Energy Economics 49 (2015) 451–459

Contents lists available at ScienceDirect

Energy Economics

journal homepage: www.elsevier.com/locate/eneco

A method for analyzing pollution control policies: Application to SO_2 emissions in China

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ARTICLE INFO

Article history:
Received 13 May 2014
Received in revised form 25 January 2015
Accepted 20 March 2015
Available online 2 April 2015

JEL classification:
Q01
Q40
Q52
Q53
Q56
Q58

Keywords:
SO_2
Emission price
Emission rate
Abatement cost
Pollution levy
Cap and trade

ABSTRACT

Market-based pollution control mechanisms such as pollution levy and cap and trade have received increasing attention from both academics and practitioners. A good understanding of the optimal pollution price under these mechanisms is a premise for regulators to make sound pollution control policies. In this paper, we propose a method for deriving the optimal pollution price for a given pollution target. This method consists of two steps that integrate cost function estimation and market equilibrium analysis: First, historical data is used to estimate the pollution abatement cost functions of the polluters; second, market models are used to solve the equilibrium pollution price under each control mechanism. For illustration, we apply the method to investigate SO_2 emission control policies in China, using a dataset of SO_2 emissions and abatement costs from three major industry sectors (Electricity, Gas, and Water Supply; Manufacturing; and Mining). Our analysis shows that the optimal levy rate is significantly higher than the actual rate adopted by the Chinese government. For example, the optimal levy rate should be 4.92 RMB/kg, while the actual rate is 1.26 RMB/kg in 2010. As a result, the actual emission structure is much less efficient: The overall cost savings would be 49.7% for all three sectors during 2006–2010 if the optimal emission structure is achieved. These findings have useful policy implications for the Chinese government. In addition, the method may be applied to analyze control policies at different aggregate levels (for example, provincial economies) or for other pollutants (for example, CO_2 and chemical oxygen demand).

1. Introduction

China has experienced rapid economic growth during the past few decades. Although economic development has lifted a vast Chinese population out of poverty, increased energy consumption accompanying industrialization and urbanization has led to severe air quality problems (Hao et al., 2007; Yi et al., 2007; Chan and Yao, 2008; Schreifels et al., 2012). In recent years, smog caused by worsening air quality in large cities including Beijing and Shanghai has frequently made headlines in the Chinese media. According to the Ministry of Environmental Protection of China, more than 80% of Chinese cities failed to meet the national air quality standard in December, 2013 (Wu, 2014). Polluted air may cause serious environmental damage (such as acid rain) as well as health-related problems (such as respiratory diseases). For example, an air pollutant of the most concern is the so-called fine particulate matter (PM), which accounts for approximately 800,000 premature deaths worldwide in urban areas alone each year (Johnson et al., 2011). As a matter of fact, it is the high concentration of $PM_{2.5}$ and PM_{10} that has caused the notoriously polluted air in Chinese cities (Shukman, 2014). One of the major contributors to $PM_{2.5}$ in China is atmospheric SO_2 emissions (Li et al., 2009; Pathak et al., 2009; Mo et al., 2013), which can be largely attributed to energy consumption. For example, coal was used to meet approximately 69% of China's total primary energy demand, and 85% of SO_2 emissions were from direct coal combustion in 2000 (International Energy Agency, 2002; Yang et al., 2002). According to the European Environment Agency (EEA), energy production and distribution alone accounts for 70% of overall SO_2 emissions in 2009 among EEA member countries.[1]

It has been widely recognized that rampant environmental pollution has led to social unrest and put China's long-term economic growth at risk (Zhang, 2013). To fight the air pollution problem, the Chinese government has carried out a series of SO_2 emission control policies over the past four decades. Ellerman (2002) provides a detailed review of the evolution of these environmental policies in China from the 1970s to 2000. In the late 1970s and 1980s, the main approach to regulating

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[1] http://www.eea.europa.eu/data-and-maps/indicators/eea-32-sulfur-dioxide-so2-emissions-1/assessment-1.

http://dx.doi.org/10.1016/j.eneco.2015.03.015
0140-9883/

SO_2 emissions was *command and control*. The government selected specific facilities to be put under control, and directly allocated resources to abate the SO_2 emissions from these facilities. Most existing facilities were unaffected because they were not designated for control at that time. Thus, this centrally directed, project-specific approach had a rather limited effect. In 1982, the first general measure to address SO_2 emissions, the *pollution levy*, was applied to industrial SO_2 emissions.[2] In April 2000, the People's Congress amended the 1987 Air Pollution Prevention and Control Law (APPCL) to shift the emphasis of control from emission rates to total emission discharges, change the base of the pollution levy from excess emissions to total emissions, and establish emission permits as the vehicle by which national policy will be implemented at the local level.

In both the 10th (2001–2005) and 11th (2006–2010) Five Year Plan (FYP) periods, the Chinese government established national goals to reduce SO_2 emissions by 10%. The target was not met in the 10th FYP, whereas it was over-achieved (with a 14.29% reduction) in the 11th FYP. Schreifels et al. (2012) have identified a variety of factors that contributed to this achievement. In the 12th FYP (2011–2015), the Chinese government again set targets to reduce energy intensity, CO_2 emissions per unit GDP, chemical oxygen demand (COD), and SO_2 emissions by 16%, 17%, 8%, and 8%, respectively. To achieve these targets, the government emphasizes the use of market-based instruments such as *cap and trade* as potential valuable policy tools.

Under cap and trade, the regulator allocates emission permits to polluters who may then trade among themselves. The first large-scale application of such a mechanism is the U.S. Acid Rain Program, which has been considered successful in general (Ellerman et al., 2000). In 2002, SO_2 emission trading was suggested by the State Council of China as part of its 10th FYP for preventing and controlling acid rain, but it turned out to be largely unsatisfactory (Chang and Wang, 2010; Han et al., 2012; Hill, 2013; Zhang et al., 2013). The success of the trading schemes was mainly constrained by lack of sufficient administrative capacity, a lagging-behind legal system, and unsatisfactory emission measurement accuracy (Tao and Mah, 2009; Chang and Wang, 2010). Xue et al. (2014) point out that a lack of national regulations and weak enforcement of planned missions are also obstacles to the implementation of emission trading. Through a case study of the sulfur dioxide market in Jiangsu province, Zhang et al. (2013) demonstrate that pre-existing environmental regulations have a significant impact on the performance of the emission trading market. Building a fully autonomous market with strong legal support for emission trading in China is expected to take time, and its successful implementation still requires many experiments.

Nevertheless, it is clear that the future trend favors market-based mechanisms, i.e., the pollution levy or cap and trade mechanism. In the academic literature, much attention has been devoted to the comparison of these mechanisms. It has been shown that under certain conditions, there is broad equivalence between the mechanisms in the sense that they can both achieve the most efficient market outcome (Pezzey, 1992; Farrow, 1995; Ekins and Barker, 2001; Ellerman, 2002). However, in practice the two instruments differ significantly in terms of implementation costs, information requirements, and distribution characteristics. After examining the European Union Emission Trading System (EU ETS) in terms of emission reductions and cost to the public, Wittneben (2009) draws the conclusion that a carbon tax lowers emissions quickly with lower costs to the public and has no upper bounds in terms of reduction potential. Pizer (2009), considering the potential long-term damages of climate change and the costs of emission control, also favors price-based policy instruments (e.g., emissions tax). On the other hand, Keohane (2009) argues that cap and trade gives important flexibility to resolve distribution issues, and also promotes cost-effective abatement and broad participation in the context of international policy. Goulder and Schein (2013) summarize the dimensions along which carbon tax and emission trading produce different outcomes, and the dimensions that do not discriminate between the options.

