China & World Economy

China & World Economy / 72–92, Vol. 27, No. 1, 2019

Taxation and the Environment–Health–Poverty Trap: A Policy Experiment Perspective

*Sumei Chen, Ling-Yun He**

Abstract

Under pressures related to economic growth and environmental protection, China is facing an increasingly severe "environment–health–poverty" trap risk. Fuel taxation is generally considered an effective policy to counter such a risk. Since 2009 China has raised the fuel tax rate many times to enhance tax reform. However, the effects of this policy remain unknown. Therefore, it is vitally important to estimate the impacts of China's current fuel taxation policy on environment, public health and the national economy. As the first attempt in existing literature on China, this paper builds a general equilibrium framework with the feedback effect of public health on economy. We find that that the fuel tax policy benefits the adjustment of the economic structure and improves human health; however, it is detrimental to economic growth, public welfare and price stability. In this sense, it plays a limited role in reducing the trap risk and might not be sustainable in the long term.

Key words: economic growth, environment–health–poverty trap, fuel tax, public health

JEL codes: H23, I18, P28

I. Introduction

Since 1978, China's tremendous energy-intensive economic growth has been accompanied by immense pressure on the natural environment and human health. It may even have hindered economic growth because of the loss of workdays and excess medical expenses resulting from the health issues caused by pollution.[1] In this light, it

---

*Sumei Chen, Assistant Research Fellow, Institute of Industrial Economics, Chinese Academy of Social Sciences, China. Email: csm_cau@163.com; Ling-Yun He (corresponding author), Professor, College of Economics, Jinan University, China. Email: lyhe@amss.ac.cn. The authors acknowledge financial support from the National Natural Science Foundation of China (Nos. 71874070, 71803191 and 71573258).

[1] Because of serious environmental pollution, China has been named the air pollution capital of the world (Watts, 2005). In terms of notorious haze, fine particulate matter with an aerodynamic diameter smaller than 2.5 μm ($PM_{2.5}$) is a major component and causes severe health concerns. In 2013, the annual mean $PM_{2.5}$ level in China reached 72 μg/m^3, which was 7.2 times higher than the reference standard recommended by the World Health Organization (WHO, 2006). More seriously, in 2007, China experienced a staggering GDP loss of approximately RMB361.47bn and a welfare loss of approximately RMB227.65bn as a result of air pollution (Chen and He, 2014).

is quite possible that such a vicious cycle may lead China to fall into the "environment–health–poverty" trap (Qi and Lu, 2015a, b; Chen and He, 2017).[2] This issue has attracted a great deal of attention from the general public and the government.[3] To counter this "environment–health–poverty" trap risk, China faces the difficult but vitally important task of balancing economic growth, human health and environmental protection.

In recent years, fuel consumption has been a major anthropogenic emission source with a significantly negative externality.[4] In developed countries, fuel taxation is recognized as an effective tool to abate pollution (Hibiki and Arimura, 2005; Sterner, 2007). It is particularly relevant for China with such a severe environment–health–poverty risk. As Table 1 shows, China's fuel tax rates have been raised many times, but their impact remains unknown. In particular, the rise of fuel tax rates may threaten the economy and residential welfare. It is this very concern that has led the government to proceed cautiously when making changes to the fuel tax policy. Naturally, the following questions, key to promoting fuel tax reform, need to be addressed: Does China's current fuel tax policy make a real difference? And how do key variables such as air quality, human health, GDP and social welfare, interact with the current fuel tax policy? Thus, this paper aims to analyze the impacts of China's current fuel taxation on the environment, economy and human health.

Because fuel tax is closely connected to transportation, as well as other industries that use fuel oil as an intermediate input, any changes might have wide effects on resource allocation, industry output and overall economic performance. To deal with such a complex but important issue, a computable general equilibrium (CGE) model, which can capture key features of related sectors and markets, as well as economic activities, would be appropriate. To date, a body of literature analyzing China's fuel tax policy has employed the CGE approach. Jiang (2006) found that imposing a fuel consumption tax in Beijing would improve air quality but slow economic growth. Pang et al. (2008) showed that fuel tax reform would have a significant effect on energy conservation, despite welfare loss. Xiao and Lai (2009) further found that, over time, fuel tax reform would result in more losses to capital-intensive than to labor-

---

[2] With regard to the environment–health–poverty trap, all of the underlying mechanisms in these studies refer to the relationship among environmental pollution, public health and the economy. As there is a potential risk for China to fall into this trap, further policy intervention is required.

[3] As stressed by China's President Xi Jinping, "we cannot achieve our goal of building a moderately prosperous society in all respects without protecting public health; we must place health at the heart of all policy making." Source: http://news.xinhuanet.com/politics/2016-08/21/c_129244493.htm (cited 5 June 2018).

[4] Taking Beijing as an example, the motor vehicle is the largest $PM_{2.5}$ source. See: http://www.xinhuanet.com/politics/2018-05/15/c_1122832062.htm (cited 5 June 2018).

intensive industries. However, Ye (2009) and Cao (2007) argued that if fuel taxation is appropriately designed, it is likely to benefit social welfare and GDP. Thus, there is no general agreement on the effects of fuel taxation at present.

Table 1. China's Fuel Tax Reform Process

Year	Description
1994	Official proposal to impose fuel tax. A pilot program in Hainan Province was initiated to collect a fuel charge of 1500 yuan/ton for gasoline vehicles and a four-in-one charge (including highway maintenance tax, highway toll, bridge toll and transport management fee) of 300 yuan for diesel vehicles.
1995	Fuel tax reform officially initiated.
1997	*The Highway Law of the People's Republic of China* proposed revoking the "highway maintenance fee" and establishing a "fuel surcharge" to come into effect 1 January 1998. However, this plan was vetoed.
1999	An amendment to *the Highway Law of the People's Republic of China* was approved, and "fuel surcharge" was replaced by "fuel tax."
2008	Consumption tax of 0.2 yuan for a liter of lead-free gasoline or 0.1 yuan for a liter of diesel was established.
2009	China officially introduced a fuel tax policy and revoked six charges, including highway and, waterway maintenance and transport management fees, and highway and waterway passenger and freight transport surcharges. Secondary highway toll fees were to be phased out. For tax imposed in production and import processes, the unit gasoline and diesel consumption tax rates were raised from 0.2 to 1 yuan per liter and from 0.1 to 0.8 yuan per liter, respectively. Unit tax rates for other types of refined oil were also raised.
2014	On 28 November the gasoline and diesel consumption taxes were raised from 1 to 1.12 yuan per liter and from 0.8 to 0.94 yuan per liter, respectively; on 12 December the gasoline and diesel consumption taxes were again raised, from 1.12 to 1.4 yuan per liter and from 0.94 to 1.1 yuan per liter, respectively.
2015	On 12 January the gasoline consumption tax was raised to 1.52 per liter and fuel consumption tax rates for diesel, fuel gas and jet fuel were raised to 1.2 yuan per liter.

