reduction case, GNP declines 1.5 per cent and welfare declines 1.1 per cent. Our analysis of 20 per cent reduction suggests 0.2 per cent decline in welfare. Removing the crop effect in order to compare on level with Zhang, our analysis suggests 0.4 per cent. 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 ten sectors compared to the 61 sectors here. With few sectors, substitution possibilities are inherently limited in response to a carbon tax, raising the costs of adjustment.

Garbaccio *et al.* (1999) examine costs of reducing carbon emissions over 30 years by 5, 10, and 15 per cent below baseline. They offset the carbon tax by lowering the income tax on enterprises, and find that in year 30 progressively larger emission reductions actually yield progressively larger increments to GDP, with a 15 per cent emission reduction adding almost 1 per cent to GDP in the final year: The mechanism is that a lower corporate income tax increases saving and investment. 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. These results are similar to the ones derived in our study in the sensitivity analysis where all carbon tax revenue is redistributed to corporations. Including the agricultural feedback seems less crucial in a setting where tax revenue is used to increase savings and growth.

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