There are certain challenges associated with the implementation of both mechanisms. For the emission trading mechanism, it would be helpful if the regulator is able to predict, for any imposed emission cap, the approximate equilibrium trading price and the resulting emissions from each polluter. This may help the regulator set an appropriate emission cap and devise a reasonable initial allocation. The benefit of a well-balanced initial allocation could be substantial because it can reduce both the transaction costs and the time for the market to reach equilibrium. The pollution levy, as a long-standing instrument to control emissions of all types, is more straightforward to implement. However, setting an appropriate levy rate is a significant challenge. In China, the levy rate for SO_2 emissions was set to 0.63 RMB/kg starting July 1, 2005. Then in 2007, the Chinese government planned to double the rate to 1.26 RMB/kg within three years.[3] The levy rate has not been changed since. It is widely believed that the current levy rate of SO_2 emission is less than the marginal abatement cost of SO_2, and hence insufficient to stimulate abatement to the necessary levels (Schreifels et al., 2012; Zhang, 2013). We can see that the appropriate pricing of emission is the cornerstone for both mechanisms (Hill, 2013). Unfortunately, in practice, such information is rarely readily available to the regulator, which greatly hinders the success of these market-based emission control policies.

Based on the U.S. SO_2 trading program, Boutabba et al. (2012) study empirically the determination and the dynamic behavior of SO_2 emission prices. They find the existence of a long-term relationship between SO_2 emission price, scrubbing costs, industrial production, and weather conditions. Without a well-established SO_2 trading program in China, this paper proposes a method to derive the optimal emission price for a given emission target based on historical data of SO_2 emissions and abatement costs. The method is composed of two steps, and we apply it to SO_2 emission control in China during the time period 2001–2010. Due to data availability, we focus on three industry sectors that are major contributors of SO_2 emissions in China: Electricity, Gas, and Water Supply (EGWS); Manufacturing; and Mining. In the first step, regression analysis is used to estimate the abatement cost functions for all three sectors. This step is related to the existing studies that examine the abatement cost structure for specific industry sectors; see, for example, Welsch (1988) and Kwon and Yun (1999) for the power generation sector. The World Bank has identified the abatement cost as a function of emission carrier volume and emission rate (Hettige et al., 1995; Wang and Wheeler, 1996; Dasgupta et al., 1997), which is not restricted to a particular industry and can be applied at any aggregate level. So we adopt the functional format proposed by the World Bank in our analysis. In the second step, based on the abatement cost functions, the equilibrium emission price and the associated market outcome (emission rates, abatement costs, etc.) under each mechanism are solved. Although theoretical models have been extensively studied for the two mechanisms in the literature, to our knowledge, there is little research that derives the equilibrium emission price by integrating cost function estimation and market equilibrium analysis. There are several major findings that might be useful in policy making for the Chinese regulators.

First, the optimal emission price (either equilibrium trading price or levy rate) obtained from our model is significantly higher than the actual levy rate used in practice. The optimal levy rates are 1.82 RMB/kg and 4.92 RMB/kg in 2007 and 2010, respectively, while the actual levy rate has been kept at 1.26 RMB/kg during 2007–2010. From 2006 to 2010, the optimal emission price would have been increased by 2.97 times, since SO_2 generation went up by 48% while the emission target was lowered by 15%.

[2] According to Ellerman (2002), "the use of the term 'levy' conveys an important legal distinction, denoting that the payment is not a tax falling within the jurisdiction of the national authorities, but a fee imposed and collected at the local level."

[3] Local governments are allowed to raise the levy rate above the national levels. For example, Jiangsu province doubled the levy rate for SO_2 emissions to 1.26 RMB/kg from July 1, 2007 onward (Zhang, 2013), while Guangdong province raised the rate to 1.26 RMB/kg from April 1, 2010 onward (http://www.gdpi.gov.cn/jgzc/290885.htm).

Second, the actual emission structure in practice is much less efficient than the optimal one predicted by our model. Because the levy rate is too low, polluters have weak incentives to cut emissions, so the regulator must use supplemental administrative instruments to further enforce SO_2 emission reduction. Due to a lack of understanding of the most efficient market outcome, the inappropriate administrative instruments have led to excessive abatement costs. We find that if the optimal market outcome is reached, the total cost savings during 2006–2010 can be as high as 49.7%. Close scrutiny shows that the Mining sector's SO_2 emission amount and abatement costs are insignificant compared to the other two sectors. In contrast to the actual emission structure, the optimal emission structure requires the Manufacturing sector to emit a smaller portion of their generated SO_2, while allowing the EGWS sector to emit a larger portion of their SO_2 from 2007 on. This is because the abatement cost in the EGWS sector is much more sensitive to the emission rate than that of the Manufacturing sector. The suboptimal emission structure in practice explains the remarkably higher abatement costs than those in the optimal outcome. It also points to a potential direction for the regulator to adjust the emission targets imposed on different industry sectors.

Third, since pollution levy is a main instrument for emission control in China, it is important for regulators to understand the financial burden firms have to undertake under the optimal levy rate. Our analysis indicates that when the optimal levy rate is adopted, the levy payment represents a significant portion of the total costs firms have to incur: For the three sectors as a whole, the ratio between the total levy payment and the total abatement costs increases from 76% in 2006 to 99% in 2010. Hence, despite its effectiveness in achieving the most efficient emission outcome, the optimal levy rate is unlikely to be implemented without subsidies or refunds. We show that one possible solution is to couple the pollution levy with tradable permits. By adjusting the allocated free quota and the levy rate, the regulator may arbitrarily adjust the financial burden imposed on the firms. The hybrid system reduces to a pure cap and trade mechanism if the levy rate is set to zero. In this case, obtaining the equilibrium trading price and market outcome may help the regulator set the appropriate initial quota allocation scheme so that the required transaction volume is minimized and the market can reach the desired equilibrium at a fast speed.

The rest of the paper is organized as follows. Section 2 introduces the models of the two emission control mechanisms. These models will be used later to derive the optimal emission price. The main analysis and results are presented in Section 3. In this section, we first describe the data used for analysis. Then we use regression to estimate the parameters in the abatement cost function for each sector. Based on the abatement cost functions, the optimal emission price and the associated emission behavior are obtained and compared to the actual market outcome. At the end of this section, we discuss how this method may be applied at different aggregate levels and to control other pollutants. Finally, this paper concludes with Section 4.

