# CHINA POLICY JOURNAL

Vol. 1, No. 1 Fall 2018

Theme Issue on Environmental Policy Instruments in China

PSO

# CHINA POLICY JOURNAL

*Vol. 1, No. 1, Fall 2018* • *© 2018 Policy Studies Organization*


## Theme Issue on Environmental Policy Instruments in China

**Guest editors:**

Hua Wang, *Renmin University of China*
Yuan Xu, *The Chinese University of Hong Kong*

# Water-Pollutant Discharge-Fee System in China

Dajun Shen[1] and Ali Guna[2 *]


## Abstract

The market-based pollutant discharge fee has long been argued to be more cost-effective than command-and-control policies. China adopted an environmental policy instrument for dealing with its water pollution in the early 1980s. However, the serious environmental deterioration that unfolded in the following decades cast doubts on the system's effectiveness. This article evaluates the water-pollutant discharge-fee system in China from the perspective of its design, top-down implementation, effectiveness, and external and internal driving forces. It discusses its role in China's overall water-pollution control and extends the analysis to provide constructive insights into a recent major reform that converted the fees into environmental taxes.


***Keywords:*** water-pollutant discharge fee, market-based environmental policy instrument, China

# Sistema de tarifa de descarga de contaminantes del agua en China


## Resumen

Durante mucho tiempo se ha argumentado que la tarifa de descarga de contaminantes basada en el mercado es más rentable que las políticas de comando y control. China adoptó un instrumento de política ambiental para hacer frente a su contaminación del agua a principios de los años ochenta. Sin embargo, el grave deterioro ambiental que se desarrolló en las décadas siguientes arrojó dudas sobre la efectividad del sistema. Este artículo evalúa el sistema de



[1] *School of Environment and Natural Resources, Renmin University of China, Beijing*

[2] *School of Environment and Natural Resources, Renmin University of China, Beijing*

[*] *Corresponding author:* ariuna@ruc.edu.cn




doi: 10.18278/cpj.1.1.1

tarifas de descarga de contaminantes del agua en China desde la perspectiva de su diseño, implementación de arriba hacia abajo, efectividad y fuerzas impulsoras externas e internas. Discute su papel en el control general de la contaminación del agua en China y extiende el análisis para proporcionar una visión constructiva de una reforma importante reciente que convirtió los aranceles en impuestos ambientales.

***Palabras clave:*** tarifa de descarga de contaminantes del agua, instrumento de política ambiental basado en el mercado, China

# 中国污水排放收费制度

摘要

长期以来，人们一直认为基于市场的污染物排放收费标准比命令管控政策规定的标准效益更高。中国在20世纪80年代初采用了一项处理水污染的环境政策工具。然而，在随后的几十年中出现的严重环境恶化使人们对该系统的有效性产生怀疑。本文从设计、自上而下的实施、效益、内外动力等方面对我国污水排放收费制度进行了评价。本文还讨论了该制度在中国全面水污染管控方面的作用，并就近期一项将收费转化为环境税的重大改革给予了建设性意见。

关键词：水污染物排放费，基于市场的环境政策工具，中国

## Introduction

A water-pollutant discharge-fee system is defined as a basic environmental management policy and economic instrument to control water pollution and environment deterioration. As a comprehensive and independent system, it originated from industrially developed countries and is considered as a national measure of charging organizations or individuals who discharge pollutants into water bodies (Bai and Qu 2009). The water-pollutant discharge-fee system could stimulate and facilitate pollution control, and the polluters are obliged to take social responsibility for contaminating water bodies and environment.

The contamination fee is charged based on two major principles: one is based on environment quality, which entails that discharging pollutants into water bodies would be charged for a pollution discharge fee; the other is based on an environmental standard, which entails that water pollutants exceeding the national standard would be charged by the quantity and concentration of contaminants (Lv 2009).

Water-pollutant discharge-fee system was introduced in the 1980s in China, by which the government and administrative departments charged for external environmental loss by translating the loss into internal costs for the pollutant discharger (Xu and Ni 2004). The development of water-pollutant discharge-fee system is a landmark in environmental law systems and has become one of the most significant elements contributing to environmental protection in China. The system developed during the vast expansion of environmental protection on institutions, laws, and policies in the first decade after implementing the policy in the 1980s (Xiang and Wang 2003), at which time the dramatic degradation of environmental quality aroused governmental and public concerns. Thus, China turned to economic incentives to address the environmental problems and achieve equity, fairness, and efficiency. Technically, China developed a series of management systems to deal with environmental pollution, among which the discharge-fee system was the earliest and most important one. The water-pollutant discharge-fee system was set up within the discharge-fee system, thereby aiming to regulate pollution behavior and the relationship between polluters and other social parties, stimulate enterprises and polluters to take their responsibilities, reduce and control the amount of total waste, quantify the environmental cost, and maximize social welfare (Zhang 2008).

Research on the water-pollutant discharge-fee system in China has been pursued only sporadically. Because the system is defined within the broader discharge-fee system, most studies focus on analyzing the whole system and the emission trade, rather than the water-pollutant discharge-fee system specifically. The legal framework of water-pollution control is designed based on a watershed-control zone and pollution-control unit to promote firms' pollution reduction (Ma, Wang, and Wang 2013; Wu, Xu, and Ma 2015). But the system's implementation in China's provinces is disconnected due to hydrological conditions and regional administrative regulations. The political mechanism determines the central government's implementation of the system to the county level. Although the environmental reform intensity varies across space and time, the decentralized environmental targets are poor to fulfill due to economic growth is overriding (Ge and Wang 2001; Zhou and Chen 2008). In addition, simple emission-reduction targets with career-promotion opportunities for local governors has appeared too aggressive to examine detailed problems during the fee system's enforcement (Genia and William

2012). Thus, the gap in the top–bottom enforcement approach and the mechanism's disconnection provide more space and flexibility for local officials to geographically transfer pollution (Kahn, Li, and Zhao 2015). The existing empirical evidences focus primarily on the correlation between economic growth and industrial pollution (Gao, Su, and Yang 2014; Li 2015; Wang, Wu, and Yan 2008), and on finding a technical solution for the water-pollutant discharge (Hammer 1996), so that the discussion of the water-pollution discharge-fee system has limited to pollutants theoretical analysis (Liu 2009). In-depth systematic and comprehensive studies combining the characteristics of water-resource utility are insufficient and unable to provide strong support for decision making.

## 1. Design of the Water-Pollutant Discharge-Fee System

During China's transformation to a market-oriented economic structure, the country's environmental deterioration cast a shadow on its unprecedented economic growth. The environmental externality was paid by the environment and the public that was exposed to the pollution, rather than by polluters. Water pollution issues became highly sensitive for the conventional water-utilization mode in the early developing stage. Increasing contaminated water and the conflict between water supply and demand stimulated the government towards water-resource sustainability.

### *1.1. Development of the Water-Pollutant Discharge-Fee System*

The water-pollutant discharge-fee system in China was designed as an indispensable component of the broader discharge-fee system and enforced in 1982. It was approved as an independent section in the *Temporary Regulation of Pollutant Discharge Fee*, which aimed to balance the water-resource shortage and the environmental costs by using price lever. The issue of water-resource protection and sewage discharge was first officially mentioned in the *Environment Protection Report* of 1978. In accordance with the polluter pays principle (PPP), the report stressed that polluters should pay for pollutants in the wastewater they discharged. In 1979, water-pollutant discharge fee was written in the Environmental Protection Conditional Law, which stated that pollutants exceeding the national standard should be charged. The water-pollutant discharge-fee collection was systematically redefined in the *Temporary Regulation of Pollutant Discharge Fee* of 1982, including revisions of its purpose, objects, charging standard, and management method, based on two years of pilot implementation and contamination facts. It was applied through the top–bottom political mechanism at the central, provincial, prefectural, and county levels. The approval of the temporary regulation provided legal support for the fee collection of water-pollutant discharge.

Then the approval of the water pollution prevention law in 1984 was an important step. It was the first independent law defining water pollution from different perspectives and strengthening its strategic meaning and impact. Subsequently, the rules for water-fee collection were adjusted in the environment protection law in 1989 from simply charging for the total amount to a standard-exceeding charge based on pollutant quantity and concentration. Moreover, the standard was broadened by expanding the pollutant types. During the 1992 nationwide shift in economic structure, an increasing amount of enterprises contributed substantially not only to gross domestic product (GDP) growth, but also to environmental pollution. In 1993, the *Inform of Fee Collection on Wastewater Discharge* was adopted and the work on water-pollution reduction was strengthened under the pressure of water-quality deterioration. In 1996, the *Integrated Water Pollutant Discharge Standard* was approved to enhance and normalize the fee-collection work.

In 2003, the pollutant discharge-fee system was systematically revised and the *Pollutant Discharge Fee Collection and Management Regulation* was promulgated together with the pollutant discharge-fee-standard calculation method as supporting documents. It was regarded as a comprehensive reform, clearly stating that the water-pollutant fee should be calculated and charged based on the equivalent of pollutant concentration and quantity in relation to the national pollutant standard. Furthermore, the areas covered by the system's policy were enlarged from inland water bodies to sea and ocean areas through the approval of the *Ocean Environment Protection Law* in the same year. This revision and extension were marked as a breakthrough in the pollutant discharge-fee system.

For the further promotion of structural change on economic development, the National Development and Reform Commission (NDRC), the Ministry of Finance, and the Ministry of Environmental Protection (MEP) jointly promulgated the *Adjustment of Pollutant Discharge Fee Collection with Related Issues* and the *Pollutant Discharge Fee Usage Management Approach* in 2014. The water-pollutant discharge-fee standard was declared more stringent than ever on national level. These laws and regulations formed the legal framework for the water-pollutant discharge-fee system in China. With the deepening reform of the market-oriented economy, changes appeared within the system to meet the actual needs. The system's evolution, as presented in Table 1, developed in accordance with social and industrial progress, which can be demonstrated by the change of the fee amount, as discussed in Section 3.

## *1.2. Framework of Water-Pollutant Discharge System*

### 1.2.1. The Fee Standard

The fee-collection standard was enforced in 1982, by which a water-pollutant discharge-fee was collected based on enterprises' standard-exceeding concentration of pollutants. The orig-

**Table 1.** Development of the Water-Pollutant Discharge-Fee System in China

| Year | Laws and Regulations | Features |
|---|---|---|
| 1978 | Environment Protection Report | The practice of Polluters Pays Principals in China |
| 1979 | Environmental Protection Conditional Law | Standard exceeding charge based on concentration & quantity |
| 1982 | Temporary Regulation of Pollutant Discharge Fee | Provide legal support for fee collection of water pollutant discharge |
| 1984 | Water Pollution Prevention Law | First independent law in terms of water pollution control |
| 1989 | Environment Protection Law (adjusted) | Pollution charge method revise and pollutant spices enlarged |
| 1993 | The Inform of Fee Collection on Wastewater Discharge | Water-pollution reduction work was strengthened |
| 1996 | Integrated Water Pollutant Discharge Standard | Enhance and normalize the wastewater fee collection work |
| 2003 | Pollutant Discharge Fee Collection and Management Regulation | System reform |
| 2003 | Ocean Environment Protection Law | System coverage area was enlarged from inland water to sea water |
| 2014 | The Adjustment of Pollutant Discharge Fee Collection with Related Issues & Pollutant Discharge Fee Usage Management Approach | System adjustment according to actual need |

inal water-pollutant discharge-fee system did not reflect the principle that a homogenous amount of contaminants was charged equally: it only charged for the highest contaminant amount, which failed to meet the standard when the firm discharged more than one kind of pollutants in its wastewater. The fee per ton was formulated differently for each pollutant. The pollution-fee charge was defined only in the standard-exceeding amount, which in essence entailed that any pollutant discharge lying within the range of the national standard had been legally permitted. And, the penalty fare was doubled for behaviors that went beyond the temporary regulation.

The water-pollutant discharge-fee system practiced a serious attempt to address pollution problems through an empirical application suiting the boost of firms, thus the fee standard was revised in 2003. This reform was a rational innovation. The fee-collection principle was changed from one based on excessive charge to one based on the discharged equivalent of water-pollutant. Once a pollutant was discharged into a water body, a pollution fee was charged, instead of restricting the discharge to a standard. The fee standard was set at 0.7 RMB per pollutant equivalent and the penalty fare was doubled for pollutants that failed to meet the standard. Moreover, enterprises that suffered economic loss could apply for half to full pollution-fee exemption.

Current water-pollutant-fee standard is defined according to the 2014 *Adjustment of Pollutant Discharge Fee Collection and Related Issue*. It declared an increase in the water-pollutant discharge-fee standard from 0.7 RMB to 1.4 RMB per equivalent. Within each discharging point, toxic pollutants and heavy metals should be added to the calculation. Other pollutants need to be sorted according to their concentration from the highest to the lowest, whereby the total fee charge should include no more than three types of pollutants. In addition, local governments are allowed to adjust the fee standard according to regional conditions. For pollution-control areas, heavily polluted areas, and economically powerful areas, standards are allowed to be set higher than the national level. The fee level dropped from 1982 to 2003 and doubled in 2014. As the half-full pollution-fee exemption is banned in the adjustment of 2014, the water-pollutant discharge fee notably increased and sped up the external environmental cost, as well as the cost within the fee system. The obstinate concept of "if you pay more you can discharge more" should be precluded. The 11$^{th}$ five-year plan set the target at "reducing 10% of the total pollutants," thus making it necessary to monitor enterprises to limit the total pollutant discharge amount. A critical penalty on standard-exceeding discharge would assure the system's operation and water-resource sustainability, theoretically limiting wastewater discharge to the largest extent.

### 1.2.2. The Fee Structure

The evolution of the water-pollutant discharge-fee system is also reflected in the fee structure. In 1974, wastewater was defined partially by an industrial waste-discharge standard. In 1982, the temporary regulation claimed that those discharging more than two pollutant types in wastewater should be charged by the highest one. The single-factor charge principle based on a standard-exceeding amount determined the water-pollution discharge fee. Enterprises from different areas adopted the same wastewater-discharge standard, though certain factors such as difference in economic development level, pollution transfer (even within same river-basin area), and water-resource capacity and function were taken into account but would lead to different results. Any production plant's marginal cost of pollution control monotonically decreased the function of the pollution-emission quantity, so that the larger the pollution emission, the lower the marginal cost (Hou 2008).

According to the 2003 regulation, different levels of the water-pollutant discharge fee were defined as pollutant discharge fee plus pollutant standard-exceeding charge. The pollutant discharge fee was reformed to cover both the standard-meeting and standard-exceeding parts. The latter was doubled for excessive pollutant concentration. Until 2014, the block-rate pollutant structural change was designed for different situations (Xu 2014). The revision of the fee structure was partially tested in certain provinces and cities. The block-rate pollutant charge was first conducted in Tianjin. The pollutant fee increased to 7.5 RMB on COD and 9.5 RMB on ammonium and nitrogen: a tenfold increase compared to the previous standard. Which indicated that the smaller the amount of wastewater, the lower the fee. A differential charge standard was set in Tianjin in the *Inform of Block-Rate Based Pollutant Discharge Fee Standard Adjustment*, which regulated that pollutant discharge concentrations of 90%–100% should be charged according to a general standard, wastewater concentrations of 80%–90% should be charged by multiplying the discharge amount by 90%, concentrations of 70%–80% should multiply the fee by 80%, concentrations of 60%–70% should multiply the fee by 70%, and concentrations of 50%–60% should multiply the fee by 60%, while concentrations under 50% should be charged less than 50%. The different stages fit for different wastewater-discharging scales on firms' production abilities, which was quickly imitated and spread to other industrially developed provinces.

The other fee-structure improvement was conducted in ShanXi province, where the wastewater discharge fee increased to 1.4 RMB per pollutant equivalent. In addition, pollution concentration exceeded the national and provincial limits, or amounts exceeding the aggregate value would be double charged. If both of the conditions had been met, the fee would be tripled. Furthermore, the water-pollutant discharge-fee structure was set up based on different industries. Fees for the petrochemical, packaging, and printing industries were increased to 1.8 RMB per pollutant equivalent. The water-pollutant fee was graded by charging for every 10% for concentration percentages from 50% to 100%, while still halved for water-pollutant concentrations below 50%. Provinces with similar economic structures implemented this type of block-charge policy with adjustments concerning their regional natural resource capacity and developing mode.

The pollutant standard-exceeding charge is calculated for the following situations: the pollutant category is restricted by national regulations and regional laws, the total amount of water pollutant exceeds the upper limits, and the production equipment or the products are listed in the 2011 *Industrial Structure Adjustment Guide*. If the pollutant's discharge behavior falls under one of these situations, the penalty fee is charged once; while if it falls under two situations, the penalty is doubled; or meets all situations, the penalty go up to thrice. Moreover, to strengthen water-pollution control, the penalty fare is defined in terms 73 and 74 of the water pollution prevention law (2008): the irregular use of wastewater-treatment facilities and the demolition and idling of pollutant-treatment equipment should be charged by the upper level of the executive authority with a penalty fare of one to three times the water-pollutant discharge fee, while the national or regional standard-exceeding discharge over the total quantity-control indicators are charged with a fare of two to five times the discharge fee.

The flexible fee structure not only affects the industrial structure, but also guides the spacial distribution of enterprises avoiding environmental sensitive areas and gives preference to water-resource abundant and low development-difficulty areas, which has further practical implications from the perspective of development and management.

### 1.2.3. Pollutant Factors

With industrial development, the water-pollutant discharge-fee system was broadened by expanding the range of pollutant factors. The *Temporary Regulation of Pollutant Discharge Fee* set the standard for wastewater, waste gasses and solids, among which 20 types of water pollutants are clearly defined. In 1991, the list was extended to 29 types due to an increase in known facts about pollution. With further promotion of the water-pollutant discharge-fee system till 1993, the water-pollution fee was deepened from the content and the range. In 2003, the pollutant types were redefined and extended to 65, covering most of the water contaminants, of which 36 types were newly added, including heavy metals, radioactive materials, biochemical pollutants, etc. The fee standard changed from being single-factor dominant to being multi-factor based.

### 1.2.4. The Fee Collection Range

Environmental deterioration had been exacerbated since the 10$^{th}$ five-year plan, for prioritized achieving economic growth. Increasing transferrable or illegal point-source and nonpoint-source pollution resulted in compromised sanitation condition. In 1982, the fee system was limited to private and collective firms and units. However, supply-driven and conventional exploitation stretched the gap between water demand and the supply. It deviated from the original objective to control all wastewater discharged into water bodies, to cover and regulate any newly added pollution source within the system. Thus, the pollution fee was redefined in the *Pollutant Discharge Fee Collection and Management Regulation* of 2003, which was extended to include to all polluters, including plants, units, individuals from the industrial field, and commercial householders.

### 1.2.5. Calculation Method

The calculation method was reformed from being based on pollution concentration and quantity to being pollution-equivalent based in a more systematic and rational way. In 1982, the water-pollutant discharge fee was calculated by the pollutant-exceeding quantity, which was specific coefficient times the highest exceeding tons. In 2003, the pollution fee was calculated by the pollutant equivalent instead of the pollutant quantity, equaling 0.7 RMB times the largest three pollutant equivalents (the pollutant equivalent is the water-pollutant amount divided by the specific pollutant's equivalent value) plus the doubled penalty fare. The 2014 adjustment increased the equation coefficient from 0.7 RMB to 1.4 RMB and then introduced the block-rate charge. The fee calculated from the equation would times the degree it belongs to. Though the reformed calculation method accounting for regional economic level, characteristics of local industrial pollution, even the environmental cost, it was still far from stimulating emission reduction. Then as balancing inflation, regional standards required to set the fee at 2.46 RMB per pollutant equivalent, based on the consumer price index (Wang et al. 2014).

## 2. Implementation and Related Issues

### *2.1. Process Underlying the Water-Pollutant Discharge-Fee System*

Over three decades of implementation, the water-pollutant discharge fee was enlarged from being collected centrally to being collected at the county level. This was implemented through institutional routines and developed in accordance with different phenomena, such as the transition from a standard-fee charge to a multiple-factors combined charge and more stringent regional standards than the unified regulation. Every step of the policy improvement formed an attempt to motivate enterprises and polluters to reduce pollution. The fee-collection process conducted through an apply–verify mechanism: the water-pollutant discharge unit applies for the total amount of discharged wastewater first, and then the environmental protection agencies verifed the quantity, thereby considering the plant's actual production scale and basing their calculation on the material balance principle. Only if the water-pollutant discharge got permitted can polluters discharge the wastewater legally. After the contaminating behavior, polluters would be charged the water-pollutant discharge fee. The organizational process of apply–verify policy is comparatively complex and several problems and failures appeared during its limited implementation. Currently, the system has been incorporated into the pollutant emission-permit system.

### *2.2. Problems in System Implementation*

### 2.2.1. Low Charging Standard and Weak Enforcement

Water-pollutant discharge-fee system has been adopted as an economic-incentive approach to controlling pollution in a cost-effective way, there is widespread debate on whether the economic-incentive approach is more suitable than a command-and-control policy in developing countries (Barde 1994; Blackman and Harrington 2000; Panaiotov 1994; Motta, Huber, and Ruitenbeek 1999; Wolverton and West 2005). The advantages and disadvantages of such an approach have been analyzed in various situations. Bell and Russel (2002) indicate that developing countries do not possess the conditions for the implementation of market-based strategies. While constructive efforts are made to practice the fee system in China to control water pollution, even its stimulating effects are not that obvious and there exists deviation from the policy's main purpose. Especially, the fee design impedes enterprises from reducing emissions: theoretically, the optimal charge level should be set at the point at which the marginal control cost equals the average marginal loss. If the pollutant-fee standard is higher than enterprises' cost on pollutant reduction, the profit would drive enterprises to pursue technological innovation and device upgrades for emission reduction, or simply to choose to pay for pollutant emission rather than investing in pollution control, so that the incentives of the water-pollutant discharge-fee system would ultimately become ineffective. As difficult to define each pollutant's marginal loss, the average cost on pollutant control has been adopted under technical assistance from World Bank's research of The Design and Implementation of Pollution Discharge Fee in China in 1994, instead of calculating pollutants' marginal treatment fee. Consequently, the cost of illegal discharge is lower than that of abiding by the law, which leads to the strange phenomenon of active contaminating payment versus passive improvement measures on pollution reduction.

### 2.2.2. Low Efficiency

The process of collecting the pollutant discharge-fee met great resistance in its initial steps and significant percentages of enterprises eager to obtain higher profits consequently ignored environmental pollution, with some even refusing to pay the pollution fee, though investing in the visible short-term benefit program. Local environmental protection bureaus are understaffed, underequipped, and underpaid, thereby generating a poor system-implementation condition. The implementation of bank transfers appeared as a milestone in normalizing the pollution discharge-fee collection. As disputes between dischargers and executors mostly occur in the disconnection and noncompliance operation of the apply–verify procedure, false and concealed reports, lax law enforcement, and negotiated charges are common. Additionally, regulators set a single fee that was applied to all plants, with no regard for different levels of environmental tolerance or the demand of the regional-development function. In principle, the system not only lacks flexibility but also obstructs polluters' adoption of abatement technologies. Furthermore, technical support is rarely mentioned and grossly limited in the system's implementation. Disconnections within the water-pollutant discharge-fee system, including discharger registration, establishment of a pollution information-management system, pollutant measurement, pollution-loads calculation, monitor-receiving water bodies, and so forth directly lower the regional environmental authorities' efficiency and weaken the system's influence and power.

### 2.2.3. Revenue Issue

The water-pollutant discharge-fee system generates revenue and should be earmarked for environmental expenditures rather than for departmental expenditures. The water-pollution discharge fee is collected by local environmental authorities. The 2003 pollution discharge-fee-management regulation stipulates that it is to be used as a specific environmental protection fund, covering the major pollution-source-control program, the regional pollution-prevention program, the new technology-application program, and other pollution-control programs. In theory, local environmental protection bureaus only act as fee-collection units, of which the management fee should not be included in the pollution-discharge fee, though departments' management funding is in fact heavily reliant on the pollution-discharge fee. The connection between revenue and expenditure reflects the contradiction in the design of the fee system's mechanism. In this collection mode and in view of departmental interests, it is not strange that local environmental-protection departments are unwilling to control pollution for the purpose of obtaining more revenue. Therefore, the actual amount of the water-pollutant discharge fee is embezzled and detained for other purposes. A fairly large amount of the pollutant fee falls outside of supervision, as the financial department is unable to acquire accurate information on the fee situation, which directly slows down the fee system's development and lowers the efficiency of pollutant control.

## 3. Effectiveness of the Water-Pollutant Discharge-Fee System

### *3.1. System Performance*

Historically, the pollution discharge-fee program has been applied in both developed and developing countries worldwide. This market-based strategy achieved results on air, water, and solids pollution control in different degrees. In France and the Netherlands, the effluent charge is designed to raise revenue for water-pollution-control funding, ultimately achieving improvements in water quality (Tietenberg 1990). Wastewater dischargers in Germany are regulated to meet the minimum pollutant standard and pay for half of the standard-exceeding amount, while the related effluent charge is calculated per pollution unit and increased to control water pollution in the short term (Zhang and Xiong 2012). The practice in European countries explored and developed an independent and mature pollution-management approach involving a legal framework, a political mechanism, organization corporation, revenue supervision (Xiao 2003; Zhou 2006), which achieved significant results in water-pollution reduction. The market-based instrument is an incentive by which to correct environmental externality and its implementation needs to be combined with related policies. The diversity of water resources in America determined the different states' water policies. Empirical study support that water-pollution regulations had significant effects on firms' production in water-polluting industries (Chakraborti 2016; Chakraborti and McConnell 2012; Rassier and Earnhart 2015; Shimshack and Ward 2008). The water policy and water quality mutually influence each other and a reverse effect has been observed in that the decline of water quality stimulates the stringency of the permitted discharge level, leading plants to consequently reduce their pollution emissions (Chakraborti 2016). Significant results in water-pollution control were achieved in developed countries that rely on well-defined property rights. The comments summarized that using market instruments and prices as the primary instruments to control pollution in developing countries is fragmented, due to the fact that the range of political, institutional, and administrative rules, practices, and processes is powerless in handling market-based strategies. In India, water governance is a challenge at almost all scales, whereby the states' limited power not only weakens their capacity to solve transboundary water issues but also makes them powerless in the field of domestic water use and pollution (Chokkakula 2012). Evidence in Malaysia suggests the financial condition limits the investment in technical-abatement measures for water-pollution control in the sewage system, and that the lack of cooperation between governments and plants impedes water-quality improvement (Muyibi, Ambali, and Eissa 2008).