Source: Authors' compilation based on policy documents.

Because pollution is not only a byproduct but also an input of production (Shen, 2006), environmental policy analysis of the feedback effect of pollution on economic growth is crucial.[5] In recent years, researchers have begun to focus on health feedback in China and other countries based on a CGE approach (Selin et al., 2009; Nam et al., 2010; Matus et al., 2012; Chen and He, 2014; Xie et al., 2016). However, these previous studies mainly concentrated on historical air quality level, future targets or new energy technology, rather than a discussion of fuel taxation. Against this background, this study proposes an integrated assessment framework based on a CGE model to describe the vicious cycle of environmental degradation, health crisis and poverty, and apply it to China's current fuel tax policy for analysis.

The rest of the paper is organized as follows. Section II introduces the structure of

---

[5] For instance, Chen and He (2017) suggested that, theoretically, whether pollution-related health damages are taken into account is crucial for the optimal allocation of energy tax revenue.

an integrated assessment framework. Section III mainly describes the policy experiment. Simulation results and a discussion are provided in Section IV, followed by some concluding remarks and policy implications in Section V.

II. Structure of an Integrated Assessment Framework

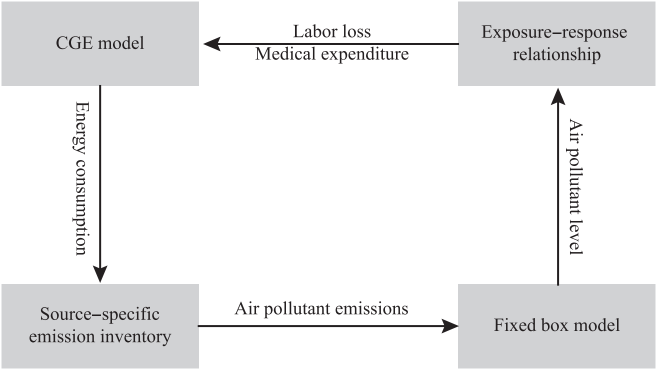
We developed an integrated assessment framework based on a CGE model to evaluate the impact of China's current fuel taxation policy. We incorporated air pollution-generated health costs, which affect the economy and social wellbeing from a general equilibrium perspective, into feedback. As $PM_{2.5}$ is known to be a much stronger risk factor to human health than particulate matter of less than 10 μm (PM_{10}) (WHO, 2013), this paper uses $PM_{2.5}$ and coarse particles ranging from 2.5 to 10 μm ($PM_{10–2.5}$)[6] as indicators of air pollution. In order to closely fit the current trap risk in China, similar to the approach taken by Chen and He (2014), we first reproduce the observed economic performance that has already been distorted by the health damages associated with the actual level of air pollution, and then estimate economic performance using health feedback under the fuel taxation scenario. It is worth noting that this feedback incorporates the differences between health damages from the fuel taxation scenario per se and our replication of actual health damages. Finally, by comparing economic performance of this fuel taxation scenario with the benchmark, we can capture the effects of China's fuel taxation on the environment–health–poverty trap risk.

More specifically, as illustrated in Figure 1, a static CGE model of the Chinese economy is built to capture the effect of fuel taxation on energy consumption. We then employ the Chinese source-specific emission inventory, which is directly linked to the CGE model, to measure the change in pollutant emissions. Given the emission levels of air pollutants, we use a simplified air concentration model, the fixed box model, to calculate national pollution concentrations and then estimate the number of cases in every pollution-related health outcome, as well as its corresponding cost, based on the existing epidemiological literature. Finally, the health damages (labor loss and medical expenditure) are entered into the CGE model as shocks to labor supply and the health service demand, thus capturing economic performance with health feedback. The details of the assessment framework are discussed in the following subsections.

---

[6] The paper adopts $PM_{10–2.5}$ rather than PM_{10} to avoid double counting, as $PM_{2.5}$ is smaller than 2.5 μm and PM_{10} smaller than 10 μm in terms of diameter.

Figure 1. A General Equilibrium Assessment Framework with Health-feedback Effect



Source: Author's own construction.
Note: CGE, computable general equilibrium.

1. A Computable General Equilibrium Model

In this paper, a 13-sector CGE model is used as the main module. Disaggregated sectors include: agriculture, coal, crude oil and gas, refined oil, electricity, service, energy-intensive industries, health service,[7] household transportation (private transportation),[8] urban public transportation, road transportation, other industrial transportation and other industries. We set the base year as 2012 as a result of data availability. The core database is the Social Accounting Matrix, mainly based on China's 2012 input–output table (NBS, 2012) and miscellaneous yearbooks and literature (MOF, 2013; NBS, 2013). Our model is composed of six basic submodules: production, income, expenditure, investment, international trade and fuel tax.[9] Because fuel tax is levied during the production process (including import process) in China, the direct taxpayers are the fuel oil production sector and importers (Lai et al., 2008). Thus, the fuel tax revenue can be calculated by:

$$RC = (QX_{oil} \times PX_{oil} + QM_{oil} \times PM_{oil}) \times TR_{oil} \times tc \times 10000, \tag{1}$$

---

[7] Including the health service sector allows us to capture the effects of medical expenditure related to air pollution.

[8] In this paper, the household transportation sector provides own-supplied transport service, using production from other industries (purchase of vehicle), service (maintenance, insurance etc.), and refined oil sectors as intermediate inputs. The details can be found in Chen and He (2014).