2. Model

We consider an economy consisting of n different industry sectors. Each sector engages in production activities that generate SO_2 emissions. The production activities for the firms in each sector are assumed to be exogenously given; however, the associated SO_2 emissions can be reduced by taking costly measures. For sector i, let W_i denote the annual SO_2 carrier in volume generated by the production activities in this sector,[4] and let g_i be the initial pollutant concentration. All SO_2-emitting firms need to decide how much of the emissions should be abated. Let η_i $(0 \leq \eta_i \leq 1)$ stand for the emission rate of sector i, which is the fraction of the pollutant to be emitted. For example, $\eta_i = 100\ \%$ means no abatement effort is exerted, while $\eta_i = 0$ implies perfect cleanup of the pollutant (zero pollution). Thus we can view η_i as the decision variable of sector i. Given η_i, the total SO_2 emissions from sector i can be written as $E_i = W_i\, g_i\, \eta_i$.

There is a cost associated with the SO_2 abatement effort, and the cost structure varies across firms and industries. The first step in our analysis is to understand the cost function for pollution abatement. Ideally, one would like to treat each individual firm in the economy as an independent decision maker; however, this requires deriving the abatement cost function for each of the firms. This is very challenging due to the vast number of heterogeneous firms in the economy. Therefore, as a practical approximation, we will consider each industry sector as an independent decision maker. This approximation is based on the implicit assumption that firms within the same industry possess similar abatement cost structures. Such an assumption is reasonable to the extent that both the production and abatement technologies present greater similarity in the same industry than across different industries. In addition, this paper presents the industry-level analysis as an illustration; the same method applies to more refined levels when data availability is not a constraint.

We follow the literature to adopt the following format for pollution abatement cost function. Since the mid-1990s, researchers from China and the World Bank have initiated a series of studies on the factors that may affect pollution abatement costs. Using large plant-level databases provided by China's National Environmental Protection Agency (now Ministry of Environmental Protection), they show that the end-of-pipe abatement costs for major air pollutants, including SO_2, mainly depend on the waste gas volume, effluent/influent ratio (which can be interpreted as either concentration ratio or volume ratio since the gas volume is constant across influent and effluent), and characteristics specific to plants or industrial sectors. They find that constant elasticity cost function generally fits the data well (Hettige et al., 1995; Wang and Wheeler, 1996, and Dasgupta et al., 1997). The proposed functional format has been accepted and successfully used in a few follow-up studies with some variation (see, e.g., Du et al., 2007 and Cao et al., 2009). Following their functional format, the SO_2 abatement cost for industry i is given by:

$$C_i = e^{\alpha_i} W_i^{\beta_i} \eta_i^{\gamma_i}, i = 1, \ldots, n,$$

where α_i, β_i, and γ_i are the parameters that are specific to industry i. The abatement cost C_i should decrease in η_i because a larger η_i means less pollutant to abate. Thus, there must be $\gamma_i < 0$. As η_i goes to 0, the abatement cost C_i approaches infinity. This is because as the pollutant concentration in the carrier becomes lower, the marginal cost of abatement becomes higher, and in reality it is infeasible to remove every pollutant atom or molecule from the carrier. Thus, the emission rate η_i can never reach 0. When η_i takes the value of 100% (i.e., all pollutant is emitted without any abatement effort), the abatement cost is a positive constant $e^{\alpha_i} W_i^{\beta_i}$, which reflects the spread fixed costs such as the depreciation costs of pollution abatement facilities and the relevant overhead costs. Notably, the format of the abatement cost function is derived based on empirical research. Because only yearly data were available, the estimation was conducted on an annual basis (Dasgupta et al., 1997). That is, W_i is the annual pollutant carrier and C_i is the annual cost for industry i.

Suppose the regulator sets an annual emission target E. The target could be set by taking a wide range of economic, social, and political factors into account, and we treat it as exogenously given in this paper. To ensure that the emission cap is nontrivial, we assume E is less than the total emission when all industries' emission rates are 100%, i.e., $\sum_{i=1}^{n} W_i g_i > E$. In the rest of the paper, we consider two market-based emission control mechanisms to achieve the environmental target: The first is the cap and trade system, where the regulator allocates tradable quota/permit to the industries for free; the second is the pollution levy system, where the regulator sets a levy rate for emissions. In both mechanisms, the objective is to minimize the overall abatement cost for all industries while ensuring the overall emission does not exceed the target E. To facilitate analysis, we assume that the pollution abatement cost function is deterministic and common information to all parties;

[4] Pollutant carrier refers to the substance in which the pollutant resides; e.g., the pollutant carrier is air for SO_2.

further, we ignore the transaction and policy implementation costs. With such a setup, it is well-known from the literature that the equilibrium permit trading price in the cap and trade mechanism, say p, will be equal to the optimal levy rate the regulator should set, say t. Next, we present the problem formulations for the two mechanisms separately. Based on these formulations, later we will apply data to derive the equilibrium market outcomes, especially the values for p and t.

2.1. Cap and trade

First we consider the cap and trade mechanism. Without losing generality, suppose the regulator allocates a proportion λ_i of the total quota E to industry i, $i = 1, 2, \cdots, n$, $\sum_{i=1}^{n} \lambda_i = 1$. Each industry tries to minimize its cost by deciding the amount of pollutant to abate, or, equivalently, the amount to emit. Excess quota will be sold in the market at the equilibrium trading price p. According to the existing literature, we know the equilibrium trading price should be equal to the marginal abatement cost. In an economy with multiple industry sectors, the marginal abatement cost is not readily available for a given emission target. Thus, we go through the following analysis to derive the equilibrium trading price p.

Given p, each industry sector solves the optimal emission rate as follows:

$$\begin{aligned} &\min_{\eta_i} Y_i = e^{\alpha_i} W_i^{\beta_i} \eta_i^{\gamma_i} + (W_i g_i \eta_i - \lambda_i E) p, \; i = 1, \cdots, n \\ &\text{s.t.} 0 \le \eta_i \le 1, \end{aligned} \tag{1}$$

where the first term in the objective function is the abatement cost for achieving the emission rate η_i, and the second term is the payment for purchasing extra emission quota (if it is negative, then the industry sector earns money by selling the excess quota in the market). By $\gamma_i < 0$, the objective function Y_i is concave in η_i. Let μ_i be the dual variable of the constraint $\eta_i \le 1$ and ν_i the dual variable of the constraint $0 \le \eta_i$. One set of the complementary slackness conditions is $\nu_i \eta_i = 0$. By $\gamma_i < 0$, $\eta_i = 0$ must be suboptimal, and hence $\nu_i = 0$. Then, the following Karush–Kuhn–Tucker (KKT) conditions characterize the industries' optimal decisions:

$$\begin{aligned} 0 \le \eta_i \le 1, \;\; & i = 1, \cdots, n \\ \mu_i \ge 0, \;\; & i = 1, \cdots, n \\ \mu_i (\eta_i - 1) = 0, \;\; & i = 1, \cdots, n \\ \gamma_i e^{\alpha_i} W_i^{\beta_i} \eta_i^{\gamma_i - 1} + W_i g_i p + \mu_i = 0, \;\; & i = 1, \cdots, n. \end{aligned} \tag{2}$$

Proposition 1. *Under the cap and trade system, industry i's optimal emission rate satisfies* $\eta_i^* = 1$ *if* $p \le \frac{-\gamma_i e^{\alpha_i} W_i^{\beta_i - 1}}{g_i}$, *and* $\frac{\partial C_i}{\partial \eta_i} = -W_i g_i p$ *otherwise.*

The proof is given in Appendix A. Recall $C_i = e^{\alpha_i} W_i^{\beta_i} \eta_i^{\gamma_i}$. The condition $p \le \frac{-\gamma_i e^{\alpha_i} W_i^{\beta_i - 1}}{g_i}$ is essentially $p \le \left(\frac{-1}{W_i g_i}\right) \frac{\partial C_i}{\partial \eta_i}$ at $\eta_i = 1$, i.e., the market price of pollution is lower than the lowest possible marginal abatement cost. Therefore, it is optimal to emit all pollutant without any abatement effort when this condition holds. If $p > \frac{-\gamma_i e^{\alpha_i} W_i^{\beta_i - 1}}{g_i}$, the trading price is high, and it is optimal for industry i to abate some pollutant.