Water and its derivative functions serve economic users, so that adopting a market instrument by which to guide water allocation and pollution issues is theoretically considered as a cost-effective approach. Therefore, the market paradigm was introduced and promoted in developed and transitioning countries, including China. To assess the market's power on water governance in China, several questions have been addressed: How successful is the water-pollutant discharge-fee system in controlling water pollution? Which factors are responsible for its success? With regard to the first question, increasing empirical evidence demonstrates that the combination of the fee system and water-pollution regulations reduces pollution-intensive activity (Chen et al. 2018; Yuan, Jiang, and Bi 2010). Additionally, the collection of water-pollution discharge fees directly revealed the system's degree of enforcement, which could be one of the main factors by which to assess its effectiveness. The national grand total pollution-fee amount from 1992 to 2014 is 237.59 billion RMB. The adjustment of the fee system in 2003 can be seen as a turning point: the growth rate of the pollutant discharge fee was below 10% until 2003, after which it doubled to reach its first peak at 17.68 billion RMB in 2008. It subsequently fluctuated due to the decreasing number of enterprises during the financial crisis. The total of the discharge-fee collection corre-

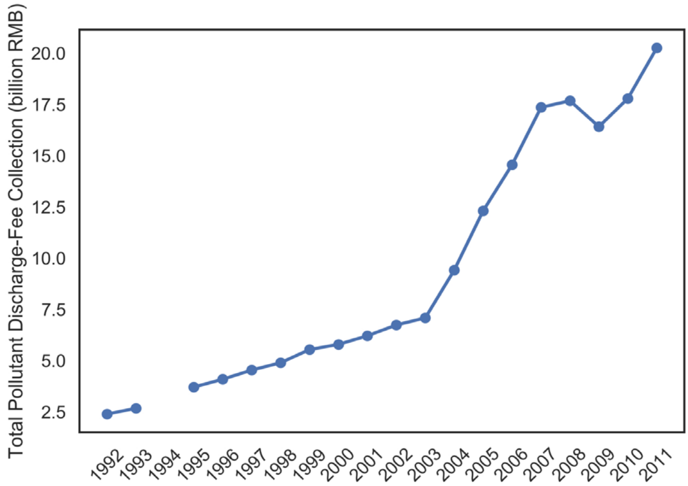



**Figure 1.** Total pollutant discharge-fee collection in China (Source: *China Statistical Year Book*)

sponds to the country's economic development and transition in industrial structure. With gradual economic recovery, the revenue increased to reach another peak at 20.48 billion RMB in 2013 (Figure 1).

Data on the water-pollutant discharge fee collected within the total pollution fee reveals more information (Figure 2). Before 2003, the water-pollutant discharge-fee revenue increased at a stable rate, as it is collected under the single-factor charging principle and not many types of pollutants are defined in the standard. The promulgation of the 2003 regulation completely altered the collection and the revenues from the fee increased to three times higher than ever before, reaching their peak in 2007. Economic factors can lead enterprises' productivity to drop, as well as the wastewater-discharge amount and the fee. In general, the water-pollutant-discharge revenue is doubled after the pollutant's equivalent charge, but with more exposed the revenue issues. As shown, the water-pollutant fee accounted for over 50% of the total pollutant-discharge fee. Theoretically, the actual water-pollutant fee amount should lie far from the total pollution fee amount. However, neither the water-pollutant discharge fee nor the total pollution fee was fully collected. Regulations do help to promote fee collection, but implementation without supervision contributes to the fee breach and absence of revenue management.

### *3.2. Assessment on Pollution Control*

The results of the water-pollutant discharge-fee system are demonstrated by pollution control. Before the system was implemented, the wastewater-discharge amount increased rapidly without charge. The situation was curbed and further deterioration was avoided since with promulgation of the temporary regulation. Subsequently, the wastewater discharge surged in line with the boost in economic growth,

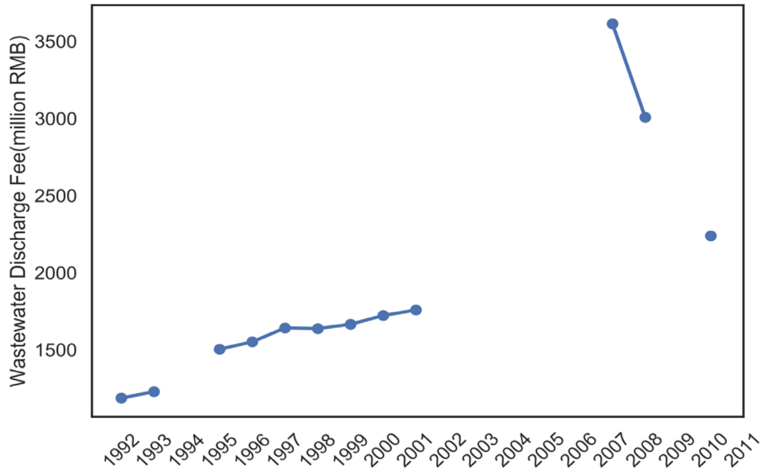



**Figure 2.** Water-pollutant discharge-fee collection in China (Source: *China Statistical Year Book*)

increasing from 1988 to 1994. The pollution amount was jumped from 1995 to 2001. Apparently, the approval of the 2003 regulation retarded the increasing rate of pollution, after which the total amount of water pollutant continued to grow, but comparatively more gently from 2006 to 2013, when the rate dropped from 8.7% to 2.3%. During the 10 years of rapid industrial development, the total wastewater discharge was not doubled but did maintain an increasing rate (Figure 3).

Pollutants such as COD and ammoniacal nitrogen within wastewater can explain water quality in detail (Figure 4). The fee system operated effectively since its reform. With slight fluctuations, the amount of COD was stable in 2008, at 13.2 million tons, after which it increased sharply from 2011 to 2014 to almost double than ever before. While the amounts of ammonium and nitrogen appeared quite stable until 2010, they doubled in 2011 at 2.604 million tons and then decreased in the following three years to 2.385 million tons. The concentrations of ammonium and nitrogen were more clearly controlled than the COD concentration during the period of rapid economic growth. Though the water quality cannot be fully determined by the levels of COD, ammonium, and nitrogen, the fee system's stimulation of pollutant reduction still contributes to the gradual improvement of water quality, though not as obviously as expected.

Water quality is also assessed from a macro perspective in seven major river basins nationwide (Figure 5). The percentage of surface water with good quality and less contamination (levels I–III) sharply declined in 2001. The amount of healthy river bodies almost equals that of contaminated rivers. The fact that the amount of healthy and unhealthy rivers increased at the same speed illustrates that the effect of the fee program was insufficiently effective in

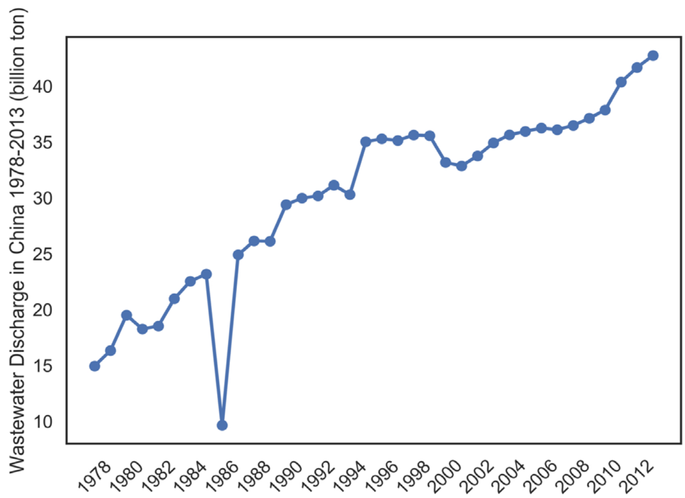



**Figure 3.** Wastewater discharge in China
(Source: *China Statistical Year Book*)

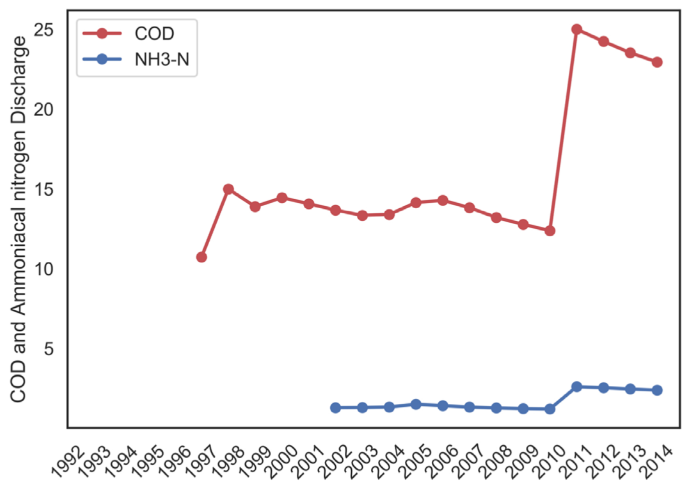



**Figure 4.** COD and ammoniacal nitrogen discharges in China
(Source: *China Statistical Year Book*)

controlling pollution before the policy reform. The most contaminated rivers boost to occupy a large amount of all seven river systems at that time, but they were identified and their water quality was largely improved since 2003. The number of rivers with good water quality experienced sharp increases twice before 2014, during which time the rivers with the worst water quality (levels IV–V) were controlled to achieve a decrease in contamination. The amount of river waters with the lowest quality levels still amounts to 30% of all river waters.

Pollution control has been assessed through water quality at both the micro and macro scales, and can also be interpreted from the perspective of industry structure. Within the rigid demand of economic development, the

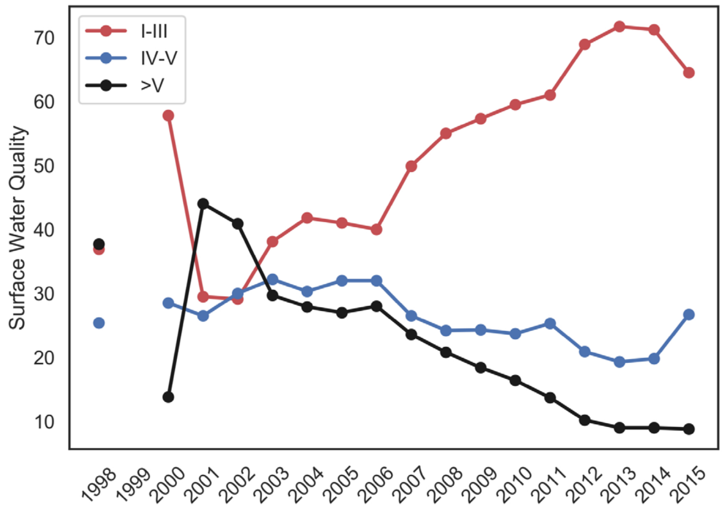



**Figure 5.** Surface water quality.
(Sources: *China Statistical Year Book*, *China Environmental Statistical Bulletin*)

discharges of COD and ammoniacal nitrogen in industrial wastewater dropped by 48.8% and 36.1%, respectively, with a particularly strong decline after the 11th five-year plan. Additionally, the reuse rate of industrial water gradually increased. The average pollutant concentration in industrial wastewater continues to decline in accordance with the changing macro-pollution situation. The average COD concentration in industrial wastewater also appeared to decline (MEP 2015), though its levels are still far from those required by the water-environment function. More generally, the water quality in most areas does not meet the aquatic-environment standard. The water-pollutant discharge-fee system adopts an integrated approach and is emphasized throughout policies and legislation, planning and management, wastewater quantity and concentration, surface and underground water quality, and so forth. However, the pollution facts interpreted above illustrate the limitation in the implementation of the fee system, and the disconnections of range links within the system impede its efficiency. No direct evidence indicates improvements of integrated wastewater management.

## 4. Driving Forces of the Water-Pollutant Discharge-Fee System

With regard to the second question, several factors relevant to the fee system require examination throughout the implementation environment. From the analysis above, the results obtained in the evaluation of the fee system show its deficiencies from a multi-scale perspective. For instance, regarding the management of the water-pollutant fee, polluters' behavioral regulation and administrative support are not fully realized. Regarding factors affected by the effective enforcement of the system, it is essential to identify the strengths and weaknesses of the process. Though the water-pollutant discharge-fee system in China underwent reform and adjustment, several thorny issues appeared within the application process, such as differences in the total fee amount, which consequently led to the derivation of the policy goal. Each element is implemented based on the legal, administrative, socioeconomic, and political circumstances. As a linking procedure, the policy's implementation relies on executors to realize the policy target through explaining, propagating, testing, conducting, and supervising. Integrated management of water resources is required to enhance the system.

### *4.1. External Factors*

#### 4.1.1. Social Factors

The water-pollutant discharge-fee system is implemented to correct the market failure and guide water use and allocation. The main objective of the water-pollutant discharge-fee system is to protect water resources' capacity and productivity. Society provides the platform for the system's implementation. System-policy enforcement refers to the two main parties: the policy executor who represents the government's behavior (Yang and Wang 2013) and the water-pollutant dischargers, which include individuals, units, and plants. Most of the system's procedures are highly reliant on governmental officers. The water-pollutant fee-collection work is not only limited by understaffing, but also by the noncompliance operations and related individual behaviors of local officials. Therefore, different extent of pollutant-fee arrears such as pollutant-fee negotiation and relationship-fee collection, less payment are quite common. The national average fee-collection rate is only 50% and government interference is also detected in the policy-enforcement process. Local protection greatly contributes to increasing the pollution intensity (Jiang, Lin, and Lin 2014). Administrative interference under regional protection can be expected to generate more space and uncertainty in the policy's implementation, which is likely to lead to rent-seeking behavior within the system's operation. Reversely, pollution behavior also affects the system's efficiency and proper implementation. False consciousness of pollution and the idea of emission as "more payment, more discharge" are formed within the discharger group by the confused relationship between the system-enforcement authorities and dischargers' noncompliance operations. Certain phenomena such as false or concealed reports in pollutant-amount applications are sure to impact the collection of the full pollution-discharge fee amount. Simultaneously, the pursuit of economic growth, which was local governments' most important target and assessment index, significantly impeded the system's implementation.

Whether firms and plants actively pay for their pollution discharge is determined by their net profit. The implementation of the fee system may alter enterprises' production activities (Wu 2015) and increases production costs, which subsequently reduces profit. Firms would choose the cheapest way to solve the pollution issue: either by discharging or adopting advanced technology. The fee system regulates the relationship between enterprises and the environment in order to offset the cost. Correspondingly, the decline in recorded dischargers since the system's reform in 2003 proves the stringency of the regulations and related standards. The declining number of enterprises versus the increasing wastewater-fee amount in the following decade implies an increase in the fee amount, which indirectly demonstrates that the standard also became more stringent, motivated by the aim of forcing enterprises to adopt abatement technology. Simultaneously, the severe environmental deterioration within that decade not only increased public concern about water-resource scarcity, but also strengthened the importance of the fee system's follow-up monitoring.

### 4.1.2. Economic Factors

As an economic incentive, the water-pollutant discharge-fee system could not be implemented independently without taking into account the macro-economic circumstances. The economic structure and its development drive the water-pollutant discharge-fee system, as revealed in the changing trend of the GDP (Figure 6). The GDP stably increased at the rate of 14.2% until its peak in 2007, after which it fluctuated, decreasing until 2014 during the economic recovery from the financial crisis, which correspondingly affected the water-pollutant discharge-fee system. The relationship between the GDP and the water-pollutant discharge fee is demonstrated by the same though more obviously changing trend in the former. This means not only that the fee program is implemented at a specific scale, but also that it has a strong correlation to the macro policy and strategy.

## *4.2. Internal Factors*

### 4.2.1. Data Management

The pollution fee is calculated based on monitoring data. There exists difficulty in obtaining data accuracy, even in the same river basin area, data monitored by different sectors with different methods do not match with each other. The water-pollutant discharge data are monitored by executors and/or the dischargers. From a governmental perspective, the data are rarely shared or combined with different data-monitoring methods, which makes it difficult to conduct further data analyses. Additionally, pollution data are also difficult to access, data sharing and transparency are low, and the coverage of the pollution monitoring system is limited, which intensifies the uncertainty and confusion. As the data for several pollution areas are absent, unified and comprehensive pollution-data management needs to be pursued to fill the gap.

### 4.2.2. Enforcement Transparency

The procedures in the apply–verify process are complex and limit the system's full promotion, and the information asymmetry for executors constrains the development of the water-pollutant discharge-fee system. The accuracy and re-

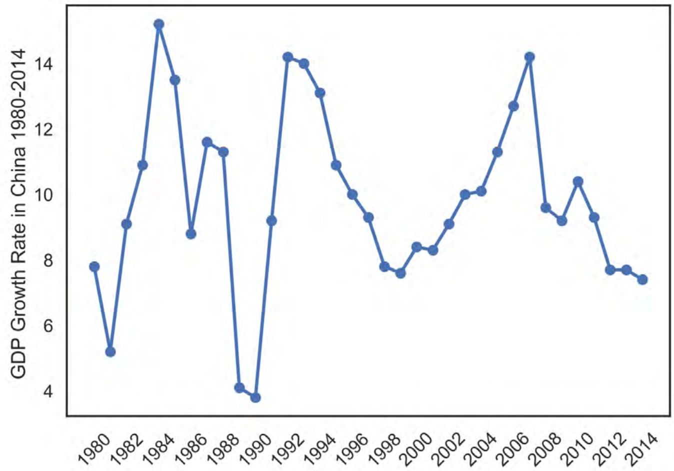



**Figure 6.** GDP growth rate in China from 1980 to 2014
(Source: *China Statistical Year Book*)

liability of related information strongly affect the system and policy conduction. Obstruction from dischargers, governmental interference, and non-compliance operations mask the actual pollution situation and hamper the fee-collection work Also with institutional barrier, the information cannot be transferred fluently and accurately. The disconnection caused by information asymmetry indicates the absence of information platforms across governmental authorities, dischargers, and benefit-related parties.

### 4.2.3. Supervision and Cooperation

Despite the reform, the water-pollutant discharge-fee system still has a long way to go in achieving efficiency, equality, and effectiveness in pollution control. The fee system takes an adaptive-management approach, the power and effectiveness of which are reflected in the actual implementation. The pollution discharge-fee program lacks supervision and a cooperation mechanism across executive authorities. Not only the fee-expenditure direction but also the fee-system operation needs supervision from other departments or a third party. Supervision and cooperation are antithetical and mutually complement each other. The supervisory participation from each side is significant for strategic cooperation. The question of how to connect different units and promote the departments' synergy becomes the challenge that needs to be addressed in order to fill the gap. Such synergy requires that the prospective management faces dynamic indicators, including industry change, policy adjustment, and technical improvement. Therefore, it not only needs to meet the rigid demand of pollution control but also better conduct the fee system's implementation. Moreover, institutional barriers lead to both external and internal conflicts during the system's operation. Such problems and disconnections reflect the passivity of pollution control, which also largely increases the cost of the fee system's implementation.

## 5. Conclusion and Discussion

This article evaluates the water-pollutant discharge-fee system in China, which adopts a progressive approach to pollution control. The system has achieved staged results and undergone reforms in multiple aspects, including fee-standard improvement, fee-calculation innovation, and pollutant-indicator construction. It is highly contextual and influenced by economic, social, and political external factors, as well as by internal theoretical and operational contexts. Through the system's reform in 2013, it was redefined as a comparatively comprehensive system with the ability to face and handle the complex pollution situation. However, problems arose due to the limited scope of legal measures and technical support, and weaknesses appeared in the system regarding departmental conflicts of interest, the contradiction of increasing environmental pollution versus decreasing fee collection, and other problems during the design, implementation, and supervision of the fee system, which adversely influenced the system's efficiency and the water quality.

With the recognition of these deficiencies, the pollution fee transitioned to an environmental tax in 2018. The levy of environmental tax may enhance the rigor of law enforcement, avoid administrative intervention and rent-seeking, and strengthen the implementation. Three lessons can be drawn from the fee system in view of better implementing the new environmental tax. First, data collection and management should be significantly improved. Data are the basis for the implementation of essentially any environmental policy, but their deficiencies plagued the fee system's implementation. The pollutant-discharge monitoring system should be fully developed according to legal requirements. Data should be shared among stakeholders, including polluters, environmental-protection agencies, and taxation departments. Second, an effective linkage needs to be developed with the pollution-emission permit system that since recently is being fully implemented. The permit system could serve as a key foundation for the levy of environmental tax. Third, environmental-protection agencies should coordinate well with taxation agencies. Since the pollution fee-to-tax reform, collection agencies are no longer taxation agencies instead of environmental-protection agencies. The collection procedures and rules for environmental tax should be made to fit with general tax-collection principles. The coordination between the two agencies should thus focus on both data sharing and tax-collection procedures/rules.

## References

Bai, Y., and X. Qu. 2009. "The Reformation Trend of Pollutant Discharge Fee System." *Environment Protection* 20: 43-44.

Barde, J.P. 1994. *Economic Instruments in Environmental Policy: Lessons from the OECD Experience and Their Rele-*

*vance to Developing Countries*. Working Paper No. 92. OECD Development Center.

Bell, R., and C. Russell. 2002. "Environmental Policy for Developing Countries." *Issues in Science and Technology Spring Press* 18 (3): 63-70.

Blackman, A., and W. Harrington. 2000. "The Use of Economic Incentives in Developing Countries: International Experience With Industrial Air Pollution." *Journal of Environment and Development* 9 (1): 5-44.

Chakraborti, L. 2016. "Do Plants' Emissions Respond to Ambient Environmental Quality? Evidence from the Clean Water Act." *Journal of Environmental Economics & Management* 79: 55-69.

Chakraborti, L., and K.E. McConnell. 2012. "Does Ambient Water Quality Affect the Stringency of Regulations? Plant-Level Evidence of the Clean Water Act." *Land Economics* 88 (3): 518–35.

Chen, Z., M.E. Kahn, and Y. Liu., Wang Z.. 2018. "The Consequences of Spatially Differentiated Water Pollution Regulation in China." *Journal of Environmental Economics & Management* 88: 468-85

Chokkakula, S. 2012. *Disputes, Politicization, and Democracy: Interstate Water Disputes in India. Research Unit for Livelihoods and Natural Resources*. Working Paper. No. 108.

Fan, X. 2013. "Multiple Views on Pollutant Discharge Fee System Research." *Ecological Economy* 11 (41): 73-78.

Gao, S., W. Su, and Q. Yang. 2014. "Assessment of the Expropriation Efficiency of Regional Pollutants Levy System in China Based on DEA-Malmquist Model." *China Population, Resource and Environment* 24 (2): 23-29.

Ge, C., and J. Wang. 2001. "Using Market Instrument to Eliminate Pollution: Discharge Fee, Environment Tax and Emission Trading." Review of *Economic Research* 40(02): 28-43.

Genia, K., and H. William. 2012. "Local Energy Efficiency Policy Implementation in China: Bridging the Gap Between National Priorities and Local Interests." *China Quarterly* 211 (211): 765-85.

Hammer, M.J. 1996. "Water and Wastewater Technology." *Microchimica Acta* 74 (5-6): 445-54.

Hou, H. 2008. "The Comparison of Pollution Fee Charge and Discharge Standard." *Journal of TongJi University* (*natural science*) 36 (6): 777-81.

Jiang, L., C. Lin, and P. Lin. 2014. "The Determinants of Pollution Levels: Firm-Level Evidence from Chinese Manufacturing." *Journal of Comparative Economics* 42 (1): 118-42.

Kahn, M.E., P. Li, and D. Zhao. 2015. "Water Pollution Progress at Borders: The Role of Changes in China's Political Promotion Incentives." *American Economic Journal Economic Policy* 7 (4): 223-42.

Li, W. 2015. "Spatial Econometrics Test of Pollutant Discharge System's Driving on Green Technological Innovation by Taking 29 Provinces and Regions' Manufacturing Industries as Examples." *Science Management* 36 (6): 1-9.

Liu, M. 2009. "Limitation During Application of Term 74 of New Water Pollution Control Law." *Journal of China University of Geosciences* 9 (6): 41-46.

Lv, J. 2009. "What Is Water Pollutant Discharge Fee System." *Jiangsu* 5: 46.

Ma, L., J. Wang, and D. Wang. 2013. "The Strategy and Policy Framework for Water Pollution Control in China During the 12th Five-Year Plan Period." *China Environment Science* 33 (2): 377-83.

MEP. 2015. *Environmental Statistic Annual Report of China*. http://www.zhb.gov.cn/gzfw.

Motta, S.D., R. Huber, and J. Ruitenbeek. 1999. "Market Based Instruments for Environmental Policy-Making in Latin America." *Environment and Development Economics* 4 (2): 177-201.

Muyibi, S.A., A.R. Ambali, and G.S. Eissa. 2008. "The Impact of Economic Development on Water Pollution: Trends and Policy Actions in Malaysia." *Water Resources Management* 22 (4): 485-508.

Panaiotov, T., 1994. "Economic Instruments for Environmental Management and Sustainable Development." Nairobi UNEP.

Rassier, D.G., and D. Earnhart. 2015. "Effects of Environmental Regulation on Actual and Expected Profitability." *Ecological Economics* 112: 129-40.

Shimshack, J.P., and M.B. Ward. 2008. "Enforcement and Over-Compliance." *Environmental Economics and Management* 55 (1): 90–105.

Tietenberg, T.H. 1990. "Economic Instruments for Environmental Regulation." *Oxford Review of Economic Policy* 6 (1): 17–33.

Wang, B., Y. Wu, and P. Yan. 2008. "Environmental Regulation and Total Factor Productivity Growth: An Empirical Study of the APEC Economics." *Economic Research* 64(5): 19-32.

Wang, J., F. Long, C. Ge, and S. Gao. 2014. "A Road Map of Reforming Pollution Charge System and Adjusting Its Charge Rates in China." *Environment Protection* 42 (19): 37-39.

Wolverton, A., and S. West. 2005. "Market-Based Policies for Pollution Control in Latin America." In *Environmental Issues in Latin America and the Caribbean*, edited by A. Romero and S. West. Springer ,Dordrecht, pp121-146

Wu, J. 2015. "Profit Thinking on Pollutant Discharge Fee System." *Environment Protection* 43 (16): 21-25.