[9] For related lists of equations, parameters and variables of submodules, see Chen and He (2014). In this paper, we apply a nonlinear CGE model with a consideration of the substitution of energy products. Specifically, the energy aggregate that includes electricity, coal, crude oil and gas, and refined oil is represented by a Cobb–Douglas (C–D) function. As a result, associations between the extent of fuel taxation and the impacts on environment, health and economy are nonlinear in our simulation result.

where RC is fuel tax revenue (RMB100m); QX_{oil} and QM_{oil} denote the domestic output and import of refined oil, respectively; PX_{oil} and PM_{oil} the before-tax domestic and import prices of refined oil, respectively; TR_{oil} the refined oil conversion factor from a value-based perspective to a physical one (10,000 tons/RMB100m); and tc is the rate of specific duty on each ton of refined oil (yuan/ton). For modeling convenience we follow the approach taken by Xiao (2009) to obtain the ad valorem fuel tax rate, dth, as follows:

$$dth = \frac{RC}{QX_{oil} \times PX_{oil} + QM_{oil} \times PM_{oil}}. \tag{2}$$

Then, the after-tax domestic price, PXC_{oil}, and import price of refined oil, PMC_{oil}, can be calculated with the following Equations (3) and (4), respectively.

$$PXC_{oil} = PX_{oil} \times (1 + dth), \tag{3}$$

$$PMC_{oil} = PM_{oil} \times (1 + dth). \tag{4}$$

2. Anthropogenic Emission Module

As mentioned above, we employ a source-specific inventory of anthropogenic emissions as an interface from all types of fossil energy use to air quality during production activities and daily life. These anthropogenic emissions are closely related to source characteristics, such as fuel type and economic sector. Thus, the resulting air pollutant emissions ($PM_{2.5}$ and $PM_{10-2.5}$) are determined using Equation (5):

$$E_m = \sum_i \sum_e A_{e,i,m} F_{e,i,m}, \tag{5}$$

where E denotes anthropogenic emissions (g); A the use of fossil energy (kg); F the corresponding emission factor (g/kg); and m, e and i denote air pollutant ($PM_{2.5}$ and $PM_{10-2.5}$), energy type (coal, refined oil and natural gas) and economic sector (household, electric power, industry, commerce and institution, agriculture, and transportation), respectively. Here, the values of parameter $A_{e,i,m}$ are decided from the simulation results of our CGE model; $F_{e,i,m}$ values are specified according to related studies, with some modifications (Klimont et al., 2002; Wang et al., 2005).[10]

3. Air Quality Module

Air quality in one location is not only related to anthropogenic emissions, but also affected by many local natural factors, such as meteorology, topography and emission sources. However, it is difficult to specify these local factors at the national level,

---

[10] Because of data availability, this paper regards the emission factor of PM_{10} as $PM_{10-2.5}$.

especially for a vast country like China. For the sake of simplicity, we employ the fixed box model whereby the whole of China is assumed to be a box with uniform pollution dispersion. Under the assumption that the natural parameters are held constant for all time, referring to the approach taken by Chen and He (2014), we can use the following equation to estimate the air quality in scenario simulations:[11]

$$\frac{E_{1,m}}{E_{2,m}}=\frac{C_{2,m}-b_m}{C_{1,m}-b_m}, \tag{6}$$

where $E_{1,m}$ and $E_{2,m}$ refer to the baseline annual emission and simulated counterfactual emission of pollutant m (g), respectively, both of which can be deduced from Equation (5); $C_{1,m}$ and $C_{2,m}$ are the baseline concentration and simulated counterfactual concentration of pollutant m (μg/m^3), respectively; and b_m is the background concentration (μg/m^3).

Because of data availability, China's PM$_{2.5}$ (or PM$_{10–2.5}$) baseline ($C_{1,m}$) and background (b_m) levels can only be attained by the conversion between PM$_{10}$ and PM$_{2.5}$ (or PM$_{10–2.5}$). Based on a PM$_{10}$ – PM$_{2.5}$ conversion factor of 0.65 used in many previous studies (Lvovsky, 2000; He et al., 2001), the PM$_{10}$ – PM$_{10–2.5}$ conversion factor is set at 0.35 in this framework. According to Wan (2005), the PM$_{10}$ background level in the north of China ranges from 60 to 90 μg/m^3, and here we assume the national level as the lowest (60 μg/m^3). The background concentrations of PM$_{2.5}$ and PM$_{10–2.5}$ (b_m) are consequently 39 and 21 μg/m^3, respectively. In 2012, the mean annual PM$_{10}$ level in China was 91 μg/m^3 (NBS, 2013). The baseline PM$_{2.5}$ (or PM$_{10–2.5}$) level ($C_{1,m}$) is thereby estimated to be 59 μg/m^3 (or 32 μg/m^3) using the same method as for the background level. Given this data, the simulated counterfactual pollutant concentration ($C_{2,m}$) is easily computed.

4. Public Health Module

Subject to the limits of epidemiological and health statistical studies, the quantifiable health endpoints used in this paper include mortality (acute and chronic), respiratory hospital admission, cardiovascular hospital admission, the number of restricted activity days (for adults), work loss days (for adults) and asthma and child bronchitis (for children). The pollution-exposed population is assumed to be China as a whole without the division of rural–urban regions; for the inhaled dose, we make no distinction between indoor and outdoor air pollution.

In order to quantify the health damages caused by air pollution, we adopt a linear exposure–response (ER) function with a zero-threshold assumption. This approach has been used in many studies, such as Quah and Boon (2003), Bell and Ellis (2004) and

---

[11] Here we assume that the natural pollution absorption rate is 100 percent. The historical cumulative inventory of air pollutants is not considered.

Wang and Mauzerall (2006). It is worth noting that if such a threshold does exist, we may overestimate the health damages caused by air pollution. However, it will be useful to depict China's risk of an environment–health–poverty trap. As for the ER coefficient, we prioritize data from China-specific studies and then use data from international literature (Table 2). The number of non-fatal health outcomes $Case_{im}^{Morbidity}$ and acute mortality $Case^{AM}$ cases are described in Equations (7) and (8):

$$Case_{im}^{Morbidity} = ER_{im} \times C_m \times P, \tag{7}$$

$$Case^{AM} = \sum_m ER_m^{AM} \times C_m \times P \times M, \tag{8}$$

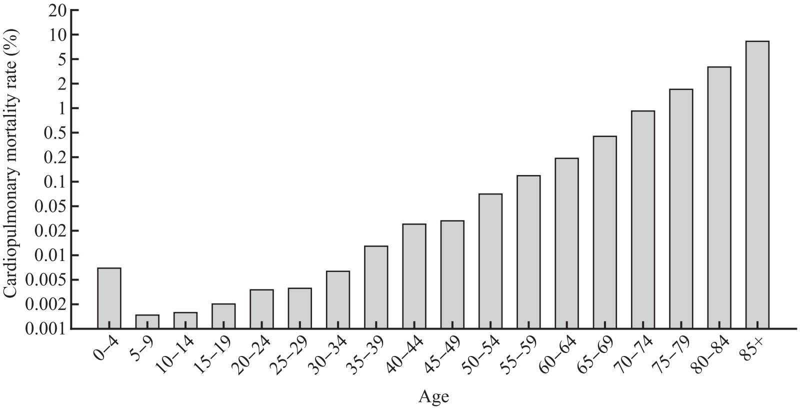
where ER_{im}, C_m and P denote the ER coefficient for non-fatal health endpoint i and pollutant m, the concentration of pollutant m, and exposed population, respectively; and ER_m^{AM} refers to the ER coefficient for acute premature death related to pollutant m, and M the overall death rate.