Proposition 1 characterizes each industry's optimal emission rate for any given equilibrium price p. To determine p, we identify an important equilibrium condition as follows.

Proposition 2. *Under the cap and trade system, in equilibrium the total emission from all industries equals the total allocated quota E, i.e.,*

$$\sum_{i=1}^{n} W_i g_i \eta_i = E. \tag{3}$$

Proposition 2 implies there will be no leftover quota in equilibrium. The equilibrium price p and the industries' optimal emission rates η_i, $i = 1, \cdots, n$, can be derived by Propositions 1 and 2, i.e., jointly solving the system of equalities and inequalities in Eqs. (2) and (3).

It is worth noting that a larger quota E allows more emission, which leads to a lower equilibrium trading price p. Moreover, since the equalities and inequalities in Eqs. (2) and (3) are independent of λ_i, the trading price as well as the emission behavior in equilibrium are also independent of the initial quota allocation. This is a useful property because it indicates that any initial quota allocation will lead to the desired equilibrium outcome. However, the initial allocation determines the distribution of cost burden among the industries, which requires careful consideration. More discussions of this issue can be found in Ekins and Barker (2001), Ellerman (2002), and Metcalf (2009), to name a few examples.

2.2. Pollution levy

Under the pollution levy system, the regulator sets a levy rate to meet the emission target E while minimizing the total pollution abatement costs across all industries. Given the levy rate t, each industry then chooses its optimal emission rate η_i. The regulator's and the industries' decisions can be formulated as the following two-stage optimization problem:

$$\begin{aligned} &\text{Stage 1(regulator)}: \; \min_t C = \sum_{i=1}^{n} C_i = \sum_{i=1}^{n} e^{\alpha_i} W_i^{\beta_i} \eta_i^{\gamma_i} \\ &\qquad \text{s.t.} \sum_{i=1}^{n} W_i g_i \eta_i \le E \\ &\text{Stage 2(industry)}: \; \min_{\eta_i} \left[e^{\alpha_i} W_i^{\beta_i} \eta_i^{\gamma_i} + W_i g_i \eta_i t \right] \qquad i = 1, \cdots, n \\ &\qquad \text{s.t.} 0 \le \eta_i \le 1. \end{aligned} \tag{4}$$

In stage 2, given the levy rate t set by the regulator, the industries choose emission rates to minimize their own costs. It can be shown that each industry's optimal emission rate is exactly the same as that in Proposition 1 if we replace p with t. From Proposition 1, we find that η_i decreases in t, i.e., a higher levy rate leads to more pollution abatement and thus less emission. In stage 1, anticipating the industries' optimal emission rates, the regulator chooses t to minimize the total abatement costs while satisfying the emission target E. Clearly, a higher t leads to more pollution abatement, so the total abatement costs increase in t. Thus, on one hand, in order to minimize the total abatement costs, the regulator should set the lowest possible levy rate. On the other hand, a lower levy rate leads to a larger amount of emission. Hence the best option for the regulator is to choose t such that the total emission equals the quota E, i.e., $\sum_{i=1}^{n} W_i g_i \eta_i = E$. This equation, together with each industry's optimal emission decision, again leads to the system of equalities and inequalities of Eqs. (2) and (3). This implies that the cap and trade and the pollution levy mechanisms are essentially equivalent in terms of the equilibrium outcome. Despite the identical outcome, there remains one important distinction between the two mechanisms: Each industry bears a heavier financial burden in the pollution levy system (recall in the cap and trade system the emission permits are distributed to the industries for free).

The above analysis shows that when the abatement cost structures are public information, the regulator can choose a levy rate to achieve the same optimal outcome as in the cap and trade mechanism. In view of the current market condition and the supporting legal system in China, pollution levy is still a primary instrument to control SO_2 emissions. Thus, it is crucial to derive the efficient levy rate from the regulator's perspective. To this end, we may take the following steps: First, we use historical SO_2 emissions and abatement cost data to calibrate the parameters for each industry's abatement cost function; second, based on the recovered abatement cost functions, we derive the optimal levy rate to use for the regulator; and finally, we obtain a better understanding of the problem by comparing the optimal emission behavior derived from our model with actual SO_2 emissions.

3. Analysis

3.1. Data

First we provide a description of the data we use for the analysis in this paper. According to China's industrial classification system, three industry sectors contribute the most to SO_2 emissions: Electricity, Gas, and Water Supply (EGWS); Manufacturing; and Mining. These three sectors account for 90%–92% of the total industrial SO_2 emissions each year from 2006 to 2010. For notational convenience, later we will use subscripts e, ma, and mi to denote these three sectors, respectively. The industrial classification system further divides each sector into a number of divisions. The EGWS sector has 3 divisions and the Mining sector contains 6 divisions. The Manufacturing sector consists of 30 divisions, among which two divisions (manufacturing of non-metallic mineral products and smelting and pressing of ferrous metals) are the main sources of SO_2 emissions. These two divisions are also the most energy-consuming within the Manufacturing sector. We have collected the data at the division level on a yearly basis.

Four categories of data have been used for each division in our analysis (all on an annual basis): the volume of SO_2 carrier (W_i), the SO_2 concentration (g_i), the SO_2 emission rate (η_i), and the incurred SO_2 abatement cost (C_i). The SO_2 abatement cost (C_i) refers to the total operating costs for abating SO_2 emissions, including utilities, facility maintenance and depreciation, materials, labor, and overhead costs. Our data set spans from 2001 to 2010. Part of the data are collected from the *China Statistical Yearbook on Environment*. However, this source does not provide a complete data record needed for analysis, so we collect the rest and majority of the data via the Ministry of Environmental Protection of China (MEP). Following the environmental data compilation rules set by MEP, we obtain the data for analysis by aggregating the collected information. Note that the data used in this study cover only Mainland China, not including Hong Kong, Macao, and Taiwan. We have adjusted all costs using the China Producer Price Index (PPI) from the *China Statistical Yearbook* to account for inflation; all costs are presented in 2010 RMB after adjustment. A summary of the aggregate data at the sectoral level is given in Table 1. It is worth noting that the SO_2 abatement cost in 2008 is 5.8 times that of its counterpart in 2007 and 4.2 times that of its counterpart in 2009. Therefore, the data of 2008 are outliers and thus will be winsorized in the regression analysis. To ensure sufficient data points for the regression analysis, we use division level data as detailed below.

3.2. Cost function estimation

As introduced in Section 2, each industry's pollution abatement cost takes the following functional format:

$$C_i = e^{\alpha_i} W_i^{\beta_i} \eta_i^{\gamma_i}, i = 1, \cdots, n.$$

Taking the logarithm of both sides gives

$$\ln C_i = \alpha_i + \beta_i \ln W_i + \gamma_i \ln \eta_i, \ \ i = 1, \cdots, n.$$

We estimate the parameters α_i β_i, and γ_i for each industry using the least squares linear regression. After winsorizing the data of 2008, we have 9 years of data. The Manufacturing sector consists of 30 divisions, among which the emission rate data for the division of waste and materials recovery is missing. Thus, there are $29 \times 9 = 261$ data points used in the regression for the Manufacturing sector. The EGWS sector consists of 3 divisions and hence $3 \times 9 = 27$ data points are used in the regression. The Mining sector consists of 6 divisions, among which the cost structure of the crude petroleum and natural gas extraction (CPNGE) division exhibits a different pattern from those of the other five divisions: coal mining, ferrous metal mining, non-ferrous metal mining, non-metal mining, and other minerals mining. This is understandable considering that the CPNGE division involves liquid and gas mining whereas the other five divisions involve solid mining. Due to its different cost structure, we remove the CPNGE division and only use the data of the other five divisions in our analysis (these five divisions account for 86% of SO_2 generation in the Mining sector); so there are $5 \times 9 = 45$ data points for the regression. The regression results are presented in Table 2. We can see that all the estimated parameters are statistically significant. The coefficient β (of the pollutant carrier volume W_i) is positive, while the coefficient γ (of the emission rate η_i) is negative for all three industry sectors. This is consistent with the intuition that the total abatement cost increases in the volume of the carrier but decreases in the emission rate.

Compared to the Manufacturing sector, the EGWS and Mining sectors have lower α values. This indicates that these two sectors incur a lower fixed cost for SO_2 abatement. A plausible explanation is as follows. The pollution abatement cost function we use is separable from the production cost function, reflecting end-of-pipe activities. The end-of-pipe pollution abatement has significant scale economies (Dasgupta et al., 1997). The EGWS sector, for instance, typically consists of very big companies with large facilities, which allows spreading fixed abatement costs across a large amount of SO_2 emission. In contrast, firms in the Manufacturing sector are quite heterogeneous in scale. There are many plants with small- or medium-sized facilities where scale economies are less representative. Thus, the α value for the Manufacturing sector is larger.

The parameter γ captures the sensitivity of the SO_2 abatement cost to the emission rate. The EGWS sector has the smallest value of γ, which implies that its SO_2 abatement cost is the most sensitive to the emission rate. That is, as the emission rate increases, the abatement cost for the EGWS sector decreases faster compared to the other two sectors. The Manufacturing sector has the largest value of γ, which means its abatement cost is the least sensitive to the emission rate. This is because the initial SO_2 concentration rate for the EGWS sector (g) is more than two times that of the Manufacturing sector. Later we

Table 1
SO_2 emission and abatement cost data for three major industry sectors in China from 2001 to 2010 (the units for W, g, and C are billion m^3, gram per m^3, and million RMB, respectively).

Year	EGWS				Manufacturing				Mining			
	W	g	η	C	W	g	η	C	W	g	η	C
2001	5412.4	1.504	0.895	2645	10,114.3	1.005	0.556	10,542	403.0	1.33	0.670	490
2002	5879.6	1.447	0.885	3241	11,103.4	1.017	0.499	18,014	405.1	1.497	0.616	583
2003	6855.4	1.42	0.889	6600	12,430.5	0.953	0.485	17,682	437.5	1.259	0.563	569
2004	8023.7	1.461	0.851	4722	15,113.7	0.915	0.502	20,765	526.1	1.083	0.586	1197
2005	8918.9	1.605	0.817	6736	17,113.1	0.895	0.485	22,611	572.5	1.354	0.530	571
2006	10,233.4	1.598	0.738	14,791	21,762.6	0.776	0.446	35,016	638.3	1.007	0.599	840
2007	12,585.1	1.580	0.578	21,007	25,432.7	0.718	0.419	36,521	655.0	1.399	0.557	820
2008	12,253.7	1.748	0.496	29,113	27,370.5	0.684	0.388	212,212	702.9	1.493	0.431	992
2009	14,439.5	1.644	0.394	39,640	28,533.9	0.732	0.343	51,028	612.7	1.948	0.343	1325
2010	18,338.5	1.552	0.317	42,922	32,815.4	0.623	0.370	59,153	718.2	1.553	0.361	1287

Table 2
Coefficient estimation for pollution abatement cost functions (the units for W and C used in the regression are 10^8 m^3 and 10^4 RMB, respectively).

Industry	α	β	γ	R^2
EGWS	2.0763***	0.9447***	−1.8836**	0.96
Manufacturing	3.3037***	0.9014***	−0.3930***	0.84
Mining	2.0771***	1.0333***	−0.6232*	0.86

*** P-value < 0.001.
** P-value < 0.01.
* P-value < 0.05.

will discuss how these features will affect the optimal emission structure that can minimize total abatement costs for the economy.

3.3. Results

Based on the abatement cost functions from Section 3.2, in this subsection we apply the data to derive the equilibrium outcome under the two emission control mechanisms. Recall from Section 2 that the optimal levy rate t^* in the levy system should be equal to the equilibrium permit trading price p^* under the cap and trade system. As a result, the emission rate for each industry (η_i) and the total abatement cost (C) are also the same under the two mechanisms. Using the models presented in Sections 2.1 and 2.2, now we are ready to derive the values of t^* (or p^*), η_i^*, and C^* for each year. Then we will compare these model-predicted results with the actual data to draw policy implications. For comparability, the actual annual emissions are taken as the emission targets E in our computation.

Table 3 summarizes the results. For ease of illustration, we focus on the time window 2006–2010 (the results are similar for years 2001–2005). The first row of the table is the actual annual SO_2 emissions from all three industry sectors, taken as the emission target E. The second row presents the total SO_2 generated by the three sectors. The difference between the first two rows is then the abated SO_2 emissions. The third row lists the actual total abatement costs C incurred by the industries to achieve the emission target E. The fourth row derives the optimal levy rate or the equilibrium permit trading price using the models in Section 2. The fifth row gives the optimal total abatement cost (denoted C^*) predicted by our model, assuming the same emission target given in the first row. The last row shows the percentage of cost savings if the optimal emission price t^* (or p^*) is used.

Despite the fact that the annual generated SO_2 has been increasing rapidly during 2006–2010 (second row of Table 3), the total emission E decreases gradually (first row of Table 3), which is consistent with the regulator's goal to reduce SO_2 emissions. To achieve these tighter emission targets under a growing amount of generated SO_2, not surprisingly, the optimal levy rate increases significantly over time. For example, although the generated SO_2 increased by 48% from 2006 to 2010 (from 33.88 to 50.00 million tons), the emission target decreased by 15% (from 19.99 to 16.99 million tons), and the optimal levy rate increased by 2.97 times (from 1.24 to 4.92 RMB/kg).