Wu, S., M. Xu, and K. Ma. 2015. "Progress of the '13th Five-Year Plan' for Key River Basin Implementing the Action Plan for Prevention and Control of Water Pollution." *Environment Protection* 43 (18): 14-17.

Xiao, J. 2003. "The Comparison and Study of Water Pollution Discharge Tax in Europe." *Journal of Central South Economics and Law University* 197 (2): 76-82.

Xu, Y. 2014. "Research on Block Fee Standard of Pollutant Discharge Fee System." *Environment Protection* 42 (20): 54-57.

Xu, K., and H. Ni. 2004. "The Revolution of Pollutant Discharge Fee System." *Price Theory and Implementation* 9: 15-16.

Yang, C., and J. Wang. 2013. "Research on Regional Pollutant Discharge Fee System." *China Population, Resource and Environment*, 23(159. s2): 313-136.

Yu, J., P. Wang, and J. Cai. 2005. "Current Pollutant Discharge Fee System and Related Discussion." *Environment Science & Technology* 28 (5): 60-62.

Yuan, Z., W. Jiang, and J. Bi. 2010. "Cost-Effectiveness of Two Operational Models at Industrial Wastewater Treatment Plants in China: a Case Study in Shengze Town, Suzhou City." *Journal of Environmental Management* 91 (10): 2038-44.

Zhang, G. 2008. "Analysis of Pollutant Discharge Fee System in China." *Economic Forum* 13: 123-24.

Zhang, H., and B. Xiong. 2012. "The Economic Analysis of German Sewage Charges System Based on the Perspective of Baumol-Oates Tax." *China Population, Resource and Environment* 22 (10): 71-79.

Zhou, J., and Y. Chen. 2008. "Chinese Regional Environmental Efficiency Analysis." *Statistic & Decision* 14: 44-46.

Zhou, G. 2006. "The Comparative Analysis of Foreign Water Resource Tax." *Journal of Economics of Water Resource* 24 (5): 28-30.

# Assessing the Implementation of Local Emission Trading Schemes in China: Econometric Analysis of Market Data

Lili Li[1]


## Abstract

Emission trading scheme (ETS) has been used for reducing emissions of industries in developed countries since the 1990s. This study contributes to existing research by focusing on the implementation of local ETSs for reducing carbon dioxide ($CO_2$) emissions in the context of China. Based on time-series analysis techniques, the study investigates into the market dynamics of the local ETSs in China from their launching dates to 30 June 2017, addressing the relations between energy prices and the prices of China's $CO_2$ emission allowance (CEA). The price values of CEA and the level of trading volumes vary across the ETS pilots due to their differences in policy features, local business environment and governments' support. Between the two provincial ETSs, Hubei ETS had less volatile price and larger weekly trading volume, while Guangdong ETS had higher CEA price on average. Among the five city ETSs, Tianjin and Chongqing ETSs were not so market-oriented considering their lower prices and much less active trading activities. The regression analyses found that the links between energy prices and CEA prices were different among local ETSs as well, which may be because of different demand and supply dynamics in the ETS markets and energy markets. There was no Granger causality from energy prices to CEA prices in Guangdong or Hubei. With respect to the city ETSs, the steam coal price granger caused the CEA price in Shanghai, and the changes in coal price had a negative short-term effect on the changes in CEA price, indicating that an increase in coal price could arrest coal associated pollution due to a substitution of coal with less carbon-intensive fuels (e.g. natural gas). In Beijing, the results show that the international oil price granger caused the CEA price, and there was a positive effect of the oil price changes on the CEA price changes in the short-run, implying a


[1] *Lee Kuan Yew School of Public Policy, National University of Singapore;* li.lili89@u.nus.edu



doi: 10.18278/cpj.1.1.2

substitution of oil with coal when oil price increases. In addition, the Shanghai Shenzhen 300 stock index granger caused the CEA price in Beijing, Shanghai and Shenzhen, suggesting that the CEA price was affected by the macroeconomic environment. Furthermore, the results suggest that in all local ETSs, the CEA prices were at a higher level at the initial stage but significantly dropped after the compliance deadline in the next calendar year due to decreases in CEA demand. As ETS will continue to be an important tool for climate change mitigation in China, the diverse policy features and performance of the local ETSs served as crucial references for the ETS at national level. The policy implication is that the national ETS should reduce the CEA price volatility caused by energy price fluctuations and put emphasis on local contingencies.

***Keywords:*** Emission trading scheme; local pilots; emission allowance price; energy price

# Evaluación de la implementación de esquemas de comercio de emisiones locales en China: análisis econométrico de datos de mercado

## Resumen

El esquema de comercio de emisiones (ETS) se ha utilizado para reducir las emisiones de las industrias en los países desarrollados desde los años noventa. Este estudio contribuye a la investigación existente al centrarse en la implementación de ETS locales para reducir las emisiones de dióxido de carbono ($CO_2$) en el contexto de China. Con base en las técnicas de análisis de series temporales, el estudio investiga la dinámica del mercado de los ETS locales en China desde su fecha de lanzamiento hasta el 30 de junio de 2017, abordando las relaciones entre los precios de la energía y los precios de la emisión de $CO_2$ en China (CEA). Los valores de los precios de CEA y el nivel de los volúmenes de negociación varían en los programas piloto de ETS debido a sus diferencias en las características de las políticas, el entorno empresarial local y el apoyo de los gobiernos. Entre los dos ETS provinciales, el ETS de Hubei tuvo un precio menos volátil y un mayor volumen de transacciones sema-

nales, mientras que el ETS de Guangdong tuvo un precio de CEA más alto en promedio. Entre los cinco ETS de la ciudad, los ETS de Tianjin y Chongqing no estaban tan orientados hacia el mercado teniendo en cuenta sus precios más bajos y actividades comerciales mucho menos activas. Los análisis de regresión encontraron que los vínculos entre los precios de la energía y los precios del CEA también eran diferentes entre los ETS locales, lo que puede deberse a las diferentes dinámicas de oferta y demanda en los mercados de ETS y los mercados de energía. No hubo causalidad de Granger por los precios de la energía a los precios del CEA en Guangdong o Hubei. Con respecto a los ETS de la ciudad, el precio del carbón a vapor causó el precio del CEA en Shanghai, y los cambios en el precio del carbón tuvieron un efecto negativo a corto plazo en los cambios en el precio del CEA, lo que indica que un aumento en el precio del carbón podría detener la contaminación debida a la sustitución del carbón por combustibles menos intensivos en carbono (por ejemplo, carbón, gas natural). En Beijing, los resultados muestran que el aumento en el precio del petróleo internacional causó el precio del CEA, y hubo un efecto positivo de los cambios del precio del petróleo en los cambios del precio del CEA en el corto plazo, lo que implica una sustitución del petróleo con el carbón cuando el precio del petróleo aumenta. Además, el índice de acciones de Shanghai Shenzhen 300 tuvo un efecto granger sobre el precio del CEA en Beijing, Shanghai y Shenzhen, lo que sugiere que el precio del CEA se vio afectado por el entorno macroeconómico. Además, los coeficientes en los break dummy sugieren que en todos los ETS locales, el precio del CEA se ubicó en un nivel más alto al principio, pero se redujo significativamente en el próximo año debido a la disminución de la demanda del CEA. Dado que el ETS seguirá siendo una herramienta importante para la mitigación del cambio climático en China, las diversas características de las políticas y el desempeño de los ETS locales sirvieron como referencias cruciales para el ETS a nivel nacional. La implicación de la política es que el ETS nacional debería reducir la volatilidad de los precios del CEA causada por las fluctuaciones de los precios de la energía y poner énfasis en las contingencias locales.

***Palabras clave:*** esquema de comercio de emisiones; pilotos locales; precio de permiso de emisión; precio de la energía

# 评价中国地方排放交易机制的实施：市场数据的计量分析


摘要

20世纪90年代以来，排放交易机制(ETS)一直用于减少发达国家的工业污染物排放。本文通过聚焦中国为减少二氧化碳($CO_2$)实施的ETS试点，从而对现有ETS研究作出贡献。本文运用时间序列分析方法研究中国ETS试点自推出之日起至2017年6月30日的市场动态，重点分析了能源价格与中国二氧化碳排放限额(CEA)价格之间的关系。CEA的价格和交易量水平因ETS试点在政策设计、当地经济环境和政府支持力度等方面的差异而有所不同。在两个省级ETS试点中，湖北ETS试点价格波动较小，平均每周交易量较高，而广东ETS试点平均CEA价格较高。在五个城市级的ETS试点中，天津和重庆的ETS试点并不以市场为导向，其价格较低，交易活动也少得多。分析结果表明，能源价格与CEA价格之间的关系在不同ETS试点中也存在差异，这可能是由于其ETS市场和能源市场的需求和供应动态不同。广东和湖北的ETS试点中，能源价格与CEA价格之间没有表现出显著的格兰杰因果关系。至于五个城市ETS试点中，上海的动力煤价格变化是推动CEA价格变化的格兰杰原因，这种影响是负的且是短期的，暗示着煤价的上升将会导致煤炭被低碳燃料（如天然气）代替，从而抑制煤炭燃烧相关的污染。北京的研究结果表明，国际油价变化是引起CEA价格变化的格兰杰原因，这种影响是正的且是短期的，表明油价的上升将会导致石油与煤炭之间的能源替代。此外，沪深300指数变化是引起北京、上海和深圳的CEA价格变化的格兰杰原因，表明CEA价格受到宏观经济环境的影响。此外，结果表明，所有试点的CEA价格在政策实施初始阶段处于较高的水平，但由于CEA需求减少，CEA价格在第二年履约期后均显著下降。ETS将继续作为中国缓解气候变化的重要工具，因而ETS试点的不同政策特点和市场表现将为ETS在全国范围内的实施提供重要参考。且ETS在全国范围内实施时，应当注意能源价格波动引起的CEA价格波动，并重视地区差异性。


关键词：排放交易机制；地区试点；排放许可价格；能源价格

## 1. Introduction

The strong interest in using "New Environmental Policy Instruments"(NEPIs)—including economic instruments that emphasize market incentives and suasive instruments that encourage voluntary environmental changes, in contrast to traditional direct government command and control (GCAC) approaches, has been prevalent in developed countries since 1980s, with numerous academic studies on their implementation and effectiveness. However, there is a lack of evidence illustrating the complexity in the design and implementation of NEPIs in developing countries with limited monitoring and enforcement resources.

More and more scientific evidence shows that greenhouse gas (GHG) emissions contribute to the global climate change, and emission trading scheme (ETS) is an important policy instrument for reducing GHG emissions that the Kyoto Protocol suggested. ETS has been used for emission abatement at the international level such as the European Union (EU) ETS, at the country level such as the South Korea ETS, and at the regional level such as the Regional Greenhouse Gas Initiative. Between 2013 and 2014, seven domestic pilots of ETS were established in China, including two provincial level ETSs and five city level ETSs. Since the end of 2017, China has started to establish the national ETS. Experiences from the pilots become important references for the national ETS.

Following the polluter-pay principle, ETS creates a market for carbon dioxide ($CO_2$) emission allowances to encourage the internalization of emission abatement costs. If the market functions well, the emission allowance price can reflect the marginal cost of emission abatement and encourage enterprises to adopt low-carbon technologies. One of the major implementation concerns is that targeting enterprises fail to respond in ways anticipated by policymakers due to low economic incentives (Weaver 2010). In practice, the emission allowance price tends to be low and volatile, hardly reflecting the real abatement cost. For instance, the carbon price of EU-ETS once decreased to almost zero in its first carbon trading period from 2005 to 2007 (Alberola, Chevallier, and Chèze 2008). In those cases, industrial participants would have little incentives to adopt costly environmental measures to reduce $CO_2$ emissions internally.

China is the largest emitter of $CO_2$ emissions, mainly caused by its enormous size of population and economy, and the high share of coal (more than 60%) in its energy mix (Olivier et al. 2015). Although China has no compulsory GHG reduction obligation in the Kyoto Protocol, it has committed to achieving intensity-based targets partly in response to increasing international pressures that proceeded the Copenhagen negotiation. As its INDC for the Paris Agreement, China promised to reduce its $CO_2$ emission intensity by 60%–65% by 2030 in relative to 2005 (NDRC of China 2015). To achieve the national $CO_2$ emission reduction target, China has used many policy instruments and ETS is one of the most important ones.

The first piloting ETS for $CO_2$ emission reduction in China is Shenzhen ETS, established in June 2013. Subsequently, Shanghai, Beijing, Guangdong, and Tianjin developed ETS pilots by the end of 2013. After that, Hubei and Chongqing developed ETS pilots in 2014. While some similarities of the policy design exist among pilots, such as the use of intensity-based cap, the inclusion of power generation sector, and the use of free allocation, there are variations regarding the policy design across pilots, such as the difference in non-compliance penalties, the difference in sectoral coverage, and the difference in the use of auction to complement free allowance allocation (see Table 1).

All seven ETS pilots in China assign intensity-based caps to participants compared to the use of absolute caps in other ETSs. One of the critical justifications for the intensity-based cap is the uncertainty of the business-as-usual output (Quirion 2005). China's GHG emissions have not reached its peak yet, and the country still keeps a rapid economic development. Under the circumstances, reaching an absolute emission cap can be costly, but setting intensity-based caps ensures flexibility considering the emissions from future

**Table 1.** Policy Features of ETS Pilots in China

| Program (duration) | ETS pilots (2013–present) |
|---|---|
| Scope | City-level ETS: BJ, SH, SZ, TJ, CQ<br>Provincial-level ETS: HB, GD |
| Identifying potential participants | Enterprises are capped if they meet thresholds as follows.<br>(1) BJ: annual emission >10,000 tons $CO_2$e on average during 2009–2012 (mandatory); enterprises with annual energy consumption > 2,000 tce can voluntarily participate.<br>(2) SH: emission >20,000 tons $CO_2$e in 2010 or 2011 for major industrial sectors; the threshold is >10,000 tons $CO_2$e for non-industrial sectors<br>(3) SZ: industrial enterprises with emission>3,000 tons $CO_2$e per year, or public buildings with area >10,000m$^2$.<br>(4) TJ: annual emission>20,000 tons $CO_2$e in any year since 2009.<br>(5) CQ: annual emission >10,000 tons $CO_2$e in any year during 2009–2012.<br>(6) GD: industrial enterprises with annual emission >10,000 tons $CO_2$e on average or any year in 2010–2012; non-industrial enterprises with emission > 5,000 tons $CO_2$e<br>(7) HB: emission > 150,000 tons $CO_2$e for major regulated sectors in 2010 or 2011 |

| | |
|---|---|
| Cap coverage | (1) BJ: covering power generation sector, cement, heat supply, petrochemical, car manufacturing, and etc.; public buildings, aviation, large restaurants, hotels and banks<br>(2) SH: covering power generation sector, steel, non-ferrous, paper, rubber, chemicals, petrochemical, textile and etc.; airlines, ports, airports, large commercial shops and hotels<br>(3) SZ: covering 26 industrial sectors as well as power generation sector, gas and water supply; 197 public use buildings; participation open to any financial institution.<br>(4) TJ: covering power generation sector, steel & iron, petrochemical, chemicals, civil construction, heat supply, oil and gas mining<br>(5) CQ: covering power generation sector, steel & iron, cement, metal alloy, calcium carbide, caustic soda, electro-plated aluminum<br>(6) HB: covering power generation sector, steel, cement, non-ferrous, paper, chemicals, automobile manufacturing, glass and etc.<br>(7) GD: covering power generation sector, steel, cement, non-ferrous, paper, ceramics, petrochemical, plastics and etc. |
| Scale | Share of the cap coverage in the total emissions in the city/province:<br>(1) BJ-40%; (2) SH-57%; (3) SZ-38%; (4) TJ-60%; (5) CQ-40%; (6) GD-55%; (7) HB-35% |
| Setting enterprise-level targets | Intensity-based emission caps are assigned to participating enterprises through either free allocation or auctioning, and the enterprises can buy more emission allowance from the ETS market:<br>(1) Free allocation through grandfathering approach is the prevalent allocation method across pilots;<br>(2) Auctioning has been used as a complementary allocation method in GD, SH and SZ to allocate a small portion of allowances. |
| Enforcing compliance | (1) Except CQ and TJ, every ETS pilot has monetary penalties for non-compliance.<br>(2) Additionally, SZ, HB and GD have a further penalty, which is deducting the excess emissions from the following complinace period's emission allowance. |

Note: Swartz (2013), Perdan and Azapagic (2011), and relevant Chinese policy documents. “BJ” is short for Beijing, “SH” for Shanghai, “SZ” for Shenzhen, “TJ” for Tianjin, “CQ” for Chongqing, “GD” for Guangdong.

economic growth in China. Besides, the intensity-based cap setting can make better adjustments for the emergence of new entrants and unexpected changes of emission reduction cost. However, as the intensity-based cap allows rapid economic growth to continue, its effectiveness on emission abatement has a higher uncertainty than using the absolute emission cap.

Compared to more mature ETS such as EU ETS, Chinese ETS is still at the trial stage. After more than four-year operation, the problems of the emission trading markets have become apparent, including low liquidity and high volatility, even though some pilots are a little better than others. China's ETSs generally had a poor performance because of the absence of legal binding forces, excessive allowance allocation, market segmentation, and lack of active investments (Tan and Wang 2017a).

The price of emission allowance is often used for analyzing an emission trading market, as it theoretically responds to the market supply and demand. Factors affecting the price of the $CO_2$ emission allowance that are commonly identified in the literature include energy prices, macroeconomic indicators, extreme temperature events and institutional events. A summary of this group of studies is shown in Table 2. In these studies, the words such as "impact", "influence", "effect" etc. mostly refer to Granger causality in the sense of inter-temporal precedence, rather than causality "in the colloquial sense of an unavoidable logical link" (Keppler and Mansanet-Bataller 2010). The empirical literature concentrates on the analysis of the EU $CO_2$ emission allowance (EUA) prices, while there are a small number of studies on the ETS in the United States (Hammoudeh, Nguyen, and Sousa, 2014a; Hammoudeh et al. 2015; Kim and Koo 2010). Literature has examined the Granger causality from crude oil price, natural gas price, and coal price to the price of $CO_2$ emission allowances, using time series techniques such as GARCH, Vector Auto-regression (VAR), Newey–West Ordinary Least Squares (NW-OLS), Autoregressive Distributed Lag (ADL), Vector Error Correction Model (VECM) and quantile regressions. Alberola, Chevallier, and Chèze (2008), Hammoudeh, Nguyen, and Sousa (2014a,b), Hammoudeh et al. (2015), and Keppler and Mansanet-Bataller (2010) also include electricity price in the analysis in addition to energy prices, assuming that changes in electricity price may affect $CO_2$ emission allowance price due to the resulting changes in the consumption of electricity, a secondary energy source. This may not happen in China since Chinese electricity price is highly regulated and the price changes are not frequent.

According to the substitution effect theory, the increase in oil price (or natural gas price) would contribute to an increase in $CO_2$ emission allowance price through a fuel substitution from oil (or natural gas) to more carbon-intensive fuels such as coal. For instance, Boersen and Scholtens (2014) found that oil price, as well as natural gas price, were positive drivers of EUA futures price during the second phase of EU ETS. Alberola, Chevallier, and Chèze

**Table 2.** Review of Empirical Studies on Factors Affecting $CO_2$ Emission Allowance Price

| Literature | ETS/ Dependent Variable | Time Period | Main Factors Examined | Method |
|---|---|---|---|---|
| Alberola, Chevallier, and Chèze (2008a,b) | EU ETS/EUA spot price | 2005.7–2007.4 | Crude oil price, natural gas price, coal price, switch price of $CO_2$ between coal and natural gas, temperature, electricity price, the compliance break, the announcement of stricter allocation | NW OLS, GARCH |
| Boersen and Scholtens (2014) | EU ETS/EUA futures price | 2008.12–2012.12 | Crude oil price, natural gas price, coal price, the switch possibility from coal to natural gas | Threshold GARCH |
| Chevallier (2011) | EU ETS/EUA futures price | 2005.1–2010.7 | Crude oil price, natural gas price, coal price, aggregated industrial production index | Markov-switching VAR |
| Hammoudeh, Nguyen, and Sousa (2014a) | EU ETS/EUA spot price | 2006.8–2013.11 | Crude oil price, natural gas price, coal price, electricity price | Bayesian Structural VAR |
| Tan and Wang (2017b) | EU ETS/EUA futures price | 2005.4–2016.1 | Crude oil price, natural gas price, coal price, macroeconomic indicators | Quantile regression |
| Creti, Jouvet, and Mignon (2012) | EU ETS/EUA futures price | 2005.6–2010.12 | Crude oil price, stock index, switch price of $CO_2$ between coal and natural gas | OLS |
| Keppler and Mansanet-Bataller (2010) | EU ETS/ EUA spot and futures price | 2005.1–2007.12 and 2008 | Natural gas price, electricity price, coal price, temperature, stock index | OLS |
| Koch et al. (2014) | EU ETS/EUA futures price | 2008.1–2013.10 | Natural gas price, coal price, macroeconomic indicators, the number of issued CERs, electricity production from wind/solar, switch price of $CO_2$ between coal and natural gas | NW-OLS |
| Kim and Koo (2010) | Chicago/ CCX[a] trading emission allowance volume | 2005.1–2008.11 | Crude oil price, natural gas price, coal price, temperature, economy crisis dummy | ADL [b] |
| Hammoudeh et al. (2015) | US ETS/ Proxied by EUA price | 2006.8–2013.11 | Crude oil price, natural gas price, coal price, electricity price | NADL[c] |
| Hammoudeh, Nguyen, and Sousa (2014b) | US ETS/ Proxied by EUA price | 2006.7–2013.11 | Crude oil price, natural gas price, coal price, electricity price | Quantile regressions |

[a] Chicago Climate Exchange (CCX); [b] Autoregressive distributed lag (ADL); [c] Nonlinear autoregressive distributed lag model (NADL).

(2008) explored the drivers of EUA spot price in two sub-periods during January 2005–April 2007 (before and after the "compliance break" in 2016), and found that oil price positively affected the EUA price in both sub-periods. However, some other studies reported a negative relationship between oil price and $CO_2$ emission allowance price when the substitution effect was not significant. Hammoudeh, Nguyen, and Sousa (2014a) found that when $CO_2$ emission allowance price was at a high level, oil prices had a substantial negative effect on $CO_2$ emission allowance price. They explained that the result might be (1) because when $CO_2$ emission allowance price was high, higher oil price might lead to a substantial drop in all energy consumption and the associated emissions, without encouraging the substitution of coal for oil, or (2) because higher oil price might raise all energy prices and encourage the use of cleaner energy resources.

Some studies revealed a negative relation between coal price and the $CO_2$ emission allowance price, which is consistent with the substitution effect theory. Hammoudeh, Nguyen, and Sousa (2014a) suggested that in the context of US during 2006–2013, coal price had a negative impact on $CO_2$ emission allowance price, as a rise in coal price could arrest coal-associated pollution. Hammoudeh et al. (2015) further stated that coal price had a negative but asymmetric impact on the $CO_2$ emission allowance price in the short term. Compared to a price increase, a price decrease of coal had a more significant impact.

Literature has also examined whether the natural gas price affects the price of the $CO_2$ emission allowance. Alberola, Chevallier, and Chèze (2008) found that the natural gas price positively impacted EUA spot price while the coal price negatively impacted EUA spot price during January 2005–April 2007. Alberola, Chevallier, and Chèze (2008) also included the switch price between natural gas and coal in the analysis, but there seemed to be a multicollinearity problem among coal price, gas price, and the switch price. Tan and Wang (2017b) summarized that during the three phases of EU ETS, the relationships between EUA price and energy prices vary from one phase to another. It is reasonable as the supply and demand curves of energy sources were changing during the three phases. Same could be said about the supply and demand of emission allowances.

Some studies examined the impact of macroeconomic indicators, such as aggregated industrial production, stock index, economy crisis dummy and so on. The macroeconomic indicators can influence $CO_2$ emission allowance price either through affecting the expectation of economic growth and the future emission allowance demand or through affecting the changes of energy price (Tan and Wang 2017b). However, only a few of the studies showed that the impact of macroeconomic indicators is significant (Chevallier 2011; Koch et al. 2014). Literature also found structural breaks of $CO_2$ emission allowance price series due to institutional decisions or events. For instance, there was often a compliance break in year

T when the regulated firms actively manage their compliance after the disclosure of verified emission and before the submission deadline of allowances valid for year T-1 (Alberola, Chevallier, and Chèze 2008).

There are limited empirical studies on the price dynamics of the piloting ETSs in China. Among a few recent empirical studies that did focus on $CO_2$ emission allowance (CEA), Zeng et al. (2017), Zhang and Zhang (2016), and Fan and Todorova (2017) examined the relationships between CEA price and energy prices. Zeng et al. (2017) employed a structural VAR approach showing that during April 2014–November 2015, coal price had a significant and positive impact on Beijing CEA price within a short period, but it became negative after two days when firms started to substitute coal with less carbon-intensive energy sources or use carbon-reduction measures. Based on a quantile regression method, Zhang and Zhang (2016) argued that oil price had a slight positive impact on the Shanghai CEA price, consistent with the substitution effect theory. Fan and Todorova (2017) investigated into the response of emission allowance price to energy prices and macroeconomic indicators in Beijing, Shenzhen, Guangdong and Hubei from the launching dates to December 2016. The results showed that Hubei CEA price was weakly related to natural gas price, while Guangdong CEA price had a significant positive relation with oil price. However, overall, the empirical studies on the influence from energy prices to CEA prices have been scarce and scattered.

Given that there are still few empirical studies on the prices of China's CEA and considering that ETS will continue to play an important role in reducing $CO_2$ emissions in China, this study aims at examining the price dynamics of CEA, with a focus on the relations between energy markets and ETS markets. The rest of the article is organized as follows. Section 2 details the data and method. Section 3 starts with the descriptive analysis of the CEA price and trading volume data, following by co-integration tests and multivariate regressions to examine the relationship between energy prices and CEA price. Section 4 presents concluding remarks and policy implications of the findings.