Table 2. Exposure–Response Coefficients

Health endpoint[a]	Pollutant	Mean (95% CI)[b]	Reference
Acute mortality	$PM_{2.5}$	0.042% (0.003%, 0.081%)	Xie et al. (2011)
	$PM_{10-2.5}$	0.06% (0.04%, 0.08%)	Bickel and Friedrich (2005)
Chronic mortality	$PM_{2.5}$	0.60% (0.40%, 0.80%)	Pope III et al. (2002)
	$PM_{10-2.5}$	0.25% (0.02%, 0.48%)	Pope III et al. (2002)
Respiratory hospital admission	$PM_{2.5}$	2.20E−04 (−1.20E−04, 5.60E−04)	Bell et al. (2008)
	$PM_{10-2.5}$	7.03E−06 (3.83E−06, 1.03E−05)	Bickel and Friedrich (2005)
Cardiovascular hospital admission	$PM_{2.5}$	8.00E−04 (5.90E−04, 1.01E−03)	Bell et al. (2008)
	$PM_{10-2.5}$	4.34E−06 (2.17E−06, 6.51E−06)	Bickel and Friedrich (2005)
Asthma attack	$PM_{2.5}$	2.10E−03 (1.45E−03, 2.74E−03)	Ko et al. (2007)
Restricted activity day (adults)[c]	$PM_{2.5}$	9.02E−02 (7.92E−02, 1.01E−01)	Bickel and Friedrich (2005)
	$PM_{10-2.5}$	5.41E−02 (4.75E−02, 6.08E−02)	Bickel and Friedrich (2005)
Work loss day (adults)	$PM_{2.5}$	2.07E−02 (1.76E−02, 2.38E−02)	Bickel and Friedrich (2005)
	$PM_{10-2.5}$	1.24E−02 (1.06E−02, 1.42E−02)	Bickel and Friedrich (2005)
Bronchitis symptoms (children)	$PM_{2.5}$	6.60E−03 (−3.80E−04, 1.35E−02)	Xie et al. (2009)
	$PM_{10-2.5}$	1.61E−03 (1.24E−04, 3.10E−03)	Holland et al. (1999)

Notes: [a]All exposure–response (ER) coefficients include all age groups except where noted. Because of the limited epidemiological literature, this paper assumes the toxicity of $PM_{10-2.5}$ to be PM_{10}. [b]The unit of ER coefficients for premature death is the increased death rate per μg/m^3 change of a pollutant's concentration, while that for morbidity is case/(year–person–μg/m^3). [c]Here, the health endpoint of restricted days includes work loss days. CI, confidence interval.

Figure 2. Cardiopulmonary Mortality Rate in China, 2007



Source: MOH (2008).

Table 3. Age-conditioned Exposure–Response Coefficients for Chronic Mortalities in China

Pollutant	Age cohort (years)				
	30–44	45–59	60–69	70–79	80–
$PM_{2.5}$[a]	0.213	0.331	0.537	0.708	0.837
$PM_{10-2.5}$[b]	0.089	0.138	0.224	0.295	0.349

Sources: [a]Based on Pope III et al. (2002) and MOH (2008). [b]Referred from Matus et al. (2012).

We estimate the age-conditioned chronic mortality caused by air pollution. As described in Figure 2, the elderly population in China is at much higher risk of cardiopulmonary mortality, which largely results from excessive particulate pollution concentrations (Holland et al., 1999). Thus, referring to the approach of Nam et al. (2010), we estimate China's age-conditioned ER coefficient for chronic mortality with the following equation:

$$ER_{mn}^{CM} = ER_{m}^{CM} \times \frac{M_n^{CPL} / M_n^{All}}{M^{CPL} / M^{All}}, \tag{9}$$

where ER_m^{CM} and ER_{mn}^{CM} are the unconditioned ER coefficients of chronic mortality related to air pollutant m and the age-specific ER coefficient for air pollutant m and age group n (Table 3), respectively; and M^{All} (M^{CPL}) and M_n^{All} (M_n^{CPL}) represent all-cause mortality (or cardiopulmonary mortality) for the whole population and for age group n, respectively. Because chronic disease generally takes many years to develop (Bickel and Friedrich

2005), we assume that people with chronic mortalities are aged above 30. The number of chronic mortalities caused by pollutant m $Case_m^{CM}$ can thus be computed as follows:

$$Case_m^{CM} = \sum\nolimits_n ER_{mn}^{CM} \times C_m \times M_n \times P_n, \tag{10}$$

where M_n and P_n are mortality and population for age group n, respectively.

Table 4. Unit Values for Various Health Endpoints in China (RMB, 2012 Price)

Health endpoint	Unit	Cost	Method	Source
Acute mortality[a]	Case	188,788	VSL	Hammitt and Zhou (2006)
Respiratory hospital admission[b]	Case	5651	COI	MOH (2009, 2013)
Cardiovascular hospital admission[b]	Case	7718	COI	MOH (2009, 2013)
Restricted activity day (adults)[a]	Day	105	WTP	Kan and Chen (2004)
Asthma attack[a]	Case	46	WTP	Kan and Chen (2004)
Bronchitis symptoms (children)[a]	Case	14,041	WTP	Hammitt and Zhou (2006)

Notes: [a]Unit values are computed using the following equation: $W = W_c \times (I / I_c)^e$, where W and I denote health cost and per capita income at the national level in 2012, respectively; W_c and I_c the health cost and per capita income for the selected city in the surveyed year, respectively; and referring to Hammitt and Zhou (2006), this income elasticity e is set at 0.1. [b]Generally speaking, the unit values for hospitalization include hospital admission costs, fees for service and wage losses. As the work loss day is regarded as a single health endpoint in this study, the unit value of hospitalization consists of hospital admission costs and fees for service. Because of data limitations, respiratory and cardiovascular hospital admission costs are represented by bacterial pneumonia and congestive heart failure, respectively. COI, cost of illness; VSL, value of a statistical life; WTP, willingness to pay.

To monetize the health loss caused by air pollution, we employ willingness to pay (WTP), cost of illness (COI) and value of a statistical life (VSL) (Table 4). Additionally, the unit value of workday loss is expressed as the average daily wage level of Chinese workers, endogenously determined within the above-mentioned CGE model. To assign a value to chronic mortality, we assume that Chinese workers generally retire at the age of 60. Therefore, if a worker dies at the age of 40, the cost of chronic mortality can be expressed as a wage loss of 20 years.