We emphasize that there is a gap between the optimal levy rate t^* in our model and the actual levy rate applied in practice. Take the year 2007 for example. The optimal levy rate is 1.82 RMB/kg in 2007 according to our model, whereas the actual rate is 0.63 RMB/kg in some provinces (e.g., Guangdong province) and 1.26 RMB/kg in the others (e.g., Jiangsu province). More importantly, the actual levy rate has been kept at 1.26 RMB/kg since 2007. We can see that the actual adopted rates in China are significantly lower compared to the optimal levy rate of 4.92 RMB/kg in 2010. There are two possible reasons behind the underestimated levy rate used in practice: First, there is a lack of understanding of the optimal levy rate the government should use; second, the optimal levy rate represents a significant financial burden to many firms and therefore may hinder economic growth considering that a lot of Chinese manufacturing firms are operating at low margins. Nevertheless, our finding implies that in order for the levy system to play a greater role in SO_2 emission control, the regulator should adjust the levy rate more aggressively over time in order to catch up with the new emission targets while facing a fast-growing amount of generated SO_2. The optimal levy rate derived above provides a useful reference for the regulator to decide how fast the levy rate should increase over time.

Given the actual low levy rate, the industries have little incentive to abate emissions. Hence, to achieve the emission target of 19.65 million tons in 2007, there must be other instruments put in effect besides the pollution levy system, whether it is command and control or similar administrative and political instruments. A detailed discussion of such instruments can be found in Schreifels et al. (2012). Clearly, these administrative instruments must have played an important supplemental role in achieving the emission target. To help understand the impact of these administrative instruments, we derive each sector's hypothetical optimal emission rate at the levy rate 1.26 RMB/kg, assuming there is no regulation at all. These hypothetical emission rates are denoted $\eta_i^{'}$ ($i = e, ma$, and mi) and reported in Table 4. For easy comparison, we also list the actual emission rates η_i^a of each sector in the same table.

It can be seen that the hypothetical emission rates $\eta_i^{'}$ are greater than the actual rates η_i^a (the only exception is for the Mining sector in 2007). This implies that regulation plays a meaningful role in all the sectors. Further, the difference between the two rates is much larger for the EGWS sector than for the Manufacturing sector. For example, in 2010 $\eta_e^{'} - \eta_e^a = 40.6$ % whereas $\eta_{ma}^{'} - \eta_{ma}^a = 13.8$ %. That is, regulation plays an even greater role for the EGWS sector than for the Manufacturing sector. This is because compared to the Manufacturing sector, the EGWS sector consists of much fewer but larger firms. Therefore, considering the fixed administrative costs, command and control will be more effective for the EGWS sector than for the Manufacturing sector.

However, without a good understanding of the different industries' characteristics, administrative approaches may fail to induce the most cost-effective emission structure, i.e., the imposed emission constraint for each industry may not match the system-optimal solution. This is why the actual total abatement cost is consistently higher than the optimal level predicted by our model: In 2007, the optimal emission structure leads to a total abatement cost 25.1% lower than the actual abatement cost. For 2006–2010, the optimal emission structure saves 49.7% of the abatement costs in total. The above analysis indicates that the higher-than-predicted abatement cost is due to a sub-optimal emission structure. So we proceed to look into each industry sector's SO_2 emission behavior. The total emission (E_i^a) of each sector is also available in our data set. The optimal emission rate (η_i^*) and amount (E_i^*) are

Table 3
SO_2 emission price and abatement cost: model-predicated results vs. actual outcome.

	2006	2007	2008	2009	2010	Total
E(million ton)	19.99	19.65	18.35	16.93	16.99	91.91
$\sum_{i=1}^{3} W_i g_i$(million ton)	33.88	39.05	41.20	45.82	50.00	209.95
C (million RMB)	50,647	58,348	242,317	91,993	103,363	546,667
t^* or p^* (RMB/ton)	1238	1816	2310	3410	4919	–
C^* (million RMB)	32,764	43,712	50,581	63,781	84,366	275,204
$\frac{C^*-C}{C}$ (%)	−35.3	−25.1	−79.1	−30.7	−18.4	−49.7

Table 4
Hypothetical optimal emission rates $\eta_i^{'}$ versus the actual emission rates η_i^a(%).

	EGWS		Manufacturing		Mining	
Year	η_e^a	$\eta_e^{'}$	η_{ma}^a	$\eta_{ma}^{'}$	η_{mi}^a	$\eta_{mi}^{'}$
2007	57.8	72.4	41.9	46.7	55.7	54.9
2008	49.6	69.9	38.8	48.1	43.1	52.8
2009	39.4	71.2	34.3	45.7	34.3	44.7
2010	31.7	72.3	37.0	50.8	36.1	51.6

derived based on our models in Section 2. Table 5 summarizes these values for all three industry sectors.

The Mining sector's emission amount is very small compared to that of the other two sectors. Consequently, the abatement cost accounts for slightly more than 1% of total abatement costs. Hence the abatement cost savings should come mainly from adjusting emission structure for the EGWS and Manufacturing sectors. Table 5 shows that except for 2006, the emission rate of the Manufacturing sector should decrease whereas that of the EGWS sector should increase. This is because the γ parameter in the abatement cost function for the EGWS sector is very small, which means that a larger emission rate will significantly reduce the abatement cost, or alternatively, the abatement cost is very sensitive to the pollutant abating amount. In contrast, the γ parameter for the Manufacturing sector is much larger, which means emitting more does not save much abatement cost. Therefore, it is more cost-effective for the Manufacturing sector to abate more than the EGWS sector.

In order to achieve the optimal emission structure, a natural solution is to increase the levy rate to the optimal level. However, this approach will dramatically increase the financial burden on all industries and lead to undesired economic consequences. It is necessary for policy makers to understand the added financial burden on different industries if the levy rate is raised to the most efficient level. Let $T_i^* = W_i g_i \eta_i^* t^*$ be sector i's levy payment for emissions and $T^* = \sum_{i=1} W_i g_i \eta_i^* t^*$ be the total levy payment of all three sectors, both under the optimal levy rate t^*. Table 6 compares these levy payments with the corresponding abatement costs from 2006 to 2010.

Since the optimal levy rate t^* increases significantly over time, payments for emissions T_i^* and T^* increase regardless of decreasing optimal emission rates. Table 6 shows the total payment of all industries increases by more than two times (from 24,746 in 2006 to 83,559 in 2010). The added financial burden ranges from 76% to 99% of the total abatement cost for 2006–2010. Compared to the Manufacturing sector, the EGWS sector's optimal emission rates are much higher and hence its T^* is also much higher. As to each industry's incurred total cost, i.e., the sum of emission payment and abatement cost ($T_i^* + C_i^*$), the Manufacturing sector incurs the largest cost from 2006 to 2008, whereas for 2009 and 2010 the EGWS sector incurs the largest total cost. This is because the EGWS sector's annual volume of SO_2 carrier (W) increased by 17.8% from 2008 to 2009 (the fastest compared to 4.3% for the Manufacturing sector and −12.8% for the Mining sector), and furthermore, the initial SO_2 concentration for the EGWS sector (g) is more than two times that of the Manufacturing sector. With such a high growth rate of SO_2 generation, the EGWS sector incurs the largest total cost to deal with SO_2 in 2009–2010.

From the above analysis we find that the current levy rate in China is too low to induce the most efficient emission structure for the economy. To achieve the target emission level, the regulator has to either use an administrative approach or raise the levy rate to the optimal level. However, both of these solutions have their own disadvantages. An administrative approach such as command and control may impose implementation challenges or inefficient emission structures (as manifested in Tables 3 and 5). The optimal levy rate, t^*, may impose too much of a financial burden on the industries. A refund measure might be needed if the levy rate is further increased. Theoretically, a hybrid system that combines the pollution levy system with tradable permits can successfully solve the problem. Specifically, each industry needs a permit for emission and for each unit of emission it needs to pay a levy rate $\hat{t}$. Suppose in this hybrid system, the equilibrium permit trading price is $\hat{p}$. Then each industry's optimal problem can be expressed as follows:

$$\begin{aligned}&\min_{\eta_i} Y_i = e^{\alpha_i} W_i^{\beta_i} \eta_i^{\gamma_i} + W_i g_i \eta_i \hat{t} + (W_i g_i \eta_i - \lambda_i E)\hat{p}, i = 1, \cdots, n \\ &\text{s.t.} 0 \le \eta_i \le 1.\end{aligned} \tag{5}$$

Note that this problem slightly differs from Problem (1) by the term $W_i g_i \eta_i \hat{t}$, but the condition $\sum_{i=1}^n W_i g_i \eta_i = E$ still holds in equilibrium. Then, substituting $p = \hat{t} + \hat{p}$, η_i and $\hat{p}$ can again be solved by the system of equalities and inequalities in Eqs. (2) and (3). The resulting values of η_i are identical to those under either the pure pollution levy system or the pure cap and trade system. Therefore, in the hybrid system, the optimal emission structure will be induced, which also leads to the most efficient abatement costs for the economy. The only difference from the pure pollution levy is that each industry's payment is reduced from $W_i g_i \eta_i^* t^* = W_i g_i \eta_i^* (\hat{t} + \hat{p})$ to $W_i g_i \eta_i^* \hat{t}$. By changing the value of $\hat{t}$, the regulator can arbitrarily adjust the financial burden imposed on the firms in each industry.

Although the hybrid system sounds attractive from a theoretical perspective, as a caveat, it requires the existence of a well-functioning emission trading market. While China slowly establishes its emission trading system, an acceptable interim solution is to charge a relatively low levy rate (to avoid overburdening firms) and at the same time use administrative instruments to guide firms' emission behavior. In this case, it is critical for the regulator to obtain a good understanding of the optimal emission price and the most efficient emission structure. This paper provides a method to derive the equilibrium emission price and the resulting emission outcome, which may serve as a useful input in the policy-making process.

3.4. Discussion

In the above analysis, we consider three industry sectors that represent major contributing sources of SO_2 emissions in the Chinese economy. By using the data collected for these sectors, we illustrate a method to analyze the market-based emission control mechanisms. It is worth emphasizing that such a method can be applied to many other problem situations. First, the analysis can be conducted at more

Table 5
SO_2 emission rate (%) and total emissions (million tons) by industry sector.

	EGWS				Manufacturing				Mining			
Year	η_e^a	η_e^*	E_e^a	E_e^*	η_{ma}^a	η_{ma}^*	E_{ma}^a	E_{ma}^*	η_{mi}^a	η_{mi}^*	E_{mi}^a	E_{mi}^*
2006	73.8	72.8	12.07	11.91	44.6	45.2	7.54	7.64	59.9	67.9	0.39	0.44
2007	57.8	63.8	11.50	12.69	41.9	36.0	7.64	6.56	55.7	43.8	0.51	0.40
2008	49.6	56.7	10.63	12.15	38.8	31.1	7.27	5.83	43.1	36.4	0.45	0.38
2009	39.4	50.4	9.35	11.96	34.3	22.4	7.17	4.68	34.3	24.2	0.41	0.29
2010	31.7	45.1	9.02	12.83	37.0	19.1	7.56	3.90	36.1	22.3	0.40	0.25

Table 6
Levy payment (T_i^*) vs. abatement cost (C_i^*) (million RMB) by industry sector.

Year	EGWS			Manufacturing			Mining			Total	
	T_e^*	$\frac{T_e^*}{C_e^*}$	$T_e^* + C_e^*$	T_{ma}^*	$\frac{T_{ma}^*}{C_{ma}^*}$	$T_{ma}^* + C_{ma}^*$	T_{mi}^*	$\frac{T_{mi}^*}{C_{mi}^*}$	$T_{mi}^* + C_{mi}^*$	T^*	$\frac{T^*}{C^*}$
2006	14,746	1.88	22,574	9459	0.39	33,527	541	0.62	1408	24,746	0.76
2007	23,028	1.88	35,254	11,915	0.39	42,230	730	0.62	1900	35,672	0.82
2008	28,049	1.88	42,940	13,471	0.39	47,746	882	0.62	2296	42,401	0.84
2009	40,822	1.88	62,494	15,928	0.39	56,455	986	0.63	2567	57,735	0.91
2010	63,120	1.88	96,631	19,216	0.39	68,108	1223	0.62	3186	83,559	0.99

refined industry levels. In the 12th FYP (2011–2015), the Chinese government for the first time sets up pollution reduction targets for several specific industrial sectors, including thermal power generation, iron and steel, cement, paper making, and so on (Xue et al., 2014). How can the pollution reduction target be set so that the most cost-effective emission structure can be achieved? Our model and analysis may help answer this question for policy makers. We only need to replace the polluters in our analysis by these specific industry sectors, and plug in the corresponding data regarding emission targets, abatement cost functions, etc. Although this may require data at more refined industry levels and involve larger-scale optimization, the essence of the analysis will not change.

Second, the method could be applied to the regional, rather than national level. In China, the central government is primarily responsible for enacting policies, while local governments are held accountable for implementing pollution control in their own geographical regions. Consider the pollution levy rate. A local government (for example, a provincial government) has the flexibility to choose its levy rate from a range under the guideline devised by the central government. Regional disparity and the associated cost differences imply the optimal levy rate should vary across regions. When a provincial government determines its optimal levy rate, it needs to take the industries within its own province as the subject of study. By using the province-level data, an analysis similar to ours can help a local government derive the optimal levy rate.[5]

As part of the effort to build a national emission trading market, certain provincial governments with large economy size and sufficient administrative capacity may take the initiative to build local markets first. An understanding of the province's cost-effective emission structure and each industry's emission need is helpful in devising an initial quota allocation that can minimize transaction costs. In practice, it takes time for the market to reach the equilibrium trading price. The trading price derived from the analysis can be sent as a signal by the government to the public in order to speed up the process of reaching the equilibrium.

Third, the same method may also be used to analyze pollution control problems for pollutants other than SO_2. An immediate application would be for CO_2 emission control. In addition, the 12th FYP (2011–2015) of China includes control targets for pollutants such as chemical oxygen demand, ammonium nitrate, and nitrogen oxides. Our analysis can be applied to help regulators design policies to control these pollutants as well.

Finally, we can see that data is the foundation underlying the analysis in this paper. More detailed and complete data will provide more accurate prediction of pollution abatement and emission activities, which serves as a more useful reference in policy making. Data collection is a direction that deserves more effort and attention.

Making more detailed data publicly available, such as data at the provincial level, will no doubt promote quantitative environmental studies, which in due course will enhance scientific-based policy making. To this end, the governments both at the central and local levels shall exert effort to monitor and collect data about all industries' emission volumes, abatement costs, emission rates, and other useful information.

4. Conclusion

Two market-based mechanisms have been widely studied in the literature to control environmental pollution: cap and trade and pollution levy. Although it has been shown theoretically that both mechanisms can yield the most efficient emission structure, there are certain challenges for implementation. In particular, the regulator generally does not know the relationship between the emission target and the associated optimal emission price. This paper proposes a method for analyzing these market-based pollution control mechanisms. The method represents an integrated approach that consists of two steps. In the first step, historical data on emissions and abatement costs from polluters are used to estimate the parameters in the abatement cost functions. Then, in the second step, the optimal emission price can be derived by solving the equilibrium in the market model formulated for each mechanism.

We apply the method to analyze the SO_2 emission control policies in China, which is the world's fastest-growing economy with escalating environmental pollution. As an illustration, we study the optimal emission price for three industry sectors (EGWS, Manufacturing, and Mining) that represent the major sources of SO_2 emissions in China. The time window for analysis is from 2001 to 2010. We find that, given the emission targets, the actual levy rates in practice are much lower compared to the optimal level. As a result, the actual emission structure leads to excessive abatement costs for the economy. Based on our analysis, the overall cost savings could be as high as 49.7% for all industries during 2006–2010 if the optimal emission structure can be achieved. For instance, from 2007 the most efficient structure allows the EGWS sector to emit more while requiring the Manufacturing sector to reduce its emissions. Clearly, these findings are helpful for regulators to devise effective policies that can lead to efficient market outcomes.

The proposed method may have a wide range of applications. In this paper, we consider three industry sectors at the national level. Similar analyses can be conducted at regional levels to help local governments in policy making. This is especially true in China because local governments are directly responsible for pollution control, and many provinces are large enough to implement their own control policies. Further, the method is applicable to other types of emissions and pollutants such as CO_2, chemical oxygen demand, and nitrogen oxides. Lastly, the objective in this paper is to minimize total abatement costs for a given emission target. The method can be modified to consider other objectives. For example, the regulator may wish to achieve the minimum possible total emissions while facing a fixed budget for abatement. As pollution control becomes a more important issue, there is an increasing need for quantitative analysis that can provide useful decision-making support. We believe that research opportunities abound along these lines in the near future.

[5] It is worth noting that in this paper we use aggregate data and do not take into account the potential difference in abatement costs across regions. Thus an analysis at the regional level helps single out the impact of such cross-region differences. The abatement costs include the cost of pollution abatement facilities, cost of chemicals and utilities, and overhead expenses. So technology and labor related costs are the two major contributing factors to the abatement cost difference across regions. It would be interesting to quantify the impact of such regional differences on our results (e.g., the optimal levy rate). This requires additional data at the regional level and is therefore left for future research.

Appendix A. Proofs of propositions

Proof of Proposition 1. Note that $\gamma_i e^{\alpha_i} W_i^{\beta_i} \eta_i^{\gamma_i - 1} + W_i g_i p$ increases in η_i because $\gamma_i < 0$. If $p < \frac{-\gamma_i e^{\alpha_i} W_i^{\beta_i - 1}}{g_i}$, which is equivalent to $\gamma_i e^{\alpha_i} W_i^{\beta_i} \eta_i^{\gamma_i - 1} + W_i g_i p < 0$ at $\eta_i = 1$, then $\gamma_i e^{\alpha_i} W_i^{\beta_i} \eta_i^{\gamma_i - 1} + W_i g_i p < 0$ for any $\eta_i \leq 1$. Thus, from $\gamma_i e^{\alpha_i} W_i^{\beta_i} \eta_i^{\gamma_i - 1} + W_i g_i p + \mu_i = 0$, we have $\mu_i > 0$. By the complementary slackness condition $\mu_i(\eta_i - 1) = 0$, $\mu_i > 0$ implies $\eta_i^* = 1$. If $p = \frac{-\gamma_i e^{\alpha_i} W_i^{\beta_i - 1}}{g_i}$, then any $\eta_i < 1$ leads to $\gamma_i e^{\alpha_i} W_i^{\beta_i} \eta_i^{\gamma_i - 1} + W_i g_i p < 0$ and $\mu_i > 0$. In this case, $\mu_i(\eta_i - 1) = 0$ cannot be satisfied. That is, there must be $\eta_i = 1$ and $\mu_i = 0$, which satisfy all conditions in Eq. (2). If $p > \frac{-\gamma_i e^{\alpha_i} W_i^{\beta_i - 1}}{g_i}$, which is equivalent to $\gamma_i e^{\alpha_i} W_i^{\beta_i} \eta_i^{\gamma_i - 1} + W_i g_i p > 0$ at $\eta_i = 1$, then $\mu_i \geq 0$ implies $\gamma_i e^{\alpha_i} W_i^{\beta_i} \eta_i^{\gamma_i - 1} + W_i g_i p + \mu_i > 0$ at $\eta_i = 1$. Thus, $\eta_i^* < 1$ must hold in order for $\gamma_i e^{\alpha_i} W_i^{\beta_i} \eta_i^{\gamma_i - 1} + W_i g_i p + \mu_i = 0$ to hold. By $\mu_i(\eta_i - 1) = 0$, $\eta_i^* < 1$ implies $\mu_i = 0$. Thus, $\gamma_i e^{\alpha_i} W_i^{\beta_i} \eta_i^{\gamma_i - 1} + W_i g_i p = 0$, which is equivalent to $\frac{\partial C_i}{\partial \eta_i} = -W_i g_i p$. □

Proof of Proposition 2. Recall we have assumed $\sum_{i=1}^{n} W_i g_i > E$. Clearly, the total emission $\sum_{i=1}^{n} W_i g_i \eta_i$ in equilibrium cannot exceed E. We argue that the total emission cannot be less than E either. Suppose $\sum_{i=1}^{n} W_i g_i \eta_i < E$. Then some industry (say, industry j) must have unused quota whereas some other industry (say, industry k) must have incurred positive abatement cost. Thus it would be profitable for both industries if industry j sells the unused quota to industry k at a price no greater than industry k's marginal abatement cost. Such a process will continue until all the quota is depleted. Combining the above arguments, we know that there must be $\sum_{i=1}^{n} W_i g_i \eta_i = E$ in equilibrium. □

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