## 2. Data and Method

### *2.1. Data*

We collected the daily CEA price data and daily transaction volume data from the website (www.tanpaifang.com) that compiles the market data published by Emission Exchanges of the seven ETS pilots. The website was created in 2012, organized by the Zhongke Carbon Information Technology Research Institute, providing data, regulatory information and consultancy about ETS. As Chinese ETS allows for only spot trading, all price data collected are closing spot trading price data. The market data of China's seven piloting ETSs can also be found from the website (www.chinacarbon.net.cn) organized by Climate Limited, which is a UN-accredited online media company. Data from the two sources are identical. This study collected the daily market data of the seven ETS pilots from their launching dates to 30 June 2017, considering that in the second half of year 2017, all ETS pilots were making adaptions since the country was going to establish the national ETS by the end of the year. The currency unit of all price data is changed from RMB to US dollar ($), using the currency conversion rates provided by OECD.

In most local ETSs, the daily CEA data, either price or trading volume, are missing for many days, which impede the ability to do the analysis at the daily frequency. Similar to Fan and Todorova (2017), we used the weekly price data ($Price_t$, \$/ton $CO_2$e) for analysis. For instance, given any week T, if the daily CEA price data is available for n days (n≤7) of the week, the weekly price data will be calculated by averaging the daily price data: $Price_T = (\sum_1^n \mathrm{Price_i})/n$. Also, we generated the weekly trading volume data ($Volume_t$, 1000 ton $CO_2$e/week) by totaling the daily trading volume in the week. Given any week T, if the daily trading volume data is available for n days (n≤7) of the week, the weekly trading volume data will be calculated by $Volume_T = \sum_1^n \mathrm{Volume_i}$. Thus, in week T, we can think it as the $Volume_T$ amount of CEA was traded in the week at the $Price_T$. Further, a time-series variable of price return is generated for each price variable for analysis using the equation $\ln(Price_t/Price_{t-1})$.

On energy markets, we collected data on the oil price, coal price, and natural gas price, in order to address the relations between energy prices and CEA prices. Coal consumption is one of the major sources of $CO_2$ emission in China, and the consumption of crude oil or natural gas is much less. We expect that the demand for CEA will increase due to substitution effect if the price of less carbon-intensive fuel (e.g. oil) increases, or if the price of more carbon-intensive fuel (e.g. coal) decreases. Given there is no coal price index for each pilot region, the coal price we use is the Bohai Rim 5500 kcal/kg stream coal spot price ($Coal_t$, RMB/ton, weekly), which was also used by Fan and Todorova (2017). It is a most important benchmark domestic coal price index in China. It is based on the average price of 5500 kcal coal at Qinhuangdao, Tianjin, Caofeidian, Jingtang, Huanghua and Guotoujingtan ports. The data is published by Qinhuangdao Maritime Coal Market Co., Ltd and can be collected from the website www.coalchina.org.cn. We collected natural gas price data from CEIC database, which is monthly average spot price data (RMB/ton) of Liquefied Natural Gas (LNG) released by China Petroleum and Chemical Industry Federation. We converted the monthly data to a weekly price variable of LNG ($LNG_t$, RMB/ton, weekly) by assuming that different weeks during the same month have the same price. The oil price ($Brent_t$, \$/barrel, weekly) is the weekly Europe spot price of Brent crude oil and petroleum products, collected by the website of US Energy Information Administration. Chinese oil import dependence is larger than 60%, so we use the Brent crude oil price to reflect the influence of the international price shocks. $Coal_t$ and

$Brent_t$ are collected from 2013 Week 25 when the first piloting ETS started to operate in Shenzhen. Available observations of $LNG_t$ only started from 1st January 2014. Data are all collected till 30th June 2017. The currency unit is all converted to US dollar ($) using OECD currency conversion rates.

Further, we include the Shanghai Shenzhen 300 stock index data (31st December 2014=1000, daily) to capture the relation between the macroeconomic level and the CEA price. The data was collected from CEIC database for the time period of 2013 Week 25–2017 Week 26 and converted to weekly data ($Stock_t$, weekly) by averaging the daily stock index data. The macroeconomic level can affect the demand and supply of CEA through affecting the production activities and the associated $CO_2$ emissions. Thus, we expect that CEA prices increase when $Stock_t$ increases.

We also add three dummy variables to test the structural changes of the regressions. They are *Break2014*, *Break2015*, and *Break2016*, respectively taking a value of 1 from 2014 Week 27, 2015 Week 27 and 2016 Week 27. So there are four sub-periods of the dataset, launching dates -June 2014 (1st Period), July 2014–June 2015 (2nd Period), July 2015–June 2016 (3rd Period), and July 2016–June 2017 (4th Period). The compliances of regulated enterprises in any local ETS follow a particular calendar. At the beginning of year T in each ETS pilot, the regulated enterprises receive their CEA allocations for year T. The regulated enterprises have to submit their emission reports to the regulator in March of year T or the end of February of year T, with dates varying over pilots, and then a third party will verify their reports. In April, the regulated enterprises should submit the verified emission

**Table 3.** Test for Structural Breaks of $\ln Price_t$

| ETS | Full Sample Period | | AR Lags | Break2014 | Break2015 | Break2016 |
|---|---|---|---|---|---|---|
| | From Launching Date | To | | (2014 Week 26) | (2015 Week 26) | (2016 Week 26) |
| | | | | Chi Statistic | Chi Statistic | Chi Statistic |
| BJ | 28-Nov-2013 | 30-Jun-2017 | 3 | 4.275 | 16.827*** | 19.453*** |
| CQ | 19-Jun-2014 | 30-Jun-2017 | 2 | — | 2.577 | 2.920 |
| SH | 26-Nov-2013 | 30-Jun-2017 | 2 | 2.237 | 10.251** | 8.198** |
| SZ | 18-Jun-2013 | 30-Jun-2017 | 3 | 41.766*** | 18.991*** | 5.935 |
| TJ | 26-Dec-2013 | 30-Jun-2017 | 2 | 9.561** | 2.541 | 24.578*** |
| GD | 16-Dec-2013 | 30-Jun-2017 | 3 | 9.699** | 11.305** | 3.419 |
| HB | 02-Apr-2014 | 30-Jun-2017 | 1 | 0.021 | 5.311* | 5.175* |

Note: $lnPrice_t$ refers to the logarithmic form of CEA price. ***, ** and * denote significance at 1%, 5% and 10% levels. "BJ" is short for Beijing ETS, "CQ" for Chongqing ETS, "GD" for Guangdong ETS, "HB" for Hubei ETS, "SH" for Shanghai ETS, "SZ" for Shenzhen ETS, and "TJ" for Tianjin ETS. "—" denotes the statistics are not available due to the characteristics of the original data.

reports, and at the end of June (around week 26 of the year), they have to submit the allowances valid during year T-1 to comply with their targets of year T-1. So, the trading of emission allowances is relatively active between April and June of year T. We ran an auto-regression (AR) model of logarithmic CEA price data and used a Wald test to see if there is a structural break of the coefficients after the week 26 of each year. The results in Table 3 show that the AR coefficients of $lnPrice_t$ do have structural changes either after 2014 Week 26, or after 2015 Week 26, or after 2016 Week 26, in all ETS pilots except Chongqing. Chongqing ETS has very low-level trading activities and its CEA price might be artificially set at a certain level during weeks when trading volumes were zero, not reflecting the market demand and supply. Thus, it is not surprising that Chongqing CEA price does not exhibit compliance break as other piloting ETSs.

## *2.2. Methods*

This study uses time-series analysis techniques. The Akaike Information Criterion (AIC) is used for choosing the number of lags in all tests. Augmented Dickey-Fuller (ADF) tests are applied to check the unit roots of variables. This study performed three forms of ADF tests: random walk model with drift, random walk model with a trend, and pure simple random walk model. The Philips-Perron (PP) test is performed with and without trend as a robustness check.

To investigate the relationships between energy prices and CEA prices, this study first uses Johansen's test to see if there are long-run equilibrium relations between energy prices and CEA prices. Second, it performed time-series regressions using the technique of multivariate OLS or NW-OLS which works with stationary data series. If there is a significant serial correlation problem, the Newey-West (NW) heteroscedasticity-and-autocorrelation-consistent estimator will be used. If not, the regressions will be estimated with the robust estimator. Three types of post-estimation tests were performed: Durbin's alternative test for serial correlation, the Breush-Godfrey serial correlation Lagrange Multiplier test, and the joint F-test.

Therefore, the role played by Brent crude oil price and coal price on CEA price is estimated using the first specification (Eq.1):

$$\text{lnReturn}_{i,t} = a_{i,0} + a_{i,1}(LD)lnReturn_{i,t} + a_{i,2}(LD)\text{ln}Brent_t + a_{i,3}(LD)lnCoal_t + a_{i,4}Break2014 + a_{i,5}Break2015 + a_{i,6}Break2016 + \mu_{i,t}. \tag{1}$$

The second specification (Eq.2) takes account of stock index:

$$\text{lnReturn}_{i,t} = a_{i,0} + a_{i,1}(LD)lnReturn_{i,t} + a_{i,2}(LD)\text{ln}Brent_t + a_{i,3}(LD)lnCoal_t + a_{i,7}(LD)lnStock_t + a_{i,4}Break2014 + a_{i,5}Break2015 + a_{i,6}Break2016 + \mu_{i,t}. \tag{2}$$

The third specification (Eq.3) adds the price data of LNG, which is only available since 2014 week 1.

$$\text{lnReturn}_{i,t} = a_{i,0} + a_{i,1}(LD)lnReturn_{i,t} + a_{i,2}(LD)\text{ln}Brent_t + a_{i,3}(LD)lnCoal_t + a_{i,7}(LD)lnLNG_t + a_{i,8}(LD)lnStock_t + a_{i,4}Break2014 + a_{i,5}Break2015 + a_{i,6}Break2016 + \mu_{i,t}. \tag{3}$$

In the equation, $lnReturn_{i,t}$ is the price return of CEA at period $t$ in the ETS pilot $i$. Price return series are used because they are all stationary. $L$ denotes the lag operator. $D$ denotes the first difference. $\mu_{i,t}$ is the error term. Eq.3 is the main equation we want to estimate, while Eq.1 and Eq.2 are the restricted equations. The equations are adjusted from ADL model. Coefficients on lagged log differences of a variable reflect the short-run effects, while the joint F-test on all values of a variable tells the Granger causality.

## 3. Empirical Analysis

### *3.1. Descriptive Analysis*

Table 4 displays the descriptive statistics of CEA price, price returns and trading volumes. Beijing and Shenzhen ETSs had the highest CEA prices on average, which were 7.805 and 6.911 \$/ton $CO_2$e respectively. Shenzhen CEA price is highly skewed, with a large right-handed tail, meaning that the majority of the price data is lower than the average price. The distribution of Beijing CEA price, on the other hand, is both positively skewed and leptokurtic. The largest CEA price happened at Shenzhen which was 18.360 \$/ton $CO_2$e in week 42 of 2013. Chongqing ETS had the lowest average CEA price, 3.32 \$/ton $CO_2$e. Based on a two-region dynamic CGE model, Wang et al. (2015) suggested that the carbon price should be about 38\$ $CO_2$e for reaching the Copenhagen target of 40%–50% reduction of $CO_2$ emission intensity toward 2020 in relative to 2005 level. However, even in Shenzhen, its highest carbon price was less than 20\$/ton $CO_2$e, far from the ideal price suggested by Wang et al. (2015). For Chongqing ETS, its price once decreased to 0.163 \$/ton $CO_2$e, which provided very poor incentives for emitters to make environmental changes.

In most piloting ETSs, CEA prices were at a high level in the initial operation stage and exhibited a general decrease trend over time, which may explain the positive skewness in most cases (see Table 4). Take Shenzhen ETS as an example. Its carbon prices were at a level of more than 10 \$/ton $CO_2$e during June 2013 to June 2014. Then, the CEA price gradually decreased to about 5 \$/ton $CO_2$e by the end of 2014. From January 2015 to June 2017, the CEA price reached a relatively stable status, ranging between 3 and 8 \$/ton$CO_2$e. Hubei CEA prices decreased from about 4 \$/ton $CO_2$e in 2014 to less than 3\$/ton $CO_2$e in 2017, but the distribution showed a slight negative skewness. Chongqing CEA price, however, had a peak in the end of 2016 due to stricter CEA allocation, but it decreased to less than 1\$/ton $CO_2$e in the second quarter of 2017.

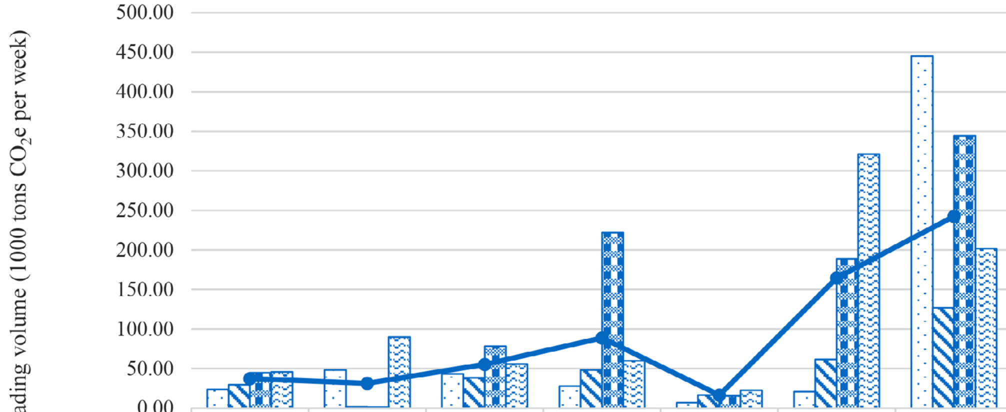
We can see the descriptive statistics of the spot trading volumes from Table 4 and Figure 1. $Volume_t$ denotes the total trading volume of CEA in each week. It was quite different across ETS pilots, ranging from 16422 tons $CO_2$e/week in Chongqing ETS to 242541 tons $CO_2$e/week in Guangdong ETS. The two provincial level ETSs, Guangdong, and Hubei, obviously have the higher amount of weekly trading volumes than the five city-level ETSs. But, the maximum trading volume happened

**Table 4.** Descriptive Statistics of CEA Price, Price Returns and Trading Volumes

| Market | | City Level ETS | | | | | Provincial Level ETS | |
|---|---|---|---|---|---|---|---|---|
| | | BJ | CQ | SH | SZ | TJ | GD | HB |
| $Price_t$ (\$/ton $CO_2e$) | Obs. | 187 | 159 | 184 | 210 | 183 | 184 | 170 |
| | Mean | 7.805 | 3.133 | 3.894 | 6.911 | 3.424 | 4.150 | 3.326 |
| | Std. Dev. | 1.091 | 1.726 | 2.114 | 3.048 | 1.156 | 3.019 | 0.718 |
| | Min. | 5.343 | 0.163 | 0.676 | 3.102 | 1.054 | 1.295 | 1.582 |
| | Max. | 12.507 | 7.152 | 7.814 | 18.360 | 7.421 | 12.053 | 4.399 |
| | Skewness | 0.587 | 0.131 | 0.032 | 1.199 | 0.468 | 1.279 | −0.457 |
| | Kurtosis | 5.786 | 1.793 | 1.640 | 3.404 | 3.355 | 3.091 | 1.666 |
| $lnReturn_t$ | Obs. | 186 | 158 | 183 | 209 | 182 | 183 | 169 |
| | Mean | 0.000 | −0.020 | 0.001 | 0.000 | −0.006 | −0.008 | −0.004 |
| | Std. Dev. | 0.060 | 0.198 | 0.095 | 0.113 | 0.100 | 0.098 | 0.051 |
| | Min. | −0.210 | −0.931 | −0.355 | −0.346 | −0.544 | −0.314 | −0.259 |
| | Max. | 0.225 | 0.814 | 0.320 | 0.530 | 0.547 | 0.303 | 0.225 |
| | Skewness | 0.363 | −0.424 | 0.024 | 0.585 | 0.018 | −0.004 | −0.701 |
| | Kurtosis | 6.447 | 11.558 | 6.398 | 6.169 | 13.522 | 4.462 | 10.513 |
| $Volume_t$ (1000 ton $CO_2e$ per week) | Obs. | 187 | 159 | 184 | 210 | 183 | 184 | 170 |
| | Mean | 37.238 | 31.296 | 55.309 | 88.875 | 16.422 | 164.625 | 242.541 |
| | Std. Dev. | 84.351 | 185.844 | 137.582 | 380.085 | 105.425 | 336.122 | 380.081 |
| | Min. | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | 25th percentile | 0.620 | 0.000 | 0.053 | 3.640 | 0.000 | 0.770 | 64.343 |
| | Median | 5.000 | 0.000 | 12.447 | 16.904 | 0.360 | 23.540 | 133.683 |
| | 75th percentile | 25.523 | 0.144 | 48.594 | 68.291 | 1.380 | 174.190 | 258.081 |
| | Max. | 565.091 | 2166.620 | 1113.833 | 4010.471 | 1083.470 | 2449.509 | 3217.207 |
| | Skewness | 3.489 | 9.954 | 4.797 | 9.408 | 8.280 | 3.630 | 4.773 |
| | Kurtosis | 16.516 | 111.895 | 29.626 | 93.829 | 74.976 | 19.120 | 31.790 |
| | Number of zero observations | 3 | 111 | 43 | 9 | 60 | 19 | 2 |

Note: "*Std. Dev.*" indicates standard deviation; "*Obs*" indicates the number of observations. "*Min*" and "*Max*" are short for minimum and maximum respectively. $Price_t$ refers to a time-series variable of weekly CEA prices. $Volume_t$ refers to a time-series variable of weekly CEA trading volume. $lnReturn_t$ refers to the logarithmic form of CEA price returns. $lnReturn_t = \ln(Price_t/Price_{t-1})$.

at Shenzhen ETS, when 4 million tons $CO_2e$ were traded in 2016 week 12. Among the city-level ETSs, Shenzhen has the largest weekly trading volume on average, followed by Shanghai and Beijing. In Chongqing and Tianjin, however, there are respectively 111 weeks and 60 weeks when zero transactions were made on the ETS markets, implying very low-level participation of the regulated enterprises. It may reflect deeper problems of the two ETSs: loose enforcement and over-supply of CEA.




| | BJ | CQ | SH | SZ | TJ | GD | HB |
|---|---|---|---|---|---|---|---|
| 1st Period | 23.50 | 48.33 | 43.40 | 27.90 | 6.83 | 20.89 | 444.86 |
| 2nd Period | 29.71 | 1.74 | 38.36 | 48.35 | 16.37 | 61.61 | 127.00 |
| 3rd Period | 44.56 | 1.20 | 78.32 | 221.71 | 15.54 | 188.47 | 344.57 |
| 4th Period | 45.63 | 89.97 | 55.66 | 59.89 | 22.34 | 321.20 | 201.58 |
| Full sample | 37.24 | 31.30 | 55.31 | 88.87 | 16.42 | 164.63 | 242.54 |

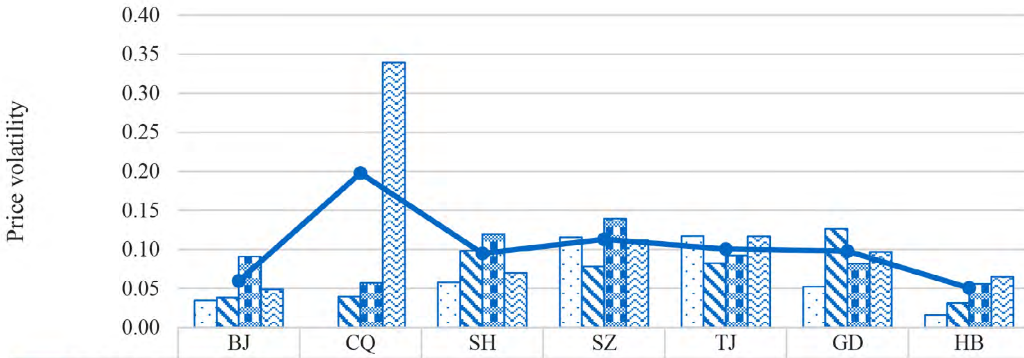
Note: Full sample period started from the first operation dates of the ETS pilots till 30 June 2017.

**Figure 1.** Average Weekly Trading Volume of CEA in Seven ETS Pilots

In each ETS, the trading activities are more frequent and involve larger trading volumes toward the compliance deadline. In most time of the year, the trading activities are relatively sporadic and with smaller trading volumes. Thus, the trading volume variables all exhibit the asymmetric and leptokurtic distribution, with high-level positive skewness and pronounced excess kurtosis.

The standard deviation of CEA price returns ($lnReturn_t$) can be considered as a measure of price volatility. We use it to compare the market risks between local ETSs. Figure 2 shows the price volatility over different time periods, launching dates -June 2014 (1st Period), July 2014–June 2015 (2nd Period), July 2015–June 2016 (3rd Period), and July 2016–June 2017 (4th Period). We can see that each ETS pilot exhibits a variation of price volatility in different sub-periods. Although Hubei ETS had low CEA price on average, its price volatility was the lowest compared to other piloting ETSs, but it became more and more volatile over time. Beijing ETS had a relatively low volatility but experienced a higher level of volatility during July 2015–June 2016. Chongqing CEA price had the highest volatility, and it was the most volatile during July 2016–June 2017.

We can see that different local ETSs perform so differently that the operation of the national ETS can be expected to be more challenging. Overall, Beijing and Shenzhen ETSs seem




| | BJ | CQ | SH | SZ | TJ | GD | HB |
|---|---|---|---|---|---|---|---|
| 1st Period | 0.03 | | 0.06 | 0.12 | 0.12 | 0.05 | 0.02 |
| 2nd Period | 0.04 | 0.04 | 0.10 | 0.08 | 0.08 | 0.13 | 0.03 |
| 3rd Period | 0.09 | 0.06 | 0.12 | 0.14 | 0.09 | 0.08 | 0.06 |
| 4th Period | 0.05 | 0.34 | 0.07 | 0.11 | 0.12 | 0.10 | 0.06 |
| Full sample | 0.06 | 0.20 | 0.09 | 0.11 | 0.10 | 0.10 | 0.05 |

Note: price volatility is estimated by the standard deviation of price returns.

**Figure 2.** Price Volatility of CEA in Seven ETS Pilots

to operate better than other city-level ETSs, with their high-level CEA prices and active trading activities. Regarding provincial level ETS, Guangdong has higher CEA price, while Hubei has lower volatility and larger weekly trading volume. Chongqing and Tianjin ETSs are not so successful considering their low CEA prices and scarce trading transactions. Therefore, we do not include Chongqing and Tianjin in the regression analysis, as their CEA prices may be artificially set in the weeks with zero transactions, not really relating to the market demand and supply.

Table 5 contains the descriptive statistics of crude oil price, coal price and Shanghai Shenzhen 300 stock index during 2013 week 25–2017 week 26, and LNG price data during 2014 week 1–2017 week 26. Shanghai Shenzhen 300 stock index has the largest volatility because of the nature of the stock index data.

### *3.2. Unit Root Tests and Johansen's Tests*

We took logarithmic forms of all price variables and tested their unit roots. Table 6 displays the results. Not all of the logarithmic forms of the variables is integrated of order zero, meaning that the logarithmic forms of some variables are not stationary. The logarithmic differences of all CEA price variables (i.e. price returns) are stationary, and the logarithmic differences of energy prices and the stock index are also stationary. Thus, we use logarithmic differences of the price variables throughout the following regression analysis.

We applied Johansen's tests for *I(1)* logarithmic price variables that are integrated of order one, including Chongqing CEA price, Hubei CEA price, Shanghai CEA price, Brent oil price and LNG price. We used the tests for every two *I(1)* logarithmic variables in order to see whether they have long-run equilibrium relationships. However, we can see from Table 7 that there is no significant long-run equilibrium relationships between the variables.

**Table 5.** Descriptive Statistics of Energy Prices and Stock Index

| Var. | | Coal Price ($Coal_t$) | Brent Oil Price ($Brent_t$) | LNG Price ($LNG_t$) | Shanghai Shenzhen 300 Stock Index ($Stock_t$) |
|---|---|---|---|---|---|
| Full sample | From | 2013 Week 25 | 2013 Week 25 | 2014 Week 1 | 2013 Week 25 |
| | To | 2017 Week 26 | 2017 Week 26 | 2017 Week 26 | 2017 Week 26 |
| Unit | | $/ton | $/barrel | $/ton | Dimensionless |
| Obs. | | 210 | 210 | 182 | 210 |
| Mean | | 78.123 | 69.378 | 586.724 | 3096.085 |
| Std. Dev. | | 12.421 | 28.166 | 129.759 | 704.039 |
| Min. | | 55.840 | 27.760 | 428.959 | 2116.750 |
| Max. | | 101.840 | 116.030 | 804.167 | 5324.406 |
| Skewness | | −0.380 | 0.493 | 0.215 | 0.508 |
| Kurtosis | | 1.791 | 1.550 | 1.474 | 3.111 |

**Table 6.** Test for Unit Roots

| Price Returns | Stationary? | Obs. | ADF, Simple | PP, Simple | ADF, with Trend | PP, with Trend | ADF, with Drift | Integ. Order |
|---|---|---|---|---|---|---|---|---|
| | | | Z(t) | Z(t) | Z(t) | Z(t) | Z(t) | |
| BJ: CEA | Yes | 183 | −9.249*** | −14.191*** | −9.224*** | −14.151*** | −9.249*** | I(0) |
| CQ: CEA | Yes | 154 | −7.366*** | −7.660*** | −7.421*** | −7.677*** | −7.366*** | I(1) |
| GD: CEA | Yes | 179 | −9.554*** | −12.210*** | −9.616*** | −12.218*** | −9.554*** | I(0) |
| HB: CEA | Yes | 165 | −10.173*** | −13.940*** | −10.174*** | −13.928*** | −10.173*** | I(1) |
| SH: CEA | Yes | 179 | −4.577*** | −10.568*** | −4.680*** | −10.604*** | −4.577*** | I(1) |
| SZ: CEA | Yes | 205 | −9.581*** | −19.583*** | −9.667*** | −19.647*** | −9.581*** | I(0) |
| TJ: CEA | Yes | 178 | −8.455*** | −9.646*** | −8.444*** | −9.632*** | −8.455*** | I(0) |
| Coal | Yes | 205 | −4.610*** | −7.406*** | −4.729*** | −7.625*** | −4.610*** | I(0) |
| Brent Oil | Yes | 205 | −7.433*** | −10.641*** | −7.436*** | −10.621*** | −7.433*** | I(1) |
| LNG | Yes | 177 | −5.450*** | −13.144*** | −5.432*** | −13.120*** | −5.450*** | I(1) |
| shsz300 | Yes | 205 | −5.682*** | −11.200*** | −5.670*** | −11.184*** | −5.682*** | I(0) |

Note: "Z(t)" refers to the statistic of ADF or PP unit root test. ***, ** and * denote significance at 1%, 5% and 10% levels. "Integ. Order" refers to order of integration.