Finally, we enter these medical costs and labor losses into our CGE model as shocks to health service demand and labor supply available from a general equilibrium perspective, respectively. Note that the loss of labor supply derives from two health endpoints: workday loss and chronic mortality caused by air pollution. Here, acute mortalities are not considered, as some of these are children or retired workers. Workday loss from chronic mortality is calculated by multiplying the number of deaths of workers aged 30–59 years by 260 (i.e. the average number of workdays for a worker per year). Although this approach may underestimate the economic burden of air pollution, we believe it can provide useful information for environmental policy analysis.

III. Description of the Policy Experiment

In this paper, we focus on the short-term and long-term impacts of the current fuel tax policy on the environment, health and the economic system. Because the health feedback effect from pollution to economic growth is taken into account in policy simulations, it is difficult to employ a recursive dynamic CGE model to estimate the long-term impacts. In this case, similar to Allan et al. (2014), we assume that the short-term results generate the impact in period one, during which both capital stock and labor force are fixed at their base-year values, and the long-run results apply when these production factors have fully adjusted to the disturbance. This study does not assess the feasibility of the fuel tax policy in China, but comprehensively investigates the effects of China's current fuel tax policy through a series of exogenous shocks. As a consequence, this policy simulation is conducted only for gasoline consumption tax without consideration of other refined oils, such as diesel and jet fuel.

As shown in Table 1, China's gasoline consumption tax has been raised from 1 yuan per liter in 2009 to 1.52 yuan per liter in 2015 (Table 1). For simplicity, the fuel tax rate is assumed to be 1.52 yuan per liter in 2012 in the counterfactual scenario, while it is 1 yuan per liter in the baseline scenario. Thus, the exogenous shock is designed as a fuel tax rate rise of 0.52 yuan per liter. Notably, there is no need to take the fuel tax-for-fee (*Fei Gai Shui*) reform into account in the baseline scenario as China phased out the road maintenance fee in 2009.

IV. Results and Discussion

Using the integrated assessment framework, in this section we estimate the impacts of China's current fuel tax with the health feedback. First, the impacts on energy demands and air quality are assessed. The corresponding changes in public health, sectoral output and price are then estimated. Finally, a general review of the macroeconomic and welfare effects is presented.

1. Impacts on Energy Use and Air Quality

Table 5 reports the estimated impacts of China's current fuel tax policy on energy use and air quality. We observe via the price mechanism that the higher fuel tax rate has considerable potential for energy conservation. In particular, fuel oil use decreases most significantly in the short run (12.59 percent), followed by its upstream, that is, crude oil and gas (12.12 percent). Meanwhile, coal and electric power consumption also decline by 6.54 and 7.51 percent, respectively, mainly because of the substantial loss in economic production. In the long run, as firms have enough time to adjust their industrial structure, fossil energy can partly be substituted by value-added factors as intermediate inputs. Along with the economic recovery, this would finally stimulate energy use and diminish the foregone energy conservation effect.

As air pollution stems from the excessive use of fossil energy in most cases, the higher fuel tax rate inevitably results in significant improvement of air quality. In the short run, China's $PM_{2.5}$ and $PM_{10–2.5}$ emissions drop largely by 6.95 and 6.80 percent, respectively; and the pollution levels are accordingly reduced by 1.39 $\mu g/m^3$ and 0.74 $\mu g/m^3$, respectively. Yet in the long run, the pollution mitigation effect is relatively weak because of the economic recovery. In this sense, the simulation result has shown an important pollution mitigation implication of fuel taxation in China, especially in the short run.

Table 5. Impacts of China's Current Fuel Taxation on Energy Saving and Pollution Mitigation

	Short-term	Long-term
Crude oil and gas (%)	–12.12	–8.46
Fuel oil (%)	–12.59	–9.09
Coal (%)	–6.54	–2.09
Electric Power (%)	–7.51	–3.34
$PM_{2.5}$ emissions (%)	–6.95	–2.44
$PM_{10–2.5}$ emissions (%)	–6.80	–2.21
$PM_{2.5}$ concentration ($\mu g/m^3$)	–1.39	–0.48
$PM_{10–2.5}$ concentration ($\mu g/m^3$)	–0.74	–0.24

Source: Authors' own calculation.

2. Impact on Public Health

Even small reductions in population exposure to risk lead to substantial health gains (Rose, 2001). As Table 6 shows, because of the considerable improvement in air quality, public health would significantly improve. In the short term, an increase in the fuel tax rate would save around 0.09 million lives in China, almost 9 percent of the annual average premature deaths caused by air pollution.[12] Meanwhile, cardiovascular and respiratory hospital admissions and asthma attacks would also decline dramatically. In a general equilibrium context, these health damages would impact the macroeconomy with a huge increase in the available labor supply. As the labor force is a key driver of economic development, its increase would naturally benefit the economy. Furthermore, a healthier worker will be more productive (Liu et al., 2008; Zivin and Neidell, 2012). In the case of China, in particular, the aging population will inevitably result in a structural labor shortage, and thus sufficient supply of a skilled labor force is vital for China to sustain rapid economic growth (Cai and Wang, 2006). Under these circumstances, China's current fuel taxation policy yields great economic benefit by increasing the available labor force. In addition, the increased labor supply would accordingly push up the total household income. Along with the drop in additional health expenses, this would finally lead to a substantial increase in total disposable income (RMB85.70bn), which can be also regarded as a total monetized health benefit. In this case, the current fuel taxation has important implications for developing a healthy China.

Table 6. Impact of China's Current Fuel Taxation on Public Health

Health endpoint	Short-term		Long-term	
	Case (1000)	Monetary value (RMBbn)	Case (1000)	Monetary value (RMBbn)
Acute mortality	–9	–1.83	–3	–0.62
Chronic mortality	–78	–1.9	–27	–0.66
Respiratory hospital admission	–421	–2.38	–148	–0.83
Cardiovascular hospital admission	–1510	–11.65	–531	–4.09
Restricted activity days (adults)	–134,812	–14.15	–46,516	–4.88
Work loss days (adults)	–30,928	–2.28	–10,672	–0.79
Asthma attack	–3954	–0.18	–1390	–0.06
Bronchitis symptoms (children)	–3655	–51.33	–1274	–17.88
Total		–85.70		–29.81

Source: Authors' own calculation.

---

[12] According to Lelieved et al. (2005), global air pollution causes over 3 million premature deaths every year, a third of which are in China.

Obviously, the improvement to public health would be weak in the long run as economic recovery would naturally lead to an increase in air pollutant emissions, regardless of advanced abatement technology. Nevertheless, China's current fuel tax policy will lead to substantial improvement in public health.

3. Impacts on Sectoral Output and Price

Table 7 shows that the refined oil price increases by 18.35 percent in the short run. It is anticipated that such a huge increase in the refined oil price exerts upward pressure on the prices of energy-intensive products, followed by other non-energy-intensive products, and thereby depresses outputs in all sectors through a cost-push shock. Moreover, such upward pressure on price is particularly striking for the sectors closely related to the refined oil sector. For example, prices in the energy-intensive sector, household transportation and other purchased transportation would increase by 12.59, 12.21 and 25.38 percent, respectively. Regarding sectoral output, other industrial transportation suffers the largest decrease (15.32 percent), followed by the crude oil and gas sector (13.46 percent), refined oil (12.65 percent) and other industries (8.52 percent).[13] In contrast, as households use public transport for daily travel, the decrease in the output of urban public transportation seems relatively weak (6.86 percent). This result indicates that China's current fuel tax plays a significant role in promoting green transport.

Table 7. Impacts of China's Current Fuel Tax on Sectoral Output and Price

Sector	Short-term (%)		Long-term (%)	
	Output	Price	Output	Price
Agriculture	–8.12	12.48	–4.22	5.59
Service	–5.83	14.03	–3.54	7.04
Energy-intensive industries	–8.39	12.59	–4.08	5.67
Other industries	–8.52	11.03	–3.82	4.17
Health service	–3.66	10.91	–2.08	4.11
Crude oil and gas	–13.46	12.26	–10	5.39
Refined oil	–12.65	18.35	–9.32	11.1
Coal	–5.95	9.72	–1.68	2.99
Electricity	–7.3	11.11	–3.33	4.29
Urban public transportation	–6.86	15.75	–4.96	8.65
Road transportation	–7.11	14.55	–3.8	7.51
Other industrial transportation	–15.32	25.38	–12.12	17.65
Household transportation	–7.97	12.21	–6.56	5.29

Source: Authors' own calculation.

---

[13] The output of other industries falls by such a high magnitude mainly because this sector includes vehicle manufacturers and automotive components closely linked to refined oil use.

In the long run, as producers will be able to lower their operation costs via structural adjustment, the sectoral outputs will be increased, and the prices will obviously fall compared to the prices in the short run. The loss of household transportation output reduces from 7.97 percent in the short run to 6.56 percent in the long run, and this output improvement is the smallest among all sectors. Although the burden of fuel taxation levied in the production process would finally shift from refined oil manufacture to the whole economy, the rebound of household transportation output (i.e. private transportation) seems somewhat slow in the long run.

4. Impacts on the Macroeconomy and Social Welfare

We finally turn to the macroeconomic and social welfare impacts of China's current fuel tax policy. As shown in Table 8, even if the improved health outcomes benefit the economy by increasing the available labor supply and reducing medical expenses, the real GDP would also drop by 7.35 percent in the short term. This can be plausibly explained from both the supply and demand perspectives. On the supply side, the rise in fuel tax would drive up all commodity prices via substantial inter-industry linkage, the sectoral output profits will be reduced and producers will correspondingly reduce production. On the demand side, as commodity prices increase, producers prefer to substitute the intermediate inputs with value-added factors and reap the increase in returns to labor and capital; the increase of household disposable income (5.68 percent) is not large enough to completely counteract the rise in the consumer price index (CPI) (12.63 percent), and hence, could significantly reduce household real consumption (6.04 percent) and further cause social welfare loss measured by real purchasing power in terms of Hicksian equivalent variations (RMB913.71bn).[14] In addition, the increase in domestic output prices would somewhat lower export competitiveness, thus resulting in a reduction of total export (8.78 percent). Meanwhile, the total import will also drop by 8.25 percent as a result of the decline in domestic demand.

Over time, capital and labor factors will be able to freely enter or exit the market. In the long run, the average wage will fall. With increasing investment and capital stock, return to capital will also be reduced to an extent. Once the production cost drops, it will induce a drop in the output prices and then in CPI. As a result of stimulated household consumption demand, real GDP loss will be decrease from 7.35 percent in the short run to 3.83 percent. Moreover, the loss in social welfare will fall from RMB913.71bn to

---

[14] When computing social welfare, we do not include health status as the relevant data cannot be determined in the calibration of the CGE model. This approach, found in Matus *et al*. (2012) and Chen and He (2014), may overestimate the welfare loss associated with the current fuel taxation policy, but provides useful information for policymakers.

RMB699.63bn. We do not consider the consumption of health services in the valuation of welfare because the decreased consumption of health services resulting from improved health outcomes represents an improvement to citizens.

Compared to previous studies (e.g. Lai et al., 2008; Xiao and Lai, 2009; Xia and Liu, 2010), our estimation of the economic loss resulting from China's fuel taxation policy is significantly higher, mainly because of the use of different benchmark data. We set the base year as 2012, while previous studies have used 1997. It is widely believed that China has experienced many dramatic changes over these 15 years, particularly to economic structure. According to the NBS (2012, 2013), the domestic use of refined oil was approximately 8 percent of GDP in 2012, while this proportion was 4 percent in 1997. Thus, China's economy depends on refined oil more than ever before. Thus, our estimation of the positive shock to the fuel tax rate suggests greater economic loss than previous studies. In this regard, we believe that our policy analysis using the latest data provides a more accurate account of the economic effects of China's current fuel taxation policy.

Table 8. Macroeconomic Impact of China's Current Fuel Tax Policy

Macroeconomic variable	Short-term	Long-term
Real GDP (%)	–7.35	–3.83
Household welfare (RMBbn)	–913.71	–699.63
Average wage level (%)	6.47	0
Capital price (%)	11.15	4.24
Employment (%)	0	0.93
Capital stock (%)	0	–5.34
Real consumption (%)	–6.04	–4.6
Real export (%)	–8.78	–4.31
Real import (%)	–8.25	–4.11
GDP deflator (%)	3.97	1.26
Consumer price index (%)	12.63	5.71
Total tax revenue (%)	5.33	3.12
Household disposable income (%)	5.68	0.85

Source: Authors' own calculation.

The positive impact of improved public health is not sufficient to offset the negative impact caused by the rise in commodity prices on GDP and social welfare. Therefore, GDP and social welfare would decline in both the short-term and long-term. As a result, China's current fuel tax policy is insufficient to reduce the potential environment–health–poverty trap risk.

V. Conclusions and Policy Implications

In this paper, we examined the effects of China's current fuel tax policy on energy, environment, health, the economy and social welfare using an integrated assessment framework based on a CGE model. We find that the fuel tax policy may significantly reduce fossil energy consumption, improve air quality, and thereby achieve a substantial health benefit of RMB85.70bn in the short run; however, in the long run, these positive effects will gradually weaken as a result of changes to the economic structure. Even if health improvement could promote economic growth, the current fuel tax policy would produce a staggering GDP decline of 3.83–7.35 percent and residential welfare loss of RMB699.63–913.71bn. This suggests that China's current fuel tax policy has limited potential to overcome the environment–health–poverty trap. Based on these findings, some policy implications are summarized as follows.[15]

Raising the fuel tax rate is not a sustainable method of environmental protection. Without doubt, a rise in the fuel tax rate would improve air quality and human health. However, such environmental improvement may be accompanied by sizable GDP and welfare losses, and its effect on energy conservation and emission abatement become gradually weaker over time. As a result, the fuel tax rate plays a limited role in addressing the conflict between economic growth and environmental protection. In practice, tax policies include tax rates, collection processes and the allocation of tax revenues. As far as fuel taxation is concerned, adjusting the fuel tax rate is the most convenient method for the government but it is not a panacea. In light of the potential environment–health–poverty trap risk, our research is intended to inform both Chinese residents and the government that further tax reform is required, and to provide evidence of concerns over further tax reform.

Increases to fuel tax rates should be made with caution to ensure the right timing.

---

[15] Because the fuel tax policy could induce a substitution effect between energy products (fuel oil, coal, electric power and petroleum) and value-added inputs through the price mechanism, their substitution elasticities in the production function will be key to the extent of energy conservation achieved. Thus, we conducted sensitivity analysis of these elasticities and find that our general conclusions still hold.

Our results indicate that a fuel tax levy to production of 0.52 yuan per liter would lead to a CPI increase as high as 5.71–12.63 percent. The main barrier to fuel tax reform may be public concern over soaring prices that threaten social stability. When the international crude oil price is relatively low, raising the fuel tax rate may lead to higher oil prices and a higher CPI. Thus, reform of the oil pricing method might be needed, with the final price being constant or even falling. Such reform will be welcomed by the public as they are more interested in whether reform will lead to a refined oil price hike, rather than a change to the pricing method.

References

Allan, G., P. Lecca, P. McGregor and K. Swales, 2014, “The economic and environmental impact of a carbon tax for Scotland: A computable general equilibrium analysis,” *Ecological Economics*, Vol. 100, April, pp. 40–50.

Bell, M. and H. Ellis, 2004, “Sensitivity analysis of tropospheric ozone to modified biogenic emissions for the mid-atlantic region,” *Atmospheric Environment*, Vol. 38, No. 13, pp. 1879–89.

Bell, M. L., K. Ebisu, R. D. Peng et al., 2008, “Seasonal and regional short-term effects of fine particles on hospital admissions in 202 US counties, 1999–2005,” *American Journal of Epidemiology*, Vol. 168, No. 11, pp. 1301–10.

Bickel, P. and R. Friedrich, 2005, *ExternE – Externalities of Energy: Methodology 2005 Update*, Luxembourg: European Commission.

Cai, F. and M. Y. Wang, 2006, “Challenge facing China’s economic growth in its aging but not affluent era,” *China & World Economy*, Vol. 14, No. 5, pp. 20–31.

Cao, J., 2007, *Essays on Environmental Tax Policy Analyses: Dynamic Computable General Equilibrium Approaches Applied to China*, PhD dissertation, Harvard University, Cambridge, MA.

Chen, S. M. and L. Y. He, 2014, “Welfare loss of china’s air pollution: How to make personal vehicle transportation policy,” *China Economic Review*, Vol. 31, pp. 106–18.

Chen, S. M. and L. Y. He, 2017, “Environment, health and economic growth: The optimal allocation of energy tax revenue,” *Jingji Yanjiu* (*Economic Research Journal*), No. 4, pp. 120–34.

Hammitt, J. K. and Y. Zhou, 2006, “The economic value of air-pollution-related health risks in China: A contingent valuation study,” *Environmental and Resource Economics*, Vol. 33, No. 3, pp. 399–423.

He, K. B., F. M. Yang, Y. L. Ma et al., 2001, “The characteristics of PM2.5 in Beijing, China,” *Atmospheric Environment*, Vol. 35, pp. 4959–70.

Hibiki, A. and T. Arimura, 2005, “An empirical study of the effect of the fuel tax in Japan on vehicle selection and NOx emission,” *Department of Social Engineering Discussion Paper* No. 05–02, Tokyo Institute of Technology, Tokyo.

Holland, M., J. Berry and D. Forster, 1999, *ExternE – Externalities of Energy, Vol. 7: Methodology 1998 Update*, Luxembourg: European Commission.

Jiang, L., 2006, “Integrated impact assessment model of environmental policy and its application,” *Huanjing Kexue* (*China Environmental Science*), No. 5, pp. 1035–40.

Kan, H., Chen B., 2004, “Particulate air pollution in urban areas of Shanghai, China: Health-based economic assessment,” *Science of the Total Environment*, Vol. 322, No. 1, pp. 71–79.

Klimont, Z., J. Cofala, I. Bertok, M. Amann, C. Heyes and F. Gyarfas, 2002, *Modelling Particulate Emissions in Europe: A Framework to Estimate Reduction Potential and Control Costs*, Interim Report IR-02-076, International Institute for Applied Systems Analysis, Laxenburg.

Ko, F. W., W. Tam, T. W. Wong et al., 2007, “Effects of air pollution on asthma hospitalization rates in different age groups in Hong Kong,” *Clinical & Experimental Allergy*, Vol. 37, No. 9, pp. 1312–19.

Lai, M. Y., H. Xiao, W. Chen and S. J. Zhu, 2008, “Dynamic general equilibrium analysis of fuel tax collection in different stages and policy options,” *Shijie Jingji* (*The Journal of World Economy*), No. 1, pp. 65–76.

Lelieved, J., J. S. Evans, M. Fnais, D. Giannadaki and A. Pozzer, 2005, “The contribution of outdoor air pollution sources to premature mortality on global scale,” *Nature*, Vol. 525, No. 7569, pp. 367–71.

Liu, G. G., W. H. Dow, A. Z. Fu, J. Akin and P. Lance, 2008, “Income productivity in China: On the role of health,” *Journal of Health Economics*, Vol. 27, No. 1, pp. 27–44.

Lvovsky, K. 2000, “Environmental costs of fossil fuels: A rapid assessment method with application to six cities,” *Environment Department Paper* No. 78, World Bank, Washington DC.

Matus, K, K. M. Nam, N. E. Selin, L. N. Lamsal, J. M. Reilly and S. Paltsev, 2012, “Health damages from air pollution in China,” *Global Environmental Change*, Vol. 22, No. 1, pp. 55–66.

MOF (Ministry of Finance of the China), 2013, *Finance Yearbook of China 2013*, Beijing: China Finance Magazine (in Chinese).

MOH (Ministry of Health of China), 2008, *China Health Statistics Yearbook 2007*, Beijing: Peking Union Medical College Press (in Chinese).

MOH (Ministry of Health of China), 2009, *An Analysis Report of National Health Services Survey in China 2008*, Beijing: Peking Union Medical College Press (in Chinese).

MOH (Ministry of Health of China), 2013, *China Health Statistical Yearbook 2010*, Beijing: Peking Union Medical College Press (in Chinese).

Nam, K. M., N. E. Selin, J. M. Reilly and S. Paltsev, 2010, "Measuring welfare loss caused by air pollution in Europe: A CGE analysis," *Energy Policy*, Vol. 38, No. 9, pp. 5059–71.

NBS (National Bureau of Statistics of China), 2012, *Input–Output Table of China 2012*, Beijing: China Statistics Press (in Chinese).

NBS (National Bureau of Statistics of China), 2013, *China Statistical Yearbook 2013*, Beijing: China Statistics Press (in Chinese).

Pang, J., J. Zou and S. Fu, 2008, "Analysis of the economic impact of China's fuel tax collection using a CGE model," *Jingji Wenti Tansuo* (*Inquiry into Economic Issues*), No. 11, pp. 69–73.

Pope III, C. A., R. T. Burnett, M. J. Thun et al., 2002, "Lung cancer, cardiopulmonary mortality, and long-term exposure to fine particulate air pollution," *The Journal of the American Medical Association*, Vol. 287, No. 9, pp. 1132–41.

Qi, Y. and H. Y. Lu, 2015a, "Mechanism of environment–poverty trap and China's environmental turning point," *Zhongguo Renkou Ziyuan Yu Huanjing* (*Chinese Journal of Population, Resources and Environment*), No. 10, pp. 71–8.

Qi, Y. and H. Y. Lu, 2015b, "Pollution, health and inequality: Transcending the environment–health–poverty trap," *Guangli Shijie* (*Management World*), No. 9, pp. 32–51.

Quah, E. and T. L. Boon, 2003, "The economic cost of particulate air pollution on health in Singapore," *Journal of Asian Economics*, Vol. 14, No. 1, pp. 73–90.

Rose, G., 2001, "Sick individuals and sick populations," *International Journal of Epidemiology*, Vol. 30, No. 3, pp. 427–32.

Selin, N. E., S. Wu, K. M. Nam et al. 2009, "Global health and economic impacts of future ozone pollution," *Environmental Research Letters*, Vol. 4, No. 4, pp. 1–9.

Shen, J. Y., 2006, "A simultaneous estimation of environmental Kuznets curve: Evidence from China," *China Economic Review*, Vol. 17, No. 4, pp. 383–94.

Sterner, T., 2007, "Fuel taxes: An important instrument for climate policy," *Energy Policy*, Vol. 35, No. 6, pp. 3194–202.

Wan, Y., 2005, *Integrated Assessment of China's Air Pollution-induced Health Effects and Their Impacts on National Economy*, PhD dissertation, Department of Social Engineering, Tokyo Institute of Technology, Tokyo.

Wang, X. and D. L. Mauzerall, 2006, "Evaluating impacts of air pollution in China on public health: Implications for future air pollution and energy policies," *Atmospheric Environment*, Vol. 40, No. 9, pp. 1706–21.

Wang, X., D. L. Mauzerall, Y. Hu et al., 2005,"A high-resolution emission inventory for eastern China in 2000 and three scenarios for 2020," *Atmospheric environment*, Vol. 39, No. 32, pp. 5917–33.

Watts, J., 2005, "China: The air pollution capital of the world," *Lancet*, Vol. 366, No. 9499, pp. 1761–62.

WHO (World Health Organization), 2006, *WHO Air Quality Guidelines for Particulate Matter, Ozone, Nitrogen Dioxide and Sulfur Dioxide: Global Update 2005*, Geneva: World Health Organization.

WHO (World Health Organization), 2013, *Health Effects of Particulate Matter: Policy Implications for Countries in Eastern Europe, Caucasus and Central Asia (2013)*, Copenhagen: WHO Regional Office for Europe.

Xia, C. W. and Y. W. Liu, 2010, “A dynamic CGE research of the fuel tax reform on China’s energy-saving emission reduction,” *Jingji Wenti* (*On Economic Problems*), No. 2, pp. 64–9.

Xiao, H., 2009, *Dynamic General Equilibrium Analysis of China’s Fuel Tax Collection during Financial Crisis and Policy Optimization*, Hunan: Hunan University Press (in Chinese).

Xiao, H. and M. Y. Lai, 2009, “Dynamic general equilibrium analysis of fuel tax reform,” *Jingji Shuxue* (*Mathematics in Economics*), No. 3, pp. 53–9.

Xie, P., X. Y. Liu, Z. R. Liu, T. T. Li, L. J. Zhong and Y. R. Xiang, 2011, “Human health impact of exposure to airborne particulate matter in pearl river delta, China,” *Water, Air, & Soil Pollution*, Vol. 215, No. 1–4, pp. 349–63.

Xie, P., X. Y. Liu, Z. R. Liu, T. T. Li and Y. H. Bai, 2009, “Study on particulate pollution exposure–response relationship for Chinese populations,” *Zhongguo Huanjing Kexue* (*China Environmental Science*), Vol. 29, No. 10, pp. 1034–40.

Xie, Y., H. C. Dai, T. Hanaoka and M. Toshihiko, 2016, “Health and economic impacts of PM2.5 pollution for people living in Beijing–Tianjin–Hebei region,” *Zhongguo Renkou Ziyuan Yu Huanjing* (*Chinese Journal of Population, Resources and Environment*), No. 11, pp. 19–27.

Ye, Z. H., 2009, “Determination of fuel tax rates: CGE-based analysis,” *Tongji Yanjiu* (*Statistical Research*), No. 5, pp. 86–93.

Zivin, G. J. and M. Neidell, 2012, “The impact of pollution on worker productivity,” *American Economic Review*, Vol. 102, No. 7, pp. 3652–73.

(Edited by Yuanfang Li)