### *3.3. Multivariate Regressions for Provincial ETSs*

Table 8 displays the regression results for the two provincial level ETSs, Guangdong ETS and Hubei ETS. The dependent variables are the CEA price returns ($lnReturn_t$) of Guangdong and Hubei respectively, while the independent variables are the logarithmic dif-

**Table 7.** Johansen's Test for Co-Integration Ranks

| Hypotheses | | H0: Rank=0<br>H1: Rank=1 | H0: Rank=1<br>H1: Rank=2 |
|---|---|---|---|
| Logarithmic price variables | CQ CEA & Brent Oil | 6.687 | 1.255 |
| | HB CEA & Brent Oil | 4.380 | 1.176 |
| | SH CEA & Brent Oil | 8.859 | 1.715 |
| | CQ CEA & LNG | 4.849 | 0.134 |
| | HB CEA & LNG | 9.642 | 1.017 |
| | SH CEA & LNG | 3.351 | 1.491 |
| 5% critical value | | 15.410 | 3.760 |

Note: ** denote significance at 5% level. Logarithmic prices were used for the Johansen's co-integration tests.

**Table 8.** Multivariate Regressions for Provincial Level ETSs

| | Guangdong ETS | | | Hubei ETS | | |
|---|---|---|---|---|---|---|
| | Eq.1 | Eq.2 | Eq.3 | Eq.1 | Eq.2 | Eq.3 |
| L1. $\ln Return_t$ | 0.089 | 0.082 | 0.090 | −0.101 | −0.102 | −0.101 |
| | (0.080) | (0.079) | (0.082) | (0.207) | (0.210) | (0.212) |
| L2.$\ln Return_t$ | −0.265*** | −0.269*** | −0.267*** | −0.106 | −0.098 | −0.096 |
| | (0.080) | (0.081) | (0.081) | (0.137) | (0.139) | (0.141) |
| L1D.$\ln Brent_t$ | 0.101 | 0.096 | 0.120 | 0.068 | 0.078 | 0.084 |
| | (0.158) | (0.154) | (0.160) | (0.095) | (0.101) | (0.092) |
| L2D.$\ln Brent_t$ | −0.058 | −0.061 | −0.067 | −0.074 | −0.075 | −0.071 |
| | (0.164) | (0.165) | (0.166) | (0.081) | (0.082) | (0.084) |
| L1D.$\ln Coal_t$ | −1.433* | −1.433* | −1.545* | 0.313 | 0.311 | 0.343 |
| | (0.786) | (0.810) | (0.838) | (0.337) | (0.348) | (0.348) |
| L2D.$\ln Coal_t$ | 0.925 | 0.864 | 0.939 | −0.116 | −0.112 | −0.143 |
| | (0.584) | (0.588) | (0.613) | (0.351) | (0.353) | (0.348) |
| L1D.$\ln LNG_t$ | | | 0.548 | | | −0.111 |
| | | | (0.487) | | | (0.199) |
| L2D.$\ln LNG_t$ | | | −0.031 | | | 0.161 |
| | | | (0.302) | | | (0.246) |
| L1D.$\ln Stock_t$ | | 0.181 | 0.135 | | 0.050 | 0.061 |
| | | (0.253) | (0.244) | | (0.110) | (0.116) |
| L2D.$\ln Stock_t$ | | 0.082 | 0.092 | | −0.220 | −0.234 |
| | | (0.199) | (0.203) | | (0.148) | (0.146) |
| Break2014 | −0.029 | −0.034* | −0.033 | 0.010 | 0.012 | 0.012 |
| | (0.020) | (0.019) | (0.020) | (0.008) | (0.009) | (0.008) |

| | | | | | | |
|---|---|---|---|---|---|---|
| Break2015 | 0.018 | 0.025 | 0.026 | −0.013 | −0.017* | −0.017* |
| | (0.019) | (0.018) | (0.018) | (0.009) | (0.009) | (0.009) |
| Break2016 | 0.026 | 0.023 | 0.020 | 0.005 | 0.007 | 0.007 |
| | (0.019) | (0.019) | (0.019) | (0.012) | (0.012) | (0.012) |
| Constant | −0.003 | −0.002 | −0.001 | −0.008 | −0.007 | −0.007 |
| | (0.013) | (0.013) | (0.015) | (0.005) | (0.005) | (0.006) |
| Obs. | 181 | 181 | 179 | 167 | 167 | 167 |
| R-squ. | 0.117 | 0.121 | 0.132 | 0.034 | 0.050 | 0.055 |
| F-stat. | 2.150 | 1.900 | 1.660 | 0.510 | 0.640 | 0.870 |
| Prob>F | 0.028 | 0.043 | 0.074 | 0.863 | 0.796 | 0.588 |
| Durbin's alternative test | 2.190 | 1.954 | 1.676 | 0.001 | 0.285 | 0.523 |
| BG LM test | 2.302 | 2.081 | 1.811 | 0.001 | 0.309 | 0.572 |
| Procedure | OLS, robust | OLS, robust | OLS, robust | OLS, robust | OLS, robust | OLS, robust |
| Joint F-stat for $D.lnBrent_t$ | 0.240 | 0.240 | 0.330 | 0.540 | 0.580 | 0.630 |
| Joint F-stat for $D.lnCoal_t$ | 2.040 | 1.850 | 1.940 | 0.470 | 0.440 | 0.510 |
| Joint F-stat for $D.lnLNG_t$ | | | 0.650 | | | 0.400 |
| Joint F-stat for $D.lnStock_t$ | | 0.370 | 0.300 | | 1.140 | 1.350 |

Note: The dependent variable is the price returns (*$lnReturn_t$*) of CEA in the ETS. Durbin's alternative test and Breusch–Godfrey (BG) LM test are the tests for autocorrelation. "*Joint F-stat*" refers to the F statistic of the joint significance test on the lags of log differences of energy variables and the stock index variable. ***, ** and * denote significance at 1%, 5% and 10% levels. Standard errors are in parentheses.

ferences of energy prices and the stock index as well as the structural break dummies. The regressions are all estimated using the robust estimator rather than Newey-West estimator as there is no significant serial correlation problem. The last four rows of Table 8 show the results of the joint F-tests, which test the joint significance of coefficients on all lag values of a variable.

We can see that the one week lagged coal price return (*$D.lnCoal_t$*) had a significantly negative effect on Guangdong CEA price returns. It means that the increase of coal price led to the decrease of Guangdong CEA price in short term, which is consistent with the theory that when coal price is high, the industrial enterprises substitute coal with less carbon-intensive fuels, reducing CEA demand. But, the short-term effect became insignificant after two weeks. The joint F-test indicates that the changes in coal price do not significant-

**Table 9.** Multivariate Regressions for City Level ETSs

| | Beijing ETS | | | Shanghai ETS | | | Shenzhen ETS | | |
|---|---|---|---|---|---|---|---|---|---|
| | Eq.1 | Eq.2 | Eq.3 | Eq.1 | Eq.2 | Eq.3 | Eq.1 | Eq.2 | Eq.3 |
| L1. $\ln Return_t$ | −0.060 | −0.060 | −0.055 | 0.220* | 0.224* | 0.227* | −0.354*** | −0.367*** | −0.544*** |
| | (0.082) | (0.083) | (0.084) | (0.127) | (0.132) | (0.130) | (0.086) | (0.087) | (0.069) |
| L2.$\ln Return_t$ | −0.217** | −0.194* | −0.207** | −0.145 | −0.145 | −0.128 | −0.176** | −0.192*** | −0.260*** |
| | (0.099) | (0.104) | (0.104) | (0.098) | (0.100) | (0.095) | (0.068) | (0.069) | (0.072) |
| L1D.$\ln Brent_t$ | 0.268** | 0.253** | 0.283** | 0.073 | 0.070 | 0.064 | 0.129 | 0.104 | 0.009 |
| | (0.116) | (0.113) | (0.119) | (0.171) | (0.173) | (0.180) | (0.270) | (0.249) | (0.256) |
| L2D.$\ln Brent_t$ | −0.189 | −0.194 | −0.196 | −0.058 | −0.055 | −0.076 | −0.100 | −0.110 | −0.133 |
| | (0.122) | (0.118) | (0.120) | (0.171) | (0.171) | (0.166) | (0.216) | (0.213) | (0.198) |
| L1D.$\ln Coal_t$ | −0.162 | −0.173 | −0.187 | −0.872** | −0.876** | −0.967** | −0.531 | −0.564 | −0.428 |
| | (0.269) | (0.257) | (0.282) | (0.424) | (0.425) | (0.451) | (0.500) | (0.482) | (0.559) |
| L2D.$\ln Coal_t$ | 0.554 | 0.523 | 0.546 | 0.290 | 0.315 | 0.390 | −0.780 | −0.863 | −0.635 |
| | (0.405) | (0.356) | (0.388) | (0.542) | (0.546) | (0.558) | (0.687) | (0.709) | (0.830) |
| L1D.$\ln LNG_t$ | | | 0.097 | | | 0.748** | | | −0.200 |
| | | | (0.192) | | | (0.379) | | | (0.386) |
| L2D.$\ln LNG_t$ | | | 0.295 | | | −0.649 | | | 0.083 |
| | | | (0.189) | | | (0.450) | | | (0.416) |
| L1D.$\ln Stock_t$ | | 0.080 | 0.070 | | −0.104 | −0.180 | | 0.207 | 0.054 |
| | | (0.150) | (0.159) | | (0.256) | (0.267) | | (0.251) | (0.279) |
| L2D.$\ln Stock_t$ | | 0.271* | 0.235 | | 0.037 | 0.106 | | 0.499* | 0.590** |
| | | (0.160) | (0.162) | | (0.272) | (0.287) | | (0.267) | (0.269) |
| Break2014 | −0.021** | −0.028*** | −0.029*** | −0.037** | −0.036* | −0.035* | −0.048** | −0.062*** | −0.032* |
| | (0.010) | (0.010) | (0.010) | (0.018) | (0.019) | (0.020) | (0.020) | (0.021) | (0.017) |
| Break2015 | 0.013 | 0.022 | 0.022 | 0.009 | 0.007 | 0.008 | 0.023 | 0.041* | 0.042** |
| | (0.013) | (0.014) | (0.013) | (0.022) | (0.024) | (0.024) | (0.020) | (0.021) | (0.021) |
| Break2016 | −0.006 | −0.010 | −0.012 | 0.043** | 0.043* | 0.042* | 0.006 | −0.001 | −0.004 |
| | (0.014) | (0.014) | (0.015) | (0.021) | (0.022) | (0.022) | (0.023) | (0.022) | (0.023) |
| Constant | 0.012 | 0.013* | 0.017** | 0.015 | 0.015 | 0.014 | 0.022 | 0.023 | −0.009 |
| | (0.007) | (0.007) | (0.008) | (0.012) | (0.013) | (0.014) | (0.017) | (0.017) | (0.013) |
| Obs. | 184 | 184 | 179 | 181 | 181 | 179 | 207 | 207 | 179 |
| R-squ. | 0.128 | 0.150 | 0.163 | 0.113 | 0.114 | 0.145 | 0.137 | 0.159 | 0.270 |

| | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|
| F-stat. | 1.680 | 1.860 | 1.780 | 1.810 | 1.510 | 1.680 | 3.400 | 3.110 | 5.700 |
| Prob>F | 0.098 | 0.048 | 0.049 | 0.069 | 0.131 | 0.070 | 0.001 | 0.001 | 0.000 |
| Durbin's alternative test | 0.736 | 0.769 | 0.888 | 0.257 | 0.126 | 0.137 | 1.083 | 0.021 | 0.371 |
| BG LM test | 0.780 | 0.824 | 0.964 | 0.273 | 0.136 | 0.149 | 1.138 | 0.022 | 0.404 |
| Procedure | OLS, robust | OLS, robust | OLS, robust | OLS, robust | OLS, robust | OLS, robust | OLS, robust | OLS, robust | OLS, robust |
| Joint F-stat for $D.lnBrent_t$ | 3.140** | 3.150** | 3.460** | 0.180 | 0.100 | 0.130 | 0.140 | 0.150 | 0.310 |
| Joint F-stat for $D.lnCoal_t$ | 0.950 | 1.080 | 0.990 | 2.340 | 2.300 | 2.480* | 2.390* | 2.830* | 1.150 |
| Joint F-stat for $D.lnLNG_t$ | | | 1.260 | | | 0.230 | | | 0.180 |
| Joint F-stat for $D.lnStock_t$ | | 2.720* | 1.930 | | 0.090 | 2.800* | | 2.030 | 2.410* |

Note: The dependent variable is the price returns (*lnReturn*$_t$) of CEA in the ETS. *, ** and *** denote significance at 1%, 5% and 10% levels. Standard errors are in parentheses.

ly granger cause the changes in CEA price of Guangdong. In Hubei, however, we found no significant short-term effects or Granger causality between energy prices and CEA prices.

The results also show that there was a slight decrease of Guangdong CEA price after June 2014 and a slight decrease of Hubei CEA price after June 2015. Guangdong ETS started to operate from the end of 2013, whereas Hubei ETS was established in April 2014. So, both ETSs had higher CEA prices at the initial operation stage, but the CEA prices decreased after the compliance period in the second calendar year.

### *3.4 Multivariate Regressions for City-Level ETSs*

We ran multivariate regressions for the three better operated city-level ETSs, Beijing ETS, Shanghai ETS and Shenzhen ETS, as shown in Table 9. The dependent variables are the CEA price returns (*lnReturn*$_t$) of Beijing, Shanghai, and Shenzhen respectively.

It was found that the one week lagged Brent oil price return (*D.lnBrent*$_t$) had a short-term positive effect on Beijing CEA price at 5% significance level. It indicates that the increase of oil price led to the short-run increase of Beijing CEA, as enterprises would substitute coal for oil to some extent, resulting in an increase in emissions and CEA demand. And, according to the joint F test, changes in oil price granger cause changes in CEA price of Beijing at 5% significance level. Additionally, the changes in stock index displayed a weakly positive effect in Eq.2 of Beijing ETS. It implies that the increase in stock returns, as a sign of future economic growth, drives the industrial output and the associated emissions, which raises CEA price.

With respect to Shanghai, the lag order one of the coal price returns had a negative effect on CEA price returns, which is similar to Beijing but more significant. And, the F-test indicates that changes in coal price granger cause changes in Shanghai CEA price. In the contrast, the 1 week lagged LNG price return ($D.lnLNG_t$) displayed a significantly positive effect on CEA price returns. The result meets our expectation based on the substitution theory that CEA price increases when LNG is substituted by cheaper and carbon-intensive fuel such as coal. No Granger causality was found between LNG price and CEA price based on the joint F-test. Eq.3 shows a slight Granger causality from stock returns to CEA price returns.

The regressions for Shenzhen ETS found no significant relations between CEA price and energy prices. However, there was a significant short-run effect of stock returns on CEA price return with a two-week lag. The positive coefficient implies that CEA price would increase under good economic conditions due to more economic output and the associated emissions. The joint F-test also indicates a significant Granger causality from stock returns to CEA price returns.

Further, we can see from Table 9 that Beijing, Shanghai, and Shenzhen all experienced a decrease in CEA price returns after *Break2014*. This is similar to the previous finding that the piloting ETSs had higher level CEA prices at the beginning of their establishment in 2013, but displayed significant decreases of the prices after the compliance break in the second calendar year (2014). However, Shanghai CEA price increased after June 2016 and Shenzhen CEA price increased after June 2015 which may be caused by the demand from new entrants.

## 4. Discussion and Policy Implications

In general, as newly born markets, the local ETS markets in China face thin trading and volatile price. One reason is the low market liquidity. It may be difficult for a seller of $CO_2$ emission allowances to quickly find a buyer that they want to make the transaction with. And, some entities with allowance surplus may want to keep the allowances for their own use in the next compliance year rather than selling them. In China, the spot trading is allowed, but derivatives trading is off-limits. This limited type of commodities in the market in addition to firms' lack of understanding and capacity of playing in the ETS market can also be the reasons for a dearth of actual trading.

The local CEA prices are far less than the ideal prices that can cause substantial low-carbon actions. The problem of low-level pricing happens at other existing ETSs as well. As a comparison, the EUA price in EU-ETS during Phase II (2008–2012) was about 23.64 \$/ton $CO_2e$ on average (Daskalakis 2013). Nonetheless, it decreases to a low level in Phase III (2013–2020). For instance, the EUA price was only about 8\$/ton $CO_2e$ in April 2015 and about 6\$/ton $CO_2e$ in April 2016 (World Bank 2015; 2016), which were similar to the level of the CEA price in Shenzhen ETS at that time. The low-level economic incentive from ETS was not able to encourage the regulated enterprises to invest in mitigation technologies in China (Yang, Li, and Zhang 2016). The enterprises rather considered participation in ETS as an approach to enhance public image and the ties with governments (Yang, Li, and Zhang 2016). However, a market-based policy instrument like ETS tends to be more acceptable by enterprises than the traditional command-and-control approach (Liu et al. 2013). Also, ETS mobilized a large number of business actors (emitters and intermediaries), local officials and researchers to work on low-carbon strategies and activities. The cooperative governance network constructed during the process is likely to be necessary and more cost efficient for long-run emission reduction in China.

The price values of CEA and the level of trading activities vary across the ETS pilots, which is reasonable considering the differences in policy design, local governments' political will and local economic context. Among the city-level ETSs, Chongqing and Tianjin are not so market-oriented, with lower CEA prices and less active transactions. Loose enforcement, the oversupply of allowances and emphasis on local economic interests can be the reasons. An implication is that even though China is keen to develop the national ETS, it may work better in some regions than others at the local level.

When investigating into the Granger causality from energy prices to CEA prices, the findings varied among local ETSs, because of local differences in market dynamics of CEA and energy resources. We found no significant Granger causality from energy price changes to CEA price changes in the two provincial level ETSs, Hubei, and Guangdong. Regarding the city-level ETSs, we found a Granger causality from oil price changes to CEA price changes in Beijing. And, there was a positive effect of oil price changes on CEA price changes with the one-week lag. It implied a short-term substitution of oil with coal which is cheaper and more carbon-intensive. In Shanghai, we found a Granger causality from coal price changes to CEA price changes in Shanghai. And the coal price changes had a negative short-run effect on the CEA price changes, indicating that an increase in coal price drives a move away from coal toward less carbon-intensive fuels (e.g. natural gas or oil). It was also found that there was a positive short-run effect of LNG price changes on Shanghai CEA price changes, which is also consistent with the substitution theory. The scale of the short-run effect of LNG price is smaller than that of coal price. This is because the coal consumption makes up a larger portion of the energy consumption so that the changes in coal price lead to greater changes in emissions and the CEA price. No Granger causality was found between LNG price and CEA price.

Additionally, we found a Granger causality from stock index changes to CEA price changes in Beijing, Shanghai, and Shenzhen. In Shenzhen, the Shanghai Shenzhen 300 stock index displayed a positive short-run effect on CEA price with a two-week lag. The same with Beijing. Therefore, the CEA price would increase at the times of economic prosperity, because the larger industrial outputs and the associated emissions can raise the demand for CEA. Further, the coefficients on break dummies show that all ETSs experienced a higher-level CEA price at the initial stage of operation but a decrease of the price after the compliance break in the next calendar year. The decrease in the CEA price could be because the demand for CEA decreased over time. So, policymakers need to consider strengthening the CEA allocation to reduce the CEA supply and increase the CEA demand, in order to keep a high-level CEA price.

Overall, this study contributes to our understanding of ETS (or other tradable permit policies) by adding empirical evidence in the context of China. The local ETSs set good examples of institutional innovations when adopting ETS and their performances show how ETS works in regions under different development stages. Our econometric analyses highlight that energy prices have significant influences on CEA prices, but the influences are different across local ETSs. In future, for either local ETSs or the national ETS, policymakers and investors should be aware of the dynamic relationships between the energy markets and CEA markets to reduce the CEA price volatility caused by the fluctuations of energy prices. A big problem of the ETSs is that the CEA price is too low. So, when coal price increase, or when LNG price and oil price decrease, the regulators can somehow buy in allowances to prevent CEA price from dropping, especially if the CEA price is low. Another implication is that policies promoting the use of cleaner energies (e.g. natural gas) or general energy efficiency programs may reduce CEA price. Such policies should be followed by methods to shorten CEA supply or increase CEA demand.

China's National Development and Reform Commission (NDRC) has announced its *Plan for Building the National Carbon Emission Trading Market for Power Generation Industry* on 18 December 2017. It says China will take about one year to complete the emission exchange system and take another one year to try out the allowance allocation and trading before it officially operates the national ETS. We can expect that it will be very challenging for China to operate ETS at the national level, considering the performance diversity of the local ETSs at the piloting stage. The differences in CEA prices between local ETSs, to some extent, reflect the differences in local marginal abatement costs. It is practical to start with a single industry sector and brings in more sectors when the institutional design is more mature and the measurement, reporting and verification (MRV) system is more complete. However, the power generation sector has faced with many other policies, such as renewable energy promotion policies. The coordination of the policy mix is important.

The integration of the local ETSs into the national ETS will be challenging too. For instance, the sectoral coverage of the national ETS is different from the current local ETSs. NDRC's solution is that the national ETS will regulate the power generation sector and gradually take in other sectors. Meanwhile, the local ETSs will continue to operate until the national ETS is fully functional. However, the standards for identifying the potential participants are different between the national level and the local level. For example, Shenzhen includes industrial enterprises that have annual emissions larger than 3000 tons $CO_2e$, but the standard set at the national level is 26000 tons $CO_2e$ annually, which means that some enterprises regulated by Shenzhen ETS may not be regulated by the national ETS. These regulatory uncertainties will discourage the current participants in local ETS from actively engaging in the market or taking low-carbon actions. The standards at the national level should be stricter than those of the local ETSs and consistent political support is crucial (Bolun et al. 2018). The central government should offer guidelines and methodologies on design and operations of ETS regarding coverage and scope, MRV, allowance allocation and enforcement, while the local governments should be given discretion with implementation taking into account local contingencies (Bolun et al. 2018).

## References

Alberola, E., J. Chevallier, and B. Chèze. 2008a. "Price Drivers and Structural Breaks in European Carbon Prices 2005–2007." *Energy Policy* 36: 787-97. https://doi.org/10.1016/j.enpol.2007.10.029.

Boersen, A., and B. Scholtens. 2014. "The Relationship Between European Electricity Markets and Emission Allowance Futures Prices in Phase II of the EU (European Union) Emission Trading Scheme." *Energy* 74: 585-94. https://doi.org/10.1016/j.energy.2014.07.024.

Bolun, N., Z. Yongguan, X. Zhihong, and F. Bojie. 2018. "Developing China's National Emission Trading Scheme: Experiences from Existing Global Schemes and China's Pilot Programs." *Chinese Geographical Science* 28: 287-95. https://doi.org/10.1007/s11769-018-0947-5.

Chevallier, J. 2011. "A Model of Carbon Price Interactions with Macroeconomic and Energy Dynamics." *Energy Economics* 33: 1295-312. https://doi.org/10.1016/j.eneco.2011.07.012.

Creti, A., P.A. Jouvet, and V. Mignon. 2012. "Carbon Price Drivers: Phase I Versus Phase II Equilibrium?" *Energy Economics* 34: 327-34. https://doi.org/10.1016/j.eneco.2011.11.001.

Daskalakis, G. 2013. "On the Efficiency of the European Carbon Market: New Evidence from Phase II." *Energy Policy* 54: 369-75. https://doi.org/10.1016/j.enpol.2012.11.055.

Fan, J.H., and N. Todorova. 2017. "Dynamics of China's Carbon Prices in the Pilot Trading Phase." *Applied Energy* 208: 1452-67. https://doi.org/10.1016/j.apenergy.2017.09.007.

Hammoudeh, S., D.K. Nguyen, and R.M. Sousa. 2014a. "Energy Prices and CO2 Emission Allowance Prices: A Quantile Regression Approach." *Energy Policy* 70: 201-06. https://doi.org/10.1016/j.enpol.2014.03.026.

Hammoudeh, S., D.K. Nguyen, and R.M. Sousa. 2014b. "What Explain the Short-Term Dynamics of the Prices of CO2 Emissions?" *Energy Economics* 46: 122-35. https://doi.org/10.1016/j.eneco.2014.07.020.

Hammoudeh, S., A. Lahiani, and D.K. Nguyen, and R.M. Sousa. 2015. "An Empirical Analysis of Energy Cost Pass-Through to CO2 Emission Prices." *Energy Economics* 49: 149-56. https://doi.org/10.1016/j.eneco.2015.02.013.

Keppler, J.H., and M. Mansanet-Bataller. 2010. "Causalities Between CO2, Electricity, and Other Energy Variables During Phase I and Phase II of the EU ETS." *Energy Policy* 38: 3329-41. https://doi.org/10.1016/j.enpol.2010.02.004

Kim, H.S., and W.W. Koo. 2010. "Factors Affecting the Carbon Allowance Market in the US." *Energy Policy* 38: 1879-84. https://doi.org/10.1016/j.enpol.2009.11.066.

Koch, N., S. Fuss, G. Grosjean, and O. Edenhofer. 2014. "Causes of the EU ETS Price Drop: Recession, CDM, Renewable Policies or a Bit of Everything?—New Evidence." *Energy Policy* 73: 676-85. https://doi.org/10.1016/j.enpol.2014.06.024

Liu, X., D. Niu, C. Bao, S. Suk, and K. Sudo. 2013. "Awareness and Acceptability of Companies on Market-Based Instruments for Energy Saving: A Survey Study in Taicang, China." *Journal of Cleaner Production* 39: 231-41. https://doi.org/10.1016/j.jclepro.2012.08.009.

National Development and Reform Commission (NDRC). 2015. "Enhanced Actions on Climate Change: China's Intended Nationally Determined Contributions (INDCs)." Submitted to UNFCCC.

Olivier, J.G.J., G. Janssens-Maenhout, M. Muntean, and J.A.H.W. Peters. 2015. "Trends in Global CO2 Emissions: 2015 Report." The Hague, The Netherlands; Ispra, Italy: PBL Netherlands Environmental Assessment Agency & European Commission's Joint Research Centre (JRC).).

Quirion, P. 2005. "Does Uncertainty Justify Intensity Emission Caps?" *Resource and Energy Economics* 27: 343-53. https://doi.org/10.1016/j.reseneeco.2005.05.002.

Tan, X. and X. Wang. 2017a. "The Market Performance of Carbon Trading in China: A Theoretical Framework of Structure-Conduct-Performance." *Journal of Cleaner Production* 159: 410-24. https://doi.org/10.1016/j.jclepro.2017.05.019.

Tan, X., and X. Wang. 2017b. "Dependence Changes Between the Carbon Price and its Fundamentals: A Quantile Regression Approach." *Applied Energy* 190: 306-25. https://doi.org/10.1016/j.apenergy.2016.12.116.

Wang, P., H. Dai, S. Ren, D. Zhao, and T. Masui. 2015. "Achieving Copenhagen Target Through Carbon Emission Trading: Economic Impacts Assessment in Guangdong Province of China." *Energy* 79: 212-27. https://doi.org/10.1016/j.energy.2014.11.009.

Weaver, R.K. 2010. "But Will It Work? Implementation Analysis to Improve Government Performance." *Governance Studies at Brookings.* No.32: 1-17. Retrieved from: https://www.brookings.edu/wp-content/uploads/2016/06/02_implementation_analysis_weaver.pdf.

World Bank 2015. "*Carbon Pricing Watch 2015*." Washington, D.C., USA: World Bank. https://doi.org/10.1596/978-1-4648-0268-3.

World Bank 2016. "*Carbon Pricing Watch 2016*." Washington, D.C., USA: World Bank. https://doi.org/10.1596/978-1-4648-0268-3.

Yang, L., F. Li, and X. Zhang. 2016. "Chinese Companies' Awareness and Perceptions of the Emissions Trading Scheme (ETS): Evidence from a National Survey in China." *Energy Policy* 98:, 254-65. https://doi.org/10.1016/j.enpol.2016.08.039.

Zeng, S., X. Nan, C. Liu, and J. Chen. 2017. "The Response of the Beijing Carbon Emissions Allowance Price (BJC) to Macroeconomic and Energy Price Indices." *Energy Policy* 106: 111-21. https://doi.org/10.1016/j.enpol.2017.03.046.

Zhang, J., and L. Zhang. 2016. "Impacts on CO2 Emission Allowance Prices in China: A Quantile Regression Analysis of the Shanghai Emission Trading Scheme." *Sustainability* 8: 1195. https://doi.org/10.3390/su8111195.

*China Policy Journal • Vol. 1, No. 1 • Fall 2018*

# Payment for Ecological Services and River Transboundary Pollution: Policy Inspirations from a Contingent Valuation (CV) Study on the Xijiang River Drainage Basin in South China

Jie He[1]


## Abstract

Based on the contingent valuation study results reported by He, Huang, and Xu (2015a), we propose a new payment standard-setting framework that could include the total transfer that a city should pay as a polluter or receive as a victim. This new framework differs from the previous mechanism by explicitly excluding the willingness to pay (WTP) reduction due to a city's own pollution discharge and focusing only on the WTP variation caused by transboundary pollution. This new framework also allows the calculation of detailed bilateral monetary transfers between cities depending on their location on a river and their contribution to the variation of water quality. One advantage of our approach is the possibility to identify not only polluters and victims but also "cleaners" who inherit bad water quality from the upstream neighbor and clean it up. The compensation regime proposed by our approach can thus determine both the compensation for negative externalities to be paid by the polluters to victims and the compensation from the "victim-to-be" to the cleaners for their efforts, which creates positive externalities and prevents their downstream neighbors from suffering from potential welfare loss. Based on our results, it seems that simply using the total WTP as the compensation standard for a better ecological service risks mixing the pollution caused by upstream cities with the pollution from a city's own activities, which thus tends to exaggerate the necessary compensation payment; for the Xijiang River, such exaggeration can range from 2 to 10 times. We also compared our results with the Xin'an River PES pilot program, whose transfer amount was arbitrarily fixed at 500 million yuan per year, which is approximately 86% of the compensation amount that Foshan city needs to pay to Zhongshan city. Our results therefore can be considered a supportive ar-


[1] *University of Sherbrooke, Canada;* Jie.He@USherbrooke.ca

 doi: 10.18278/cpj.1.1.3

gument for the general belief that the transfers currently used in the existing pilot programs are relatively low and may discourage the motivation of the cities along rivers to give sufficient effort to ecological service preservation.

Keywords: Payment for Ecological Services, payment standard, transboundary river pollution, willingness to pay (WTP), China

# Pago por servicios ecológicos y contaminación fluvial transfronteriza: inspiraciones políticas de un estudio de valoración contingente (QA) en la cuenca de drenaje del río Xijiang en el sur de China

## Resumen

Sobre la base de los resultados del estudio de valoración contingente informados por He, Huang y Xu (2015a), proponemos un nuevo marco de establecimiento de estándares de pago que podría incluir la transferencia total que una ciudad debe pagar como contaminador o recibir como víctima. Este nuevo marco difiere del mecanismo anterior al excluir explícitamente la reducción de la disposición a pagar (WTP, por sus siglas en inglés) debida a la descarga de contaminación de una ciudad y centrarse solo en la variación de WTP causada por la contaminación transfronteriza. Este nuevo marco también permite el cálculo de transferencias monetarias bilaterales detalladas entre ciudades en función de su ubicación en un río y su contribución a la variación de la calidad del agua. Una de las ventajas de nuestro enfoque es la posibilidad de identificar no solo a los contaminadores y las víctimas, sino también a los "limpiadores" que heredan y limpian la mala calidad del agua del vecino que está aguas arriba. El régimen de compensación propuesto por nuestro enfoque puede determinar tanto la compensación por las externalidades negativas que deben ser pagadas por los contaminadores a las víctimas como la indemnización de la "futura víctima" a los limpiadores por sus esfuerzos, lo que crea externalidades positivas e impide que sus vecinos aguas abajo sufran de una posible pérdida de bienestar. Según nuestros resultados, parece que el simple uso de la WTP total como el estándar de compensación para un mejor servicio ecológico corre el riesgo de mezclar la contaminación causada por las ciudades río arriba con la contaminación de las actividades propias de la ciudad, lo que tiende a exagerar el pago de la compensación necesaria; para el río Xijiang, tal exageración puede variar de 2 a 10 veces. También comparamos nuestros resultados con el programa piloto Xin'an River PES, cuyo monto de transferencia se fijó arbitrariamente en 500 millones de yuanes por año, que es aproximadamente el 86% del monto de compensación que la ciudad de Foshan debe pagar a la ciudad de Zhongshan. Por lo tanto, nuestros resultados pueden considerarse un argumento de apoyo para la creencia general de que las transferencias actualmente utilizadas en los programas piloto existentes son relativamente bajas y pueden desalentar la motivación de las ciudades a lo largo de los ríos para hacer un esfuerzo suficiente para preservar el servicio ecológico.

***Palabras clave:*** pago por servicios ecológicos, pago estándar, contaminación fluvial transfronteriza, disposición a pagar (WTP), China

# 生态服务付费与河流跨界污染：关于中国南方西江流域意愿调查研究的政策启示

摘要

根据何、黄、许(2015a)三位学者报告的意愿调查研究结果，笔者提出了一种新的支付标准制定框架。该框架可以涵盖一个城市作为污染者应支付或作为受害者应得到的全部转移费用。这一新框架与以前的机制不同，它明确排除了由于城市自身的污染排放而导致的支付意愿(WTP)减少，而只侧重于跨界污染造成的支付意愿变化。这一新框架还允许计算城市之间的详细双边货币转移，具体取决于城市所在的流域及其对水质变化的贡献。这种方法的一个优点在于，它不仅可以识别污染者和受害者，还可以识别“清洁工”，他们从上游邻市那里继承了劣质的水源，并将其清理干净。因此，笔者所提出的赔偿制度，既可以决定污染者向受害人支付的负外部性补偿，也可以决定“准受害者”向清洁工支付的努力补偿，因为它创造了积极的外部环境，并防止下游邻市遭受潜在的福利净损失。基于笔者的研究结果，单纯以总意愿

支付原则作为更好生态服务的补偿标准，可能会将上游城市所造成的污染与该城市自身活动产生的污染混为一谈。因此，这往往会夸大所需的赔偿金；对于西江流域来说，这种夸张程度可达2至10倍。笔者还将其结果与新安江PES试点项目的结果进行了对比，该项目的转移金额随意定为每年5亿元，这大约是佛山市向中山市支付赔偿金的86%。人们普遍认为，目前在现有试点项目中使用的资金转移相对较少，可能会抑制沿江城市想要努力促进生态服务保护的动机。笔者的结果为这一看法提供了有力论证。

关键词：生态服务付费，支付标准，跨界河流污染，支付意愿原则(WTP)，中国

## 1. Introduction

River flows create upstream and downstream regions. However, administrative boundaries between regions do not prevent pollution in the water from crossing regional borders. Such difficulties in clearly defining the property rights of the river water flowing through different administration jurisdictions because of the weak excludability and strong rivalry of the water resources (in both terms of quality and quantity) can lead to the non-satisfaction of the basic Samuelson rules (1954). Therefore, an upstream region that is not able to enjoy the full benefits of its water conservation and pollution control efforts may exert insufficient control, which results in the overuse of water resources and the increased discharge of pollution.

Most large-scale river basins in China (e.g., the Yangtze River, Yellow River, and Xining River) span several regional jurisdictions (provinces, regions and cities). Although environmental policy is often centrally developed and local jurisdictions can only set their own environmental standards to more stringent levels than those of the national level, implementation responsibilities are devolved to the branch offices of the Ministry of Environment Protection (MEP), which operate at the provincial, municipal and county levels (Hills and Roberts 2001). Combining these two facts, we believe that there is the possibility of a problem of transboundary river water pollution for China's rivers.

Further supportive arguments for such a possibility can be made by considering the complexities and fragmentation in water resource management between the different authorities in China. Yu (2011) has described the complex relationships between the Ministry of Water Resource Management (MWRM), which addresses water quantity and water utilization, and the MEP, which coordinates and solves environmental pollution disputes. Either overlaps or gaps that exist between the competences of the two authorities may largely compromise the efficiency of their efforts with respect to transboundary river water pollution control.

In addition to leading to higher pollution levels in neighbor regions, a more worrying aspect of transboundary pollution is its potential dynamic impacts on the motivation for regions to efficiently control their own resource usage and pollution discharge. Oates and Portney (2003) indicated that the presence of the risks of transboundary negative externalities may lead to a "race to the bottom" of regional pollution control policies since the concerns about the transboundary movements of pollution from neighbors may compromise the determination of a region to exert effective pollution control measures.

Since 2000, the payment for ecological services (PES) mechanism has become one of the most advocated environmental policy measures in China. From the beginning, many Chinese scholars have considered this policy tool to be one of most efficient measures to improve the ecological conditions of different river drainage basins and to ease the heavy pressure on China's relatively poor water resources from economic activities. Numerous pilot projects have been carried out in China for several years. These include not only the application of the PES mechanism in wetland protection projects in many key areas but also some quantity preservation and quality improvement projects in surface waterbodies (e.g., the Beijing Miyun Reservoir, Dongjiang Source Area, Thousand Island Lake Basin, Pearl River Drainage basin and the River Heihe Drainage Basin) (MEP 2013).

Fundamentally, PES is a mechanism aiming at remedying market failures caused by the nature of public good and the poorly defined property rights of ecological services. By intentionally establishing an artificial market mechanism, the logic of the PES is to motivate and institutionalize a payment system between upstream and downstream jurisdictions along a river, which can serve as a monetary counterpart to internalize the negative externalities along rivers due to transboundary pollution over-discharge.

Although the theoretical foundation of the PES mechanism seems easy to understand, its application in the real world has proven to be much more difficult. PES is a mechanism for internalizing transboundary negative externalities; determining whether and how to apply the PES mechanism requires a good understanding of the phenomena of such externalities. Although the existence of transboundary pollution has already been confirmed at both the international and province/state levels, to date, there have been few studies that directly consider its existence in China. Additionally, even if we can provide evidence about the existence of transboundary pollution along rivers in China, to build a direct measurement of such a negative externality requires explicit identification of the source and the impact of the negative externality. Moreover, to directly relate such impact measurements to an efficient payment mechanism also requires a precise understanding of how and to what extend such a negative externality affects the well-being of people living in the downstream and upstream jurisdictions. In other words, only when a reasonable measurement of the intrinsic value of the environmental/ecological service affected by the negative externality is obtained can one have confidence to hope that the payment mechanism based on such measurement can function correctly and be incentive-compatible.

In this paper, based on the related literature collected in China and in the world, and particularly the findings from the recent contingent valuation method (CVM) study conducted by He, Huang, and Xu (2015a), we try to answer the following questions. First, what is the current situation of transboundary river water pollution problem in China? Second, if it exists, how does the transboundary pollution problem affect people's perception about the efficiency of the existing water quantity and quality control policies, whose implementation is often closely related to the local government's capacity and local economic conditions? How can such concerns affect people's expected utility improvement for a targeted better water condition? Finally, how can we establish a valid payment standard for ecological services between upstream and downstream cities, and how will be this standard compare with those of other existing studies that have analyzed similar measurements?

## 2. Existence of Transboundary Pollution

There is already evidence for the existence of transboundary pollution at the international level and in foreign countries. However, we have not yet found studies that directly revealed transboundary river water pollution cases based on data from China's rivers.

Based on the data of GEMS/Water[1] biochemical oxygen demand (BOD) measured by 291 river monitoring stations in 49 countries during 1979–1990, Sigman (2002) found the BOD indicator to be significantly higher at stations that were located upstream of borders than comparable stations, at least among stations located in non-European Union (EU) countries.

Because most US federal environmental policies assign regulation, implementation and enforcement responsibilities to state-level authorities, Sigman (2005) investigated potential transboundary spillover phenomena in the US. To do so, a composite water pollution index based on five major pollutants compiled from 618 monitoring stations from 1973 to 1995 was used. Using a difference-in-difference logic, Sigman (2005) found that, all else being equal, the water quality indexes were 4% worse at stations located downstream from a state authorized and with power to implement and enforce its own regulations over river water pollution.

1 The UN's Global Environmental Monitoring System Water Quality Monitoring Project.

Using GIS water quality panel data from 321 monitoring stations across Brazil as well as jurisdictional boundary modification data for 5,500 Brazilian counties, Lipscomb and Mobarak (2008) studied whether water quality across jurisdictional boundaries deteriorates due to the concentration of polluting activities near a river's exit from a jurisdiction. Their results confirmed that within a 5-kilometer distance from a boundary, pollution increased by 2.3% for every kilometer closer a river was to the exit border.

In addition to the river transboundary pollution cases, transboundary air pollution has been studied. One interesting example is Helland and Whitford (2003). Motivated by anecdotal evidence suggesting that local regulators were more lenient in their treatment of polluters when the incidence of pollution partially fell on those outside the state, this paper examined a transboundary air pollution spillover incidence that occurred in the US and revealed significantly higher toxic chemical levels in border counties.

## 3. Standards for Payment of Ecological Services

For the PES mechanism, the most important question is how to set reasonable payment standards for the affected ecological services. By "selling" the ecological services provided by environment protectors to the beneficiaries, this mechanism aims to generate funds to increase the conservation benefits perceived by the environmental protectors and therefore reinforces their incentives to protect the environment and resources. Following this logic, we should expect the payment of the PES mechanism to be higher than or at least equal to the conversation cost and/or the opportunity cost of the existing commercial development projects to which the environmental protectors face. The larger the gap is between the perceived benefit and the cost, the more room there is for negotiation between environmental service providers and beneficiaries and the higher the probability is to bring welfare increases for both sides and to realize effective environmental protection.

However, the difficulties in assigning pertinent monetary values to affected ecological services are numerous. Although natural environments represent one of the cornerstones of the human environment and offer essential goods and services for human survival and well-being, their integration into the economic system has proven to be very complex. The process to include the total economic value of nature in a neoclassical logic requires the encounter of two fundamental elements: the physical, biotic and abiotic components of nature on the one hand and the individual's view of these elements on the other. If it is reasonable to use the wellbeing that the individual obtains from these natural components, two aspects of the difficulty of putting a dollar value to such wellbeing remain: First, not all the wellbeing obtained by an individual is exchangeable in a market. The response of the nature to the multitude of human needs, whether aesthetic, cultural or educational, are very often non-excludable and non-rival, which means that their exchange on the market is impossible. Second, many aspects of the wellbeing that an individual receives from nature are not measurable, and some elements of the total value of nature to humans may not even be observable. A good example is the non-use value of the environment and resources: simply knowing the "existence" of good functions of the ecological system and the assurance of their provision to future generations can create satisfaction that is not associated with any observable consumption behaviors.

Previous studies have proven that the non-use value may occupy a high percentage in the total value of the environment. Wattage and Mardle (2008) found that the proportion of aggregated preferences related to the use value to conserve a wetland ecosystem was 55.3%, compared with 44.7% for the non-use value. Sander, Walsh, and Loomis (1990) concluded that the use values (e.g., irrigation, swimming, fishing, and tourism) of the 15 rivers in the state of Colorado in the United States were only approximately 1/5 of their total value; the other 4/5 of the value was principally non-use value.

We can distinguish five categories of ecological goods and service (EGS) valuation methods. The first consists of the methods that are based on market prices; they only assess the direct use value of EGS referenced to their market value. The second category consists of the methods based on costs, which estimate the EGS value by the cost of avoided damage or the replacement cost of ecosystem losses. The third category consists of the revealed preference methods (e.g., hedonic price, travel costs), which are based on consumer preferences that are revealed by their behavior in an existing market. An example of this method, the hedonic price, considers the complementarity between air quality within an area and house prices (Bateman et al. 2011; Desaigues and Point 1993; Malër 1974) and uses the increase of the house value due to the better air quality as an assessment of the economic value of the better air quality, all else being equal. The fourth category consists of methods based on stated preference (e.g., contingent valuation, discrete choice experiments), which measure the value of EGS via simulated markets to identify survey respondent trade-offs between the price to pay (or compensation to accept) and improvement (or degradation) of the environment. Finally, the benefit transfer methods involve estimating the value of EGS for a target site using existing valuation estimates from primary studies for similar sites that explicitly use one of the four preceding method categories (Navrud and Ready 2007).

Each method has its advantages and disadvantages. The measure of cost/market price is relatively easy to use but only focuses on marketable characteristics of the ecosystem. Revealed preference methods depend on observable consumer behaviors in markets for complementary goods and can thus only measure the direct- and indirect-use values of EGS. Benefit transfer is a secondary method that extrapolates the results obtained by one or many primary studies; it is thus not suitable for a primary study focusing on a specific test area. Compared with the abovementioned methods, the stated preference methods provide a more flexible approach and aim at establishing a hypothetical market framework; therefore, it can include in its assessment both the use value and the non-use value of EGS. However, stated preference methods also face criticisms that are related to their hypothetical nature (e.g., Carson and Groves (2007); Harrison and Rutström 2008; List and Gallet 2001; Murphy et al. 2005) and its potential influence on collected answers from respondents, which may lead to either an over- or under-estimation of people's WTP.

Our review of the related literature provided us with a quite interesting picture about the academic efforts in evaluating such welfare benefits. Over the last several decades, many Chinese scholars conducted interesting case studies with the aim of stimulating discussion about how to set PES payment standards. Some of them employed diverse methods based on market prices or opportunity cost, such as Cai, Lu, and Song (2008), who calculated the total engineering cost of the construction project for the water source protection area in the eastern route of the South-to-North Water Transfer Project and proposed a cost-sharing plan between regions based on the potential added-value of the ecological service improvement that they would receive. Li et al. (2009) estimated the opportunity cost of the forest protection project on the mountainous regions of Hainan and proposed to determine the payment standard based on the land holdings of different regions. Shen et al. (2009) estimated the potential loss of agricultural production due to the Green Agricultural Demonstration Project on Chongming Island. We also found several papers that calculated the loss of economic development opportunities due to water conservation projects in some river drainage basins, such as Fu, Ruan, and Zhang (2011) for the Yongding river, Zhang (2011) for the Xijiang river and Shi et al. (2012) for the Dongjiang river. Other authors chose to evaluate the potential value of the conserved ecological services; for example, Xu et al. (2006) calculated the ecological service value of the Lianhua Reservoir ecological protection project, Huang, Luo, and Yang (2008) estimated the ecological service value of the Dayao Mountain's water conservation project, Jin and Wang (2008) evaluated the use and non-use value of ecological services provided by the water conservation forest on Qilian Mountain and Cai et al. (2010) estimated the ecological service value of the wetlands in the Qilihai natural protection areas.

However, is proposing some numbers better than having no numbers? One common difficulty that those studies faced was the big divergence between the numbers they proposed. Some of these differences can be explained by the differences in the methodologies used. For example, the methods based on the opportunity cost may only include the use value of the ecological services in their estimates, whereas the stated preference methods have the capacity to include the non-observable non-use values, which may represent a large percentage of the total value of the interested ecological services. A good example is the significance coefficients that He et al. (2015b) reported for the methodological dummies in their meta-analysis estimation function, which revealed the very large impact of methodological choices on the reported value of the ecological services provided by wetlands.

It is relatively easy to accept the fact that the different evaluation methods propose relatively different valuation results for ecological services. There is another potential reason to explain the differences that has not yet gained enough attention in the literature: the ambiguity among the state preference valuation studies about "what" to evaluate.

Consider the example of a typical payment scenario for ecological services related to a better water quality provided by an upstream region. Ideally, the downstream residents, as the beneficiaries of the improved ecological services provided by the better quality of river water flowing from the upstream regions, should only pay for the part of the increase in ecological services directly related to the better water quality contributed by the upstream regions. Such logic is already well reflected in the mechanism of some pilot PES projects, such as the trans-provincial project on the Xin'an River (2012–2014). This project required that the decision to transfer a proposed 500 million yuan between the upstream Anhui province and the downstream Zhejiang province be based on the water quality of the river section located in the congruent frontier between the two provinces. If the water quality meets the required class II level, the 500 million yuan will be transferred from Zhejiang to Anhui to compensate for their water pollution abatement efforts. Conversely, if the water quality does not meet the required level, the transfer will go from Anhui to Zhejiang province to compensate for the additional damage caused by the worse water quality. Whether to make a transfer depends on whether the Xin'an River water received by Zhejiang from Anhui meets the required quality target. Compared with this specificity, the welfare changes that many existing stated preference valuation studies measured unfortunately were wider; most of them focused on the potential reduction of local people's well-being due to the changes in the quality of the local ecological service. For the case of river water pollution, the local water quality changes certainly "inherit" the pollution flows from the upstream regions but are also directly affected by the injection of pollution from local economic activities and everyday life.

## 4. A New Framework for Setting Payment Standards with the Stated Preference Methods

Considering the above discussion, we use the following Figure 1 to illustrate the different sources of pollution in a transboundary river. Assuming that the river water at the starting point S has a quality equal to or better than level II, the river flows from the point S (start) to point E (end) and goes through five regions (A, B, C, D and F). All five regions discharge pollution into the river and thus can be considered polluters. They also suffer from the concentration of the pollution in the section of the river flowing through their jurisdictions and can thus be considered as victims. However, it is important to distinguish between the simple victims of river water pollution and the victims of the transboundary pollution; only the latter can be qualified for the discussion about payment ecological service compensation. Region A is located at the starting point of the river and does not have upstream neighbors. Therefore, although A is the polluter of the river and victim of the pollution in the river due to its own pollution discharge, it is not a transboundary pollution victim. Taking now the case of region B, since A is its upstream neighbor, it is possible for B to be a transboundary pollution victim. However, to confirm the victim role of region B, another condition is that the quality of the river water received by region B from region A is worse than Class II, the targeted river water quality. The same discussion can be applied to regions C and D. Finally, for region F, since it is located at the end point of the river, F can only be a transboundary victim but not a transboundary polluter since the river water, after flowing through the region F, will flow into the ocean and thus does not affect other populations living in the drainage basin.

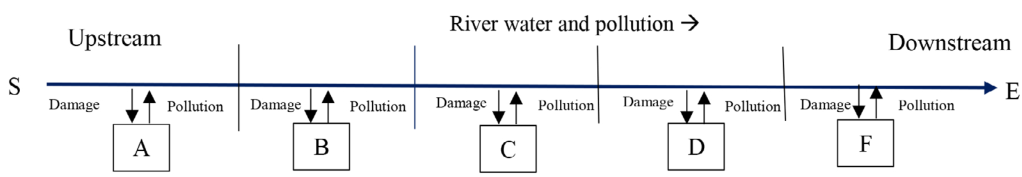



**Figure 1.** Polluter and Victim in a Transboundary Pollution Case

Once the conditions to identify transboundary pollution-related polluters and victims are clear, another related question is about when and how the PES is installed. There are two possible situations. First, although all regions discharge pollution into the river, if the water quality collected at all transboundary sections of the river between regions stays at a constant class II level, it will not be necessary to apply a PES mechanism because each region manages to keep the water quality as clean as the water they receive from their upstream neighbors; therefore, we can consider the situation as not having transboundary pollution between regions (Figure 2).

However, the situation can be much more complicated if the information about the water quality in the transboundary section of the river is organized as in Figure 3. The identification of the polluter and victim of transboundary pollution needs to con-

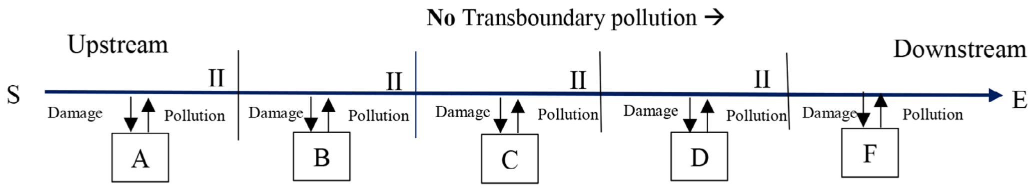



**Figure 2.** No PES Transfer Regime When There Is No Transboundary Pollution

sider the quality of the water that a region receives from its upstream neighbor and the quality of the water that it leaves to its downstream neighbor. For example, if region A, which keeps the water quality on its border with B at the required Class II level, should not be considered a transboundary polluter, then B will not be considered a transboundary victim. However, if the water quality between B and C is found to be at class III after flowing through region B, this means that B is unable to guarantee the same water quality as that received from A. In such a case, B will be regarded as a transboundary polluter and should be responsible for the damage suffered by the population living in all three downstream regions (C, D and F; the green arrows illustrate the direction of the compensation from B to the three downstream regions). A possibility for region C illustrated in the figure is that C manages to keep the water quality at level III, which is lower than the required quality but is equal to the quality of the water it receives from B; therefore, we should not consider C a transboundary polluter but simply a transboundary pollution victim (caused by B). If region D receives class III water quality but leaves its water quality even worse at class IV, D should be considered both a transboundary polluter responsible for the damage caused in region F (the blue arrow illustrates the compensation from D to F) and a transboundary pollution victim of region B.

In Figure 4, we present another possible spatial distribution pattern of transboundary water pollution. Compared with Figure 3, in this new scen-

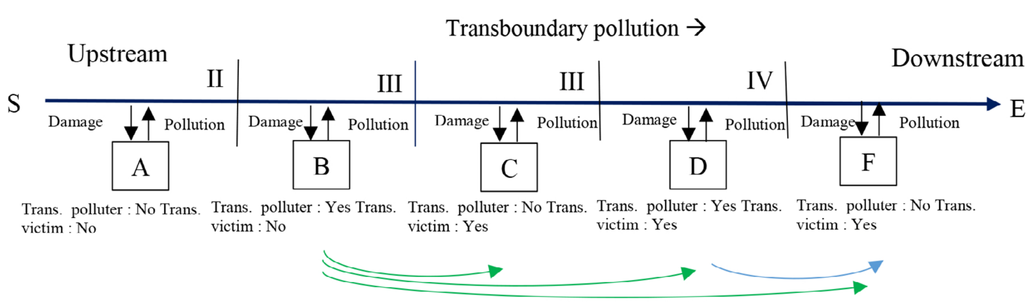



**Figure 3.** PES Transfer Regime with Transboundary Pollution: Scenario 1

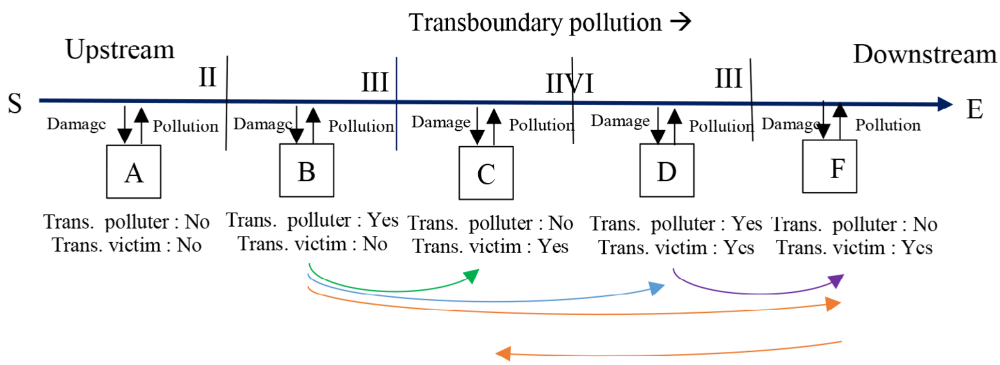



**Figure 4.** PES Transfer Regime with Transboundary Pollution: Scenario 2

ario, we assume that region C exerts more than necessary efforts and thus brings the water quality back to Class II level, which means a quality better than what it received from its upstream neighbor B. In such a situation, since C helps D and F avoid the negative externality from the Class III level polluted water, C should be eligible to recuperate the compensation paid by B to D and F (illustrated by the arrows of the same color but in opposite directions).

Based on this logic, if we conduct a state preference study among these cities with the aim of providing a standard for PES, simply questioning people's WTP for the water quality improvement target to Class II level cannot give us the correct information about the necessary amounts of payment transfers between cities because the water quality in the section of river flow through one particular city depends not only on the pollution discharged by the city itself but also on the pollution flows from the upstream cities.

We can use a WTP function to illustrate this idea for the case of the five cities A, B, C, D and F over the river. The WTP for each of the five cities can be written as

$$W_A = w\left(\overbrace{Q_A}^{+}\right) \tag{1}$$

$$W_B = w(\overbrace{Q_B}^{+}, \overbrace{Q_{AB} - II}^{-}) \tag{2}$$

$$W_C = w\left(\overbrace{Q_C}^{+}, \overbrace{Q_{BC} - II}^{-}\right) = \left(\overbrace{Q_C}^{+}, \overbrace{(Q_{BC} - Q_{AB}) + (Q_{AB} - II)}^{-}\right) \tag{3}$$

$$W_D = w\left(\overbrace{Q_D}^{+}, \overbrace{Q_{CD} - II}^{-}\right) = w\left(\overbrace{Q_D}^{+}, \overbrace{(Q_{CD} - Q_{BC}) + (Q_{BC} - Q_{AB}) + (Q_{AB} - II)}^{-}\right) \tag{4}$$

$$W_F = w\left(\overbrace{Q_F}^{+}, \overbrace{Q_{DF} - II}^{-}\right) = w\left(\overbrace{Q_F}^{+}, \overbrace{(Q_{DF} - Q_{CD}) + (Q_{CD} - Q_{BC}) + (Q_{BC} - Q_{AB}) + (Q_{AB} - II)}^{-}\right) \tag{5}$$

Here, we assume that the WTP of the respondent living in a city i, $W_i$, depends not only on the water quality situation of the section of the river flowing through the city, $Q_i$, but also on whether the quality of the transboundary section of the river with its upstream neighbors is better or worse than the targeted level Class II. In general, we believe a worse quality of the water (signifying a higher $Q_i$ motivates a higher WTP of the population in city i for the achievement of the water quality improvement target; therefore, we expect a positive relationship between $Q_i$ and $W_i$. This WTP for a better water quality target is also negatively affected by the existence of negative externality caused by transboundary water pollution because a worse transboundary pollution level signifies more difficulties in realizing the targeted water quality improvement. We therefore expect a negative relationship between $W_i$ and the water quality of the transboundary section of city i with its upstream neighbor i−1, $Q_{i-1,i}$, when $Q_{i-1,i}$ is found to be worse than Class II level. Because this reduction of the WTP of people living in city i is directly related to the transboundary pollution caused by upstream city i−1, we propose to use this part of the WTP reduction as a valid base to set the payment standard for the PES mechanism.

To distinguish the related PES transfers between cities, we further decompose the transboundary pollution-related responsibility among the cities, which corresponds to the last part of the equations to the right of the last equal sign. Simple manipulation provides an interesting responsibility sharing regime among cities according to the water quality of the transboundary sections between them. Take the spatial transboundary pollution allocation pattern given in Figure 4 as an example:

City A is not a transboundary pollution victim since it is located at the beginning part of the river; this is also revealed in its WTP determination function $W_A=w(Q_A)$, which does not include a WTP reduction term related to the transboundary section pollution level.

For city B, as its upstream city A maintains $Q_{AB}$=II, the reduction of the WTP of city B due to transboundary pollution is equal to zero, which means a zero transfer from A to B.

For city C, because its transboundary section pollution with B is at class III, higher than that between A and B at class II, we have $Q_{BC}-Q_{AB}>0$, and C should receive a compensation from city B whose amount equals the reduction of WTP, which is $-W_D(Q_{BC}-Q_{AB})$. However, city C should not be compensated by city A since A keeps the water quality to the targeted Class II level, therefore $W_D(Q_{AB}-\text{II})=0$.

The situation of city D is that its upstream neighbor city C manages to restore the water quality in the transboundary section back to class II ($Q_{C-D}$=II)from class III ($Q_{BC}$=III). Therefore, city C does not need to compensate city D. However, this does not mean that there is no money transfer between city C and city D, which can be seen from the decomposition part of the equation, where $Q_{CD}$-II is further decomposed into three parts: $(Q_{AB}-Q_{BC})<0$, $(Q_{CD}-Q_{BC})<0$, $(Q_{BC}-Q_{AB})>0$ and $(Q_{AB}-\text{II})=0$. Associating these terms with the negative correlation factors of $w_D$, we know that $w_D(Q_{CD}-Q_{BC})>0$, which signifies an increase of the welfare of people in city D because of the water quality improvement efforts of city C. This motivates a transfer from D to C equal to $w_D(Q_{CD}-Q_{BC})>0$. We also have $w_D(Q_{BC}-Q_{AB})<0$, which means a transfer to receive by D from B. Since $Q_{CD}$=II and $Q_{AB}$=II, we have $(Q_{CD}-Q_{BC})=-(Q_{BC}-Q_{AB})$; therefore, $W_D(Q_{CD}-Q_{BC})=W_D(Q_{BC}-Q_{AB})$, which signifies that the transfer from D to C for welfare increase is equal to the transfer from B to D as compensation for welfare decrease. Therefore, the transfer from B to D is simply used by D to compensate C for its abatement efforts. An intuitive way to interpret such double-transfers is that the "polluter" city B compensates the "cleaner" city C for its effort that prevents the negative externality of city B's transboundary pollution from affecting city D.

Finally, for city F, since $Q_{DF}$=III, which means that city D creates transboundary pollution to F, $W_F(Q_{DF}$-III$)>0$ However, the total WTP changes can also be decomposed into four parts: $(Q_{DF}-Q_{CD})>0$, $(Q_{CD}-Q_{BC})<0$, $(Q_{BC}-Q_{AB})>0$, $(Q_{AB}$-II$)=0$ and $(Q_{CD}-Q_{BC})=-(Q_{BC}-Q_{AB})$. Therefore, we can distinguish a compensation transfer from D to F, as $W_F(Q_{DF}-Q_{CD})<0$, and a compensation transfer from B to F, as $W_F(Q_{BC}-Q_{AB})<0$. The latter part is then transferred to the cleaner part, the city C, as $W_F(Q_{CD}-Q_{BC})=-W_F(Q_{BC}-Q_{AB})>0$. 5. Application of the New PES Standard Setting Regime using He, Huang, and Xu (2015a)

## 5. Application of the New PES Standard Setting Regime using He, Huang, and Xu (2015a)

This new PES payment standard setting regime implies an important fact: the payments between upstream and downstream cities should be based not on the current pollution situation of river sections across the cities but on that of the transboundary section flowing between the cities. This requires the stated preference studies to include both information types in the estimation of WTP, which thus makes it possible to isolate the part of the WTP variation in one city due to the transboundary pollution coming from its neighbors.

Among the numerous stated preference valuation studies that aimed to provide a payment standard for river related ecological services, we have been able to identify only one paper that directly studied the influence of the transboundary water pollution on the WTP of people in China, that of He, Huang, and Xu (2015a), which is based on an in-person CVM survey conducted in 2012 in the 20 cities of four provinces (Guizhou, Yunnan, Guangxi and Guangdong) of southern China belonging to the Xijiang river basin.

The Xijiang River is the main channel and longest tributary of the Pearl River (cf. Figure 5). The Xijiang River flows for 2,217 kilometers from the north of Yunnan province eastward across Guizhou province and Guangxi province and through the Pearl River delta in Guangdong province and final-


**Figure 5.** Xijiang River Drainage Basin with 20 Surveyed Cities
(the size of the circle signifies the population of the surveyed city)

ly terminates at the southern China Sea near Macau; it is the largest river system in southern China. Similar to other regions in China, the Xijiang River Basin has experienced trends of increasing inequality with respect to economic development over the last 35 years. Uneven development between regions naturally leads to considerably different interpretations by regional governments about the relationship between the environment and the economy. Although a wave of environmental consciousness has begun to surface in some of the richest eastern coastal provinces and cities, several western inland regions are still willing to endorse environmental damage in the interest of attracting investment in productive but polluting sectors. He, Huang, and Xu (2015a) expected the uneven economic growth levels between regions along the Xijiang River to exacerbate the transboundary pollution problem because the poor inland provinces, located upstream from the Xijiang river basin, remain willing to sacrifice the environment for growth. These inland provinces are also rich in nonferrous metal reserves, whose extraction practices highly pollute water resources.

Before the WTP questions, the survey first provided the respondents with a general description of the current water quality for the Xijiang River in which the potential contribution of transboundary pollution and the reallocation tendency of polluting industrial production toward upstream cities were explicitly mentioned.

"The Xijiang River drainage basin covers four provinces, Yunnan, Guizhou, Guangxi and Guangdong provinces. Although the water quality of Xijiang River is relatively better than other large rivers in North China, since several years, major pollution incidents frequently happened on its tributaries, affected directly health and safety of people living along the river. Given the rapid economic and social development in cities along the Xijiang River and the already observed tendency of reallocation of polluting industrial production toward upstream cities, many researchers expected large-scale deterioration of water quality in Xijiang River drainage basin in near future."

Then, the current water quality of the section of the Xijiang River flowing through the city where a respondent lived was presented with the help of the water quality ladder inspired by Mitchell and Carson (1989) and adapted to the water quality standard used in China. The river basin-level uniform water quality improvement target proposed by our hypothetical project is fixed at the swimmable level (C level on the water quality ladder illustrated in Figure 6), which corresponds to level II of the Chinese Ministry of Environment Protection classification. The respondents were then asked if they were willing to pay a monthly payment for the realization of this water quality improvement.

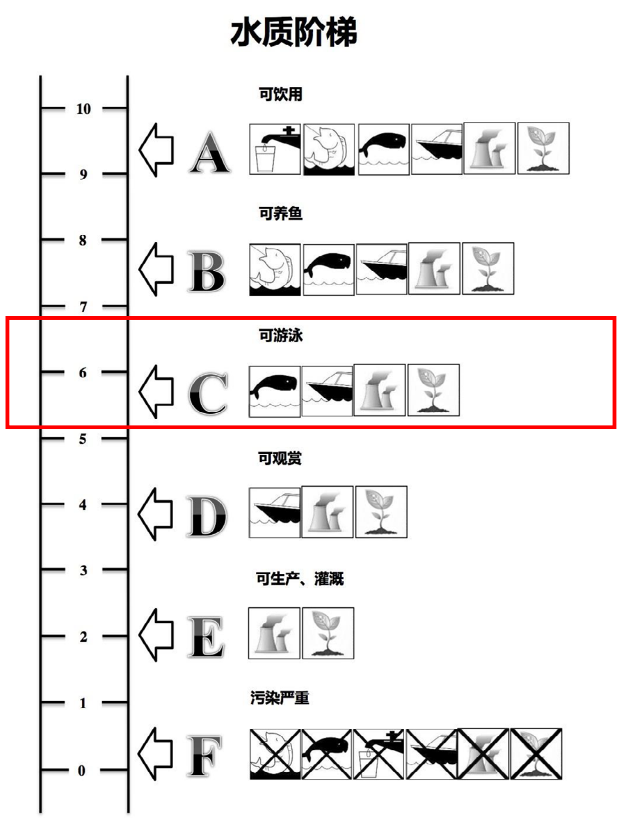



**Figure 6.** Water Quality Ladders

Stratified random sampling was used to ensure an appropriate balance of representativeness across the surveyed cities. The sample size for each city was determined to be roughly proportional to its total population size (cf. Table 1). The recruited samples generally show relatively good gender balance but a higher concentration of younger and more educated people in our sample compared with the general population (cf. Table 2).

**Table 1.** Sample Details for Each City

| City | Population[2] (Million) | Sample Size[1] |
|---|---|---|
| Guangzhou | 12.7 | 110 |
| Shenzhen | 10.4 | 90 |
| Dongguan | 8.2 | 71 |
| Fuoshan | 7.2 | 35 |
| Guiling | 7 | 55 |
| Nanning | 6.7 | 31 |
| Kunming | 6.4 | 48 |
| Qujing | 5.9 | 33 |
| Guiyang | 4.3 | 30 |
| Guigang | 4.1 | 11 |
| Zhaoqing | 3.9 | 35 |
| Liuzhou | 3.8 | 35 |
| Baise | 3.5 | 20 |
| Qiannan | 3.2 | 30 |
| Zhongshan | 3.1 | 11 |
| Wuzhou | 2.9 | 18 |
| Qianxinan | 2.8 | 23 |
| Yunfu | 2.4 | 15 |
| Yuxi | 2.3 | 11 |
| Zhuhai | 1.6 | 15 |
| **Total** | **102.4** | **727** |

[1]After the data cleaning. [2]Data source: Statistics book of sixth national population census of the People's Republic of China. The database illustrated here is the sub-sample (about the half of the total sample) of the survey using the Multiple-Bound Discret Choice format WTP question. The other part of the data that we did not use in this paper is based on a dichotomous choice format WTP question.

**Table 2.** Statistic Descriptives of the Respondents

| Variables | Definition | Mean | S.D. |
|---|---|---|---|
| Age | Years | 34.09 | 10.94 |
| Years of education | Years | 14.56 | 3.13 |
| Income level | Income (1,000 Yuan/month) | 4.44 | 4.1 |
| Income higher than need | Respondents' income can meet the needs of their daily life? (1=yes,0=no) | 0.25 | 0.43 |
| Male | Dummy for male (0=female,1=male) | 0.51 | 0.5 |

The database illustrated here is the sub-sample (about the half of the total sample) of the survey using the Multiple-Bound Discret Choice format WTP question. The other part of the data that we did not use in this paper is based on a dichotomous choice format WTP question.

In Table 3 we reproduced the part of the estimation results of He, Huang, and Xu (2015a) based on the subsample using the MBDC (Multiple Bound Discrete Choice, Wang and He 2011; Welsh and Poe 1998) format WTP questions.[2] The last two estimation functions (hypotheses 2 and 3) illustrated in Table 1 used both individual- and city-level characteristics to explain the monthly WTP. The key variables of our interest are those included in the section called water quality-related variables, where both the water quality of the river flowing through the city of a respondent's residence (degree) and the water quality of the section of river flowing through the direct upstream cities (degree_upper) were included in the explanation of monthly WTP. As can be seen from the estimation called "hypothesis 2", the monthly WTP of a respondent was positively and significantly affected by the water quality of the river crossing his/her resident cities but was negatively related to the water quality of the direct upstream city. The LR test reported at the bottom of Table 1 compared the model, including the variables of the water quality of the direct upstream city, with that excluding such variables. Including the water quality of the upstream cities significantly increased the explanative power of the estimation models and thus supported the relevance of including those variables.

However, the results associated with the variable degree_upper in the model called "hypothesis 2" do not exactly correspond to the new framework that we proposed. This is because the upstream city's water quality was used as the determinant of WTP, not the exact information of the water quality at

2 The other half split questionnaires used the double-bound dichotomous choice (DBDC) elicitation strategies, whose results illustrated obvious bias related to the starting price anchor effect.

the transboundary section of the river. We therefore believe that it is better to use the estimation result of model "hypothesis 3" to establish the payment standard since the cross-term used in this model, degree_upper×, can be considered a proxy for the water quality in the section of the river flowing through the boundary between city j and its direct upstream neighbor k, where is the distance between city k and its direct upstream neighbor j along the river. This term implies the notion of "distance decay" of the river water pollution concentration because of the auto-purification capacity of the river; that is, the further the city j is from its upstream neighbor k, the more the auto-purification function of the river can help to reduce the concentration of the pollution in the section of the river flowing into the city k. Therefore, the further city k is from city j, the lower will be the impact of its transboundary pollution on city j. Our choice was supported by the LR test, which confirmed that the use of the cross-term degree_upper× gave significantly better estimation results than the simple term degree_upper and by the improved statistical significance of the coefficients for the upstream water quality related term.

**Table 3.** Main Estimation Results of He, Huang, and Xu (2015), MBDC Version Data

| | **Hypothesis 1** | **Hypothesis 2** | **Hypothesis 3** |
|---|---|---|---|
| *Individual characteristics* | | | |
| **Bid price** | —0.011 | —0.011 | —0.011 |
| | (38.13)*** | (38.13)*** | (38.13)*** |
| **rep_gov** | —18.436 | —18.666 | —18.159 |
| | (2.19)** | (2.22)** | (2.17)** |
| **water_problem** | 7.080 | 9.700 | 10.930 |
| | (0.95) | (1.29) | (1.46) |
| **will_service** | 15.093 | 13.791 | 13.203 |
| | (3.87)*** | (3.51)*** | (3.37)*** |
| **quality_deg** | 20.245 | 20.217 | 18.151 |
| | (2.67)*** | (2.68)*** | (2.41)** |
| **age** | —0.271 | —0.177 | —0.178 |
| | (0.79) | (0.51) | (0.52) |
| | —0.554 | —0.614 | —0.534 |
| **education** | (0.46) | (0.51) | (0.44) |
| **income_level** | 4.019 | 3.825 | 3.988 |
| | (3.96)*** | (3.76)*** | (3.95)*** |

| | | | |
|---|---|---|---|
| **Income significant higher than need** | 27.176<br>(2.96)*** | 27.677<br>(3.03)*** | 27.058<br>(2.97)*** |
| **male** | —5.829 | —5.859 | —5.975 |
| | (0.81) | (0.81) | (0.83) |
| **Can see the river** | ––25.250 | —24.755 | —24.015 |
| | (2.02)** | (1.98)** | (1.93)* |
| **Far from the river** | —28.295 | —28.315 | —27.186 |
| | (2.38)** | (2.39)** | (2.30)** |
| **d_fish** | 10.770 | 9.815 | 9.611 |
| | (1.20) | (1.10) | (1.08) |
| *City level characteristics* | | | |
| **share2nd** | —0.917 | —1.383 | —1.670 |
| | (2.29)** | (3.06)*** | (3.63)*** |
| **pop_density** | —1.011 | —0.580 | —0.186 |
| | (2.83)*** | (1.43) | (0.43) |
| **gdp_growth** | 9.605 | 10.790 | 9.831 |
| | (4.16)*** | (4.57)*** | (4.29)*** |
| *Water quality related variables* | | | |
| **degree** | 5.134 | 5.264 | 5.990 |
| | (1.65)* | (1.69)* | (1.93)* |
| **degree_upper** | | —5.574 | |
| | | (2.22)** | |
| | | | —13.564 |
| **degree_upper**$\times \frac{1}{1+D_{j,k}}$ | | | (3.24)*** |
| **Constant** | —62.418 | —49.903 | —28.804 |
| | (1.27) | (1.01) | (0.58) |
| **LR** | | 4.09** | 10.45*** |

t statistics are displayed in parentheses, The stars *, ** and *** indicate the significance level at the 15%, 10%, and 1%, respectively.

This is a part of the Table 5 published in He, Huang, and Xu (2015a, P113).

The LR test reported at the bottom of the Table 1 compares the model including the variables of the water quality of the direct upstream city with that excluding such variables.

Based on the models "hypothesis 2" and "hypothesis 3" and the new framework for PES payment standard setting, we calculated two new PES payment schemes for the nine cities located along the main stream of the Xijiang Rivers, as shown in Table 4. For a specific city j, a negative number reported in the column titled  signifies a loss of the welfare of a representative person of its population due to the transboundary pollution flowing from its upstream neighbor j−1. Therefore, we could use this number to time the population size of the city j to obtain its total welfare loss caused by the pollution from its upstream city j−1. To remedy such welfare loss, the amount that the upstream city k needs to transfer to city j should be equal to the absolute value of this product, as illustrated in the last two columns of Table 1.

Additionally, in Table 5, we report the detailed bilateral transfers between cities according to their geographical location and the water quality situations in the transboundary sections between cities. The numbers proposed in the table were calculated according to equations (1) to (5). The upper panel of the table was based on the model "hypothesis 2", and the lower panel was based on the model "Hypothesis 3". A positive number signifies a transfer from city k (upstream) to city j (downstream) to remedy the negative externality caused by k to j, whereas a negative number means a transfer from city j (downstream) to city k (upstream) for the improvement of the water quality and, thus, a positive externality.

There are several interesting and logical findings from the comparison between the results based on models "hypothesis 2" and "hypothesis 3". For example, the "distance decay" nature of river water pollution can significantly reduce the compensation burden for the upstream polluters. Additionally, for a city located a very small distance along the river from its very-populous direct downstream neighbor, even a small amount of transboundary pollution signifies a large amount of compensation. In both cases, the distance plays a very important role in the determination of the compensation amount.

## 6. Discussion and Conclusion

The new payment standard setting framework proposed in our paper directly concentrates measurements on the negative or positive externality caused by transboundary water pollution. This is very different from most of the previously mentioned preference studies, which focused on measuring the impacts of isolated and hypothetical quality changes in ecological services on people's welfare that were rarely related to the transboundary pollution context.

Based on the paper of He, Huang, and Xu (2015a), our new payment standard setting framework can propose both the total transfers that a city should make as a polluter or receive as a victim. This new framework also allows the calculation of the detailed bilateral monetary transfer between cities, depending on their location on the river and their contribution to the variation of water quality. One advantage of our approach is the possibility to not only

**Table 4.** The Loss of Welfare due to Transboundary Water Pollution and the Total Transfer to Remedy Such Negative Externality

| Location of the city j on main tributary (1=most upstream, 9=most downstream) *in the bracket (water quality, population in million) | | | | | Individual mean WTP (Yuan) | | | Total transfer (\|ΔWTP\|×population, million Yuan) from city j–1 to j | |
|---|---|---|---|---|---|---|---|---|---|
| j | City j | Water quality | Population (million) | Distance to upstream neighbor (km) | WTP reported in He et al. (2015a) | $\Delta$WTP = $W_j(Q_k - II)$, k: direct upstream neighbor | $\Delta$WTP = $W_j\left(\frac{Q_k - II}{1+D_{j,k}/100}\right)$, k: direct upstream neighbor | Based on model "hypothesis 2", where $\Delta$WTP = $W_j(Q_k - II)$ | Based on model "hypothesis 3", where $\Delta$WTP = $W_j\left(\frac{Q_k - II}{1+D_{j,k}/100}\right)$ |
| 1 | Qujing | IV | 6.4 | — | 61.5 | — | — | — | — |
| 2 | Guigang | II | 3.8 | 268 | 78.1 | −10.15 | −7.37 | 38.57 | 28.01 |
| 3 | Wuzhou | III | 2.9 | 261 | 70 | 0 | 0 | 0 | 0 |
| 4 | Yunfu | III | 2.4 | 100.2 | 20.5 | −5.57 | −6.78 | 13.37 | 16.27 |
| 5 | Zhaoqing | II | 3.9 | 67.9 | 75.1 | −5.57 | −8.08 | 21.72 | 31.52 |
| 6 | Foshan | IV | 7.2 | 81.12 | 64.9 | 0 | 0 | 0 | 0 |
| 7 | Zhongshan | II | 3.1 | 133 | 29.3 | −10.15 | −11.64 | 31.47 | 36.08 |
| 8 | Guangzhou | IV | 12.7 | 31.69 | 63.6 | 0 | 0 | 0 | 0 |
| 9 | Zhuhai | IV | 1.6 | 38.6 | 36.3 | −10.15 | −19.57 | 16.24 | 31.32 |

**Table 5.** The Details of the Bilateral Transfer Between Cities

| **Based on model "hypothesis 2"** | | | | | | Cities k | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| City j | $\Delta WTP = W_j(Q_k - II)$, k: direct upstream neighbor | Water quality | Population (million) | Distance to upstream neighbor (km) | Total transfer received | PES transfer matrix $W_{j,k}$ (positive values means transfer from city k (upstream) to city j (downstream), negative values means transfer from city j (downstream) to city k (upstream)on the line to city in the column) | | | | | | | | |
| | | | | | | Qujing | Guigang | Wuzhou | Yunfu | Zhaoqing | Foshan | Zhongshan | Guangzhou | Zhuhai |
| Qujing | — | IV | 6.4 | — | | — | — | — | — | — | — | — | — | — |
| Guigang | −10.15 | II | 3.8 | 268 | 38.57 | 38.57 | — | — | — | — | — | — | — | — |
| Wuzhou | 0 | III | 2.9 | 261 | 0 | 29.44 | −29.44 | — | — | — | — | — | — | — |
| Yunfu | −5.57 | III | 2.4 | 100.2 | 13.37 | 24.36 | −24.36 | 13.37 | — | — | — | — | — | — |
| Zhaoqing | −5.57 | II | 3.9 | 67.9 | 21.72 | 39.59 | −39.59 | 21.72 | 0 | — | — | — | — | — |
| Foshan | 0 | IV | 7.2 | 81.12 | 0 | 73.08 | −73.08 | 40.10 | 0 | −40.10 | — | — | — | — |
| Zhongshan | −10.15 | II | 3.1 | 133 | 31.47 | 31.47 | −31.47 | 17.27 | 0 | −17.27 | 31.47 | — | — | — |
| Guangzhou | 0 | IV | 12.7 | 31.69 | 0 | 128.91 | −128.91 | 70.74 | 0 | −70.74 | 128.91 | −128.91 | — | — |
| Zhuhai | −10.15 | IV | 1.6 | 38.6 | 16.24 | 16.24 | −16.24 | 8.92 | 0 | −8.92 | 16.24 | −16.24 | 16.24 | — |
| Total payment due to the responsibility of transboundary pollution of city k | | | | | | 381.66 | −343.09 | 172.12 | 0 | −137.03 | 176.62 | −145.15 | 16.24 | — |
| | | | | | | | | | | | | | | |
| **Based on model "hypothesis 3"** | | | | | | Cities k | | | | | | | | |
| City j | $\Delta WTP = W_j\left(\frac{Q_k - II}{1+\ln(D_{j,k})}\right)$, k: direct upstream neighbor | Water quality | Population (million) | Distance to upstream neighbor (km) | Total transfer received | PES transfer matrix $W_{j,k}$ (positive values means transfer from city k (upstream) to city j (downstream), negative values means transfer from city j (downstream) to city k (upstream)on the line to city in the column) | | | | | | | | |
| | | | | | | Qujing | Guigang | Wuzhou | Yunfu | Zhaoqing | Foshan | Zhongshan | Guangzhou | Zhuhai |
| Qujing | — | IV | 6.4 | — | — | — | — | — | — | — | — | — | — | — |
| Guigang | −7.37 | II | 3.8 | 268 | 28.01 | 28.01 | — | — | — | — | — | — | — | — |
| Wuzhou | 0 | III | 2.9 | 261 | 0 | 12.51 | −12.51 | — | — | — | — | — | — | — |
| Yunfu | −6.78 | III | 2.4 | 100.2 | 16.27 | 8.93 | −8.93 | 16.27 | — | — | — | — | — | — |
| Zhaoqing | −8.08 | II | 3.9 | 67.9 | 31.52 | 13.27 | −13.27 | 19.73 | 11.79 | — | — | — | — | — |
| Foshan | 0 | IV | 7.2 | 81.12 | 0 | 22.24 | −22.24 | 27.96 | 11.02 | −38.98 | — | — | — | — |
| Zhongshan | −11.64 | II | 3.1 | 133 | 36.09 | 8.31 | −8.31 | 8.72 | 2.05 | −10.77 | 36.09 | — | — | — |
| Guangzhou | 0 | IV | 12.7 | 31.69 | 0 | 33.04 | −33.04 | 33.52 | 8.12 | −41.64 | 153.33 | −153.33 | — | — |
| Zhuhai | −19.57 | IV | 1.6 | 38.6 | 31.32 | 4.01 | −4.01 | 3.92 | 0.86 | −4.78 | 14.33 | −14.33 | 31.32 | — |
| Total payment due to the responsibility of transboundary pollution of city k | | | | | | 130.32 | −102.31 | 110.12 | 33.84 | −96.17 | 203.75 | −167.66 | 31.32 | — |

identify polluters and victims but also "cleaners" who inherit bad water quality from the upstream neighbor and clean it up. The compensation regime proposed in our approach can accordingly determine not only the compensation for negative externalities to be paid by the polluters to victims but also a compensation from the "victim-to-be" to the up-stream cleaners for their efforts that create positive externalities and avoid potential welfare loss. We believe such two-direction compensation systems can more efficiently discourage the creation of negative externality and encourage that of positive externality, both of which contribute to better river water quality.

It is difficult to directly compare the numbers proposed in our approach with those from previous studies. One reasonable comparison that we can make is between the compensation that we proposed in this paper and the aggregate WTP reported in He, Huang, and Xu (2015a) for the achievement of the targeted Class II level river water in corresponding cities. The latter can be regarded as an example similar to most of the previous valuation studies that have used the WTP as the compensation standard for a better ecological service quality. Referring to the three columns under the individual WTP in Table 2, we can make the general observation that using the total WTP for a better ecological service quality risks mixing up the pollution caused by the upstream cities and the pollution from a city's own activities and thus tends to exaggerate the necessary compensation payments. According to Table 5, such exaggeration ranged from 2 to 10 times for the Xijiang River. Another possible comparison is with the pilot PES project at the Xin'an River, in which the transfer between two provinces is arbitrarily fixed at 500 million Yuan per year. Taking the potential necessary transfer between Foshan and Zhongshan cities as an example, the total yearly transfer is already equal to 86% (36.09*12 months) of the total transfer between Zhejiang and Anhui provinces. Shen et al. (2015) advocated the necessity to increase the transfer amount for the Xin'an River PES pilot to reinforce the water protection motivation of both provinces; our paper can be considered as a supportive argument for their policy recommendation, although we admit the potentially big difference between the Xin'an and Xijiang Rivers.

Another advantage of the new approach for payment standard setting is to directly associate the compensation amount that a city needs to pay (for negative externality) or to receive (for positive externality) with the size of the victims/beneficiaries of the related externality. From the point of view of efficiency, for a specific city, the further it is located toward the upstream end of the river, the larger will be the size of its potential victims/beneficiaries and thus the higher will be the amount of compensation to pay or to receive if it creates negative or positive externalities. Such logic, acting with more emphasis on the more upstream cities, can largely contribute to efficiency of the control of transboundary pollution and thus facilitates the realization of the water quality improvement targets of an entire river.

From equity point of view, as most of upstream cities located in the inland China are also at the same time less developed cities, the further is the city located in the upstream end, further its river cleanup effort will be recognized and well compensated.

The numbers proposed in our study for the PES standard are certainly specific to the case of the Xijiang River. Although it is technically feasible to use the estimated coefficients from the models reported in Table 1 to extrapolate the impact of transboundary pollution on the variation of people's WTP in specific cities, such extrapolation still produces biases. These biases can come from the fact that different rivers present different bio-physical characteristics or that the cities located along a river may have particular geographical patterns and specific mutual economic relationships. Admitting that not all these specificities can be considered with the coefficients obtained from cities belonging to another river drainage basin, we welcome more high-quality stated preference studies to be conducted and used as bases for the proposal of PES payment standards. Such measures will make it possible to compare the results from different regions and river basins and thus facilitates a more practical discussion about whether it is reasonable to extrapolate the results for one drainage basin to another.

## References

Bateman, Ian J., Georgina M. Mace, Carlo Fezzi, Giles Atkinson, Kerry Turner (2011). Economic Analysis of Ecosystem Service Assessments. Environmenal and Resource Economics, 48: 177-218.

Cai, B., G. Lu, and L. Song. 2008. "The Ecological Compensation Standardization for Ecological Restoration: A Case Study of Ecological Restoration in the Water Resource Protection Zone for Eastern South-to-North Water Transfer Project." *ACTA Ecologica Sinica* 28 (5): 2413-16. (In Chinese)

Cai, W., S. Yang, S. Wang, and N. Zhou. 2010. "A Study on Externality Problems and Countermeasures of Establishing Wetlands Nature Reserve: Take Tianjin Qilihai Wetland as an Example." *Journal of Chongqing University (Social Science Edition)* 16 (6): 10-15. (in Chinese)

Desaigues, B., and P. Point. 1993. *Économie du Patrimoine Naturel. La valorisation des bénéfices de protection de l'environnement,* 317 p. Paris: Economica.

Fu, Y., B. Ruan, and C. Zhang. 2011. "Research on the Ecological Compensation Standard of Yongding River Basin." *Journal of China Institute Water Resource and Hydropower Research* 9 (4): 283-91. (In Chinese)

Harrison, G.W., and E.E. Rutström. 2008. "Experimental Evidence on the Existence of Hypothetical Bias in Value Elicitation Methods." In *Handbook of Experimental Economics Results,* edited by C.R. Plott, and V.L. Smith, 752-67. New York: North Holland.

He, J., A. Huang, and L. Xu. 2015a. "Spatial Heterogeneity and Transboundary Pollution: A Contingent Valuation (CV) Study on the Xijiang River Drainage Basin in South China." *China Economic Review* 36 (2015): 101-30.

He, J., F. Moffette, R. Fournier, J.-P., Revéret, J. Théau, J. Dupras, and J.-P. Boyeret Mathieu Varin. 2015b. "Meta-Analysis for the Transfer of Economic Benefits of Ecosystem Services Provided by Wetlands within Two Watersheds in Quebec, Canada." *Wetland Ecology and Management* 23 (2015): 707-25.

Helland, E., and A.B. Whitford. 2003. "Pollution Incidence and Political Jurisdiction: Evidence from the TRI." *Journal of Environmental Economics and Management* 46 (3): 403-24.

Hills, P., and P. Roberts. 2001. "Political Integration, Transboundary Pollution and Sustainability: Challenges for Environmental Policy in the Pearl River Delta Region." *Journal of Environmental Planning and Management* 44 (4): 455-73.

Huang, C., B. Luo, and K. Yang. 2008. "Water Conservation Function of Forest and Establishment of Its Ecological Compensation System." *Journal of Anhui Agricultural Sciences* 36 (20): 8599-601, 8627. (In Chinese)

Jin, R., and X. Wang. 2008. "Evaluation of the Ecological Recovery Eco-compensation in Qilian Mountain Water Conservation Forest Area: A Case Study of Shandan County." *Yunnan Geographic Environment Research* 20 (1): 60-64. (In Chinese)

Li, X., H. Miao, H. Zheng, Z. Ouyang, and D. Xiao. 2009. "Application of Opportunity-cost Method in Determining Ecological Compensation Standard: A Case Study in the Central Mountainous Area of Hainan Island." *ACTA Ecologica Sinica* 29 (9): 4875-83. (In Chinese)

Lipscomb, M., and A.M. Mobarak. 2017. "Decentralization and Water Pollution Spillovers: Evidence from the Redrawing of County Boundaries in Brazil."*Review of Economic Studies,* 84: 464-502.

List, J.A., and C. Gallet. 2001. "What Experimental Protocol Influence Disparities Between Actual and Hypothetical Stated Values?." *Environmental and Resource Economics* 20: 241-54.

Maler, K.G. 1974. *Environmental Economics.* Baltimore: John Hopkins University.

Ministry of Environment Protection. 2013. "Report of State Council About the Construction of the Payment for Ecological Service Regime in China." April 24, 2013. (In Chinese) http://www.mep.gov.cn/gzfw_13107/zcfg/hjjzc/gjfbdjjzcx/stbczc/201606/t20160623_355510_wap.shtml.

Murphy, J.J., T.H. Stevens, P.G. Allen, and D. Weatherhead. 2005. "A Meta-Analysis of Hypothetical Bias in Stated Preference Valuation." *Environmental & Resource Economics* 30 (3): 313-25.

Navrud, S., and R. Ready. 2007. "Lessons Learned for Environmental Value Transfer." In *Environmental Value Transfer: Issues and Methods,* edited by S. Navrud, and R. Ready, 283-90. Dordrecht, The Netherlands: Springer.

Oates, W.E., and P.R. Portney. 2003. "Chapter 8: The Political Economy of Environmental Policy." In *Handbook of Environmental Economics, vol. 1* (January): 325-54. Elsevier: North Holland.

Samuelson, P. 1954. "The Pure Theory of Public Expenditure." *The Review of Economics and Statistics* 36 (4): 387-89.

Sander, L.D., R.G. Walsh, and J.B. Loomis. 1990. "Toward Empirical Estimation of the Total Value of Protecting Reviers." *Water Resources Research* 26 (7): 1345-57.

Shen, G., L. Huang, X. Qian, D. Pan, S. Shi, and M.L. Gullino. 2009. "Ecological Compensation Criteria for Enviromental-Friendly Agriculture Production—Case Study of Green Agriculture Demonstration Project in Dongtan, Chongming Island." *Journal of Agro-Environment Science* 28 (5): 1079-84. (In Chinese)

Shen, M. et al. 2015. "Policy Suggestions for the Establishment of an Efficient Transboundary Water Environment Protection Regime Over the Xin'an River Drainage Basin." http://www.hzjczx.gov.cn/NewsDetail-566.shtml. (In Chinese)

Shi, X, X. Hu, X. Zou, M. Lin, and J. Chen. 2012. "Ecological Compensation Criteria Based on Supply Cost in Source Area of Dongjiang River." *Water Resources Protection* 28 (2): 77-81. (In Chinese)

Sigman, H. 2002. "International Trade and Pollution in Shared Resources: A Study of International Rivers." *American Economic Review* 92 (4): 1152-59.

Sigman, H. 2005. "Transboundary Spillovers and Decentralization of Environmental Policies." *Journal of Environmental Economics and Management* 50 (January): 82-101.

Wang, H., and J. He. 2011. "Willingness-to-Pay Estimation with Multiple Bounded Discrete Choice Data." *Applied Economics* 43 (19–21): 2641-56.

Wattage, P., and S. Mardle. 2008. "Total Economic Value of Wetland Conservation in Sri Lanka Identifying Use and Non-use Values." *Wetlands Ecology and Management* 16 (5): 359-69.

Welsh, M.P., and G.L. Poe. 1998. "Elicitation Effects in Contingent Valuation: Comparisons to a Multiple Bounded Discrete Choice Approach." *Journal of Environmental Economics and Management* 36 (1998): 170-85.

Xu, L., Z. Yang, L. Shuai, J. Yu, and S. Liu. 2006. "Eco-compensation of Reservoir Project based on Ecosystem Service Function Value." *China Population, Resources and Environment* 16 (4): 125-28. (In Chinese)

Yu, X. 2011. “Transboundary Water Pollution Management Lessons Learned from River Basin Management in China, Europe and the Netherlands.” *Utrecht Law Review* 7 (1): 188-203.

Zhang, T. 2011. “Research on the Calculation of the Eco-compensation Payment Standard of the Water Source Location of Xijing River Drainage Basin.” (Xi Jiang Liu Yu Shui Yuan Di Sheng Tai Bu Chang Biao Zhun Ce Suan Yan Jiu). *Guizhou Social Science* 2011 (9): 76-79. (In Chinese)

# Subjective and Objective Air Quality in Urban China: The Moderating Effect of Environmental Transparency

Liang Ma[1*] and Wenxuan Yu[2]

Abstract

Due to fast industrialization and sweeping urbanization in China, environmental pollutions have been jeopardizing China's economic and biological sustainability. Environmental pollutions have ignited public outcry and social unrest, which may undermine the ruling party's regime support and legitimacy. Citizen's political actions related to environmental degradation are largely determined by their perceptions of environmental pollutions, which are subjective and socially constructed. In this study, we use data from various sources (nationwide citizen survey, government statistics, and external assessments) and employ a multilevel modeling strategy to empirically explore the antecedents of citizens' perceptions of air quality. We examine to what extent objective air quality and environmental information availability (transparency) jointly affect citizens' perceptions of air quality. After controlling for a variety of confounding variables, we find that subjective air quality is positively related to objective air quality and environmental transparency negatively moderates this relationship. The findings generate significant theoretical and practical implications for environmental policy, government performance measurement, and transparency.

***Keywords:*** environmental pollution, air quality, China, subjective performance measurement, government transparency

# Calidad del aire subjetiva y objetiva en la China urbana: el efecto moderador de la transparencia ambiental

Resumen

Debido a la rápida industrialización y urbanización radical en Chi-

[1] *School of Public Administration and Policy, Renmin University of China*

[2] *School of Public Affairs and School of Public Policy, Xiamen University, China*

[*] *Corresponding author:* liangma@ruc.edu.cn


doi: 10.18278/cpj.1.1.4

na, la contaminación ambiental ha estado poniendo en peligro la sostenibilidad económica y biológica de China. La contaminación ambiental ha provocado indignación pública y malestar social, lo que puede socavar el apoyo y la legitimidad del régimen del partido gobernante. Las acciones políticas de los ciudadanos relacionadas con la degradación ambiental están en gran medida determinadas por sus percepciones de las contaminaciones ambientales, que son subjetivas y socialmente construidas. En este estudio, utilizamos datos de varias fuentes (encuesta nacional de ciudadanos, estadísticas gubernamentales y evaluaciones externas) y empleamos una estrategia de modelado multinivel para explorar empíricamente los antecedentes de las percepciones de los ciudadanos sobre la calidad del aire. Examinamos hasta qué punto la calidad del aire objetivo y la disponibilidad de información ambiental (transparencia) afectan conjuntamente las percepciones de los ciudadanos sobre la calidad del aire. Después de controlar una variedad de variables de confusión, encontramos que la calidad subjetiva del aire se relaciona positivamente con la calidad objetiva del aire y la transparencia ambiental modera negativamente esta relación. Los hallazgos generan importantes implicaciones teóricas y prácticas para la política ambiental, la medición del desempeño del gobierno y la transparencia.

***Palabras clave:*** contaminación ambiental, calidad del aire, China, medición del desempeño subjetivo, transparencia gubernamental

# 中国城市主客观空气质量：环境透明度的调节效应

摘要

由于中国的快速工业化和大规模城市化，环境污染已经威胁到中国的经济和生物可持续性。环境污染已经激发公众抗议，引起社会动荡，这可能会损坏执政党的政权支持力度和合法力度。公民关于环境退化所开展的政治行动在很大程度上取决于他们对环境污染的看法，而这一看法是由主观的和社会建构的。在本项研究中，笔者使用不同来源的数据(全国公民调查、政府统计和外部评估)，并采用多层建模策略对于公民对空气质量的看法成因进行了实证研究。笔者基于客观空气质量和环境信息获取(透明度)在多大程度上共同影

响公民对空气质量的看法进行了探索。在对各种混杂变量进行控制后，笔者发现主观空气质量与客观空气质量呈正相关，而环境透明度对此关系有负向调节作用。该项研究结果对环境政策、政府绩效衡量和透明度产生了重大的理论和实践意义。

关键词：环境污染，空气质量，中国，主观绩效衡量，政府透明度

## Introduction

Due to fast industrialization and sweeping urbanization in China, environmental pollutions have been jeopardizing China's economic and biological sustainability. Environmental pollutions have ignited public outcry and social unrest, which may undermine the ruling party's regime support and legitimacy. Citizen's political actions related to environmental degradation are largely determined by their perceptions of environmental pollutions, which are subjective and socially constructed. Existing studies on air pollution in China in social sciences, however, are mainly conducted by ecologists and economists, considering air pollution as independent variables and using objective air quality indicators from government archives to examine its impacts on public health, happiness, life satisfaction, and political actions such as social protest and emigration (e.g., Qin and Zhu 2018; Smyth, Mishra, and Qian 2008; Wang and Cheng 2017). With few exceptions (Li and Xue 2016; Shi 2015), studies on perceived environmental pollution in China are lacking.

Citizens' perceptions of air quality are a social and psychological construct, jointly influenced by environmental pollution and information availability. The objective measurement and subjective perception of air pollution are conceptually different and empirically discernable, and it is of theoretical relevance and policy importance to examine the discrepancies between subjective and objective environmental quality. A recent study reveals that perceived government's efforts in addressing environment issues significantly influence residents' environmental perceptions in a coal-mining region in northern China. Controlling for government efforts, however, objective environmental indicators do not significantly impact subjective environmental perceptions (Shi 2015).

In this study, we focus on one of the government efforts in addressing environmental pollution, information transparency. Environment transparency is considered as an essential part of global environmental governance (Gupta 2010). It is believed that environmental transparency can protect individuals from environmental harms, enhance environmental enforcement, and facilitate social learning (Li and Li 2012). Environmental transparency is one of the most important policy tools the central government is using to monitor and evaluate the performance of local governments in environmental protection so as to enforce environment laws and regulations.

Given the importance of subjective measures of public services and government performance in public administration research (Schachter 2010; Shingler, Van Loon, and Alter 2008), we aim to examine two research questions with significant theoretical and practical implications:

(1) To what extent do people's perceptions of air quality reflect the authentic air quality measured by government hard data in China?

(2) What policy instruments government can take to influence people's perceptions of air pollution?

In this study, we use data from various sources (nationwide citizen survey, government statistics, and external assessments) and employ a multilevel modeling (MLM) strategy to empirically explore the antecedents of citizens' perceptions of air quality. Specifically, we examine to what extent objective air quality and environmental information availability (transparency) jointly affect citizens' perceptions of air quality. After controlling for a variety of confounding variables, we find that subjective air quality is positively related to objective air quality and environmental transparency negatively moderates this relationship. The findings generate significant theoretical and practical implications for environmental policy, government performance measurement, and transparency.

The remainder of this article is structured as follows. First, we discuss the relationship between objective air pollution and subjective perceptions of air quality. Second, we review the environmental transparency literature, discussing how environmental transparency in China would influence subjective air pollution. Third, we report our data collection and research methods. Fourth, we present and discuss our findings. Lastly, we conclude with theoretical and policy implications, limitations, and future research avenues.

## Context

### *Air Pollution in China*

Since China's "Reform and Open-up" policy in the late 1970s, China has achieved stunning economic achievement. In 2010, China overtakes Japan as World's No. 2 economy. China is also experiencing unprecedented urbanization. In 2015, more than 55.6 percent of Chinese lived in urban area (CIA World Factbook 2017). It is no doubt that economic development and urbanization have drastically improved Chinese people's economic income and quality of life. However, similar to other developing countries such as India, China is also experiencing devastating environmental pollutions (Albert and Xu 2016). China has 16 of the world's 20 most polluted cities (World Bank 2007), and environmental pollutions have already become one of the most pressing threats to China's economic sustainability and social harmony.

According to an estimate of China's Ministry of Environmental Protection (MEP) in 2010, environmental pollution costs China around 1.5 trillion RMB (227 billion U.S. dollars), or roughly 3.5 percent of its gross domestic product. Not only does environmental pollution thwart economic development, environmental pollution also produces detrimental effects on public health. Studies estimate that around 11 percent of digestive-system cancers in China may stem from unsafe drinking water (He, Fan, and Zhou 2016). A recent study shows that air pollution has caused significant health consequences, including respiratory, cardiovascular, and cerebrovascular diseases, in northern China since the 1980s (Chen et al. 2017).

Among various types of pollution, air pollution has produced most visible negative economic, social, and political impacts. Air pollution has led to social unrest and collective actions, threating political trust, and undermining legitimacy (Albert and Xu 2016). Air pollution has pushed people to migrate or emigrate (Qin and Zhu 2018). Therefore, fighting again air pollution has become one of the top priorities of the central government. Local governments have also been mobilized to fight against haze. After a lasting period of "airpocalypse" in Beijing in 2013, for instance, a senior Beijing municipal government official vowed on his own head to control the choking haze.

However, Chinese government is facing tremendous challenges in addressing air pollution issues due to various reasons. First, air pollution in China is a "wicked problem" (Rittel and Webber 1973). The components and the causes of air pollution are very complex due to China's large territory and vast differences in demographic, geographic, economic, and industrial characteristics among regions.

In the past four decades, China has experienced unprecedented fast industrialization and urbanization. However, China's economic growth heavily relies on the consumption of natural resources, energy, and cheap human labor. Due to the scarcity of other natural resources and technology deficiency, coal is still the dominant source of energy (Zhang and Crooks 2012). China is the largest coal producer in the world and produces around half of global consumption (Bawa et al. 2010). Although China's National Energy Agency claimed that coal use had been declining, international observers doubted the claim because of the increased coal power plant capacity in 2015 (Albert and Xu 2016). In the past 20 years, car ownership has skyrocketed with the fast urbanization. In 2016, China has 172 million cars (Xinhua 2016). Rapid urbanization and car ownership significantly increase energy consumption and emissions, which in turn jeopardize air pollution (Liu and Diamond 2005).

Second, air pollution is the most visible pollution, and its threat to economic sustainability and public health is equivalent to (if not higher than) other type of pollutions (e.g., water and soil). Due to its visibility, however, public complaints and outcry distract government attention from addressing other more serious environmental challenges. Also, government takes fragmented, campaign-style, and short-term superficial measures instead of holistic and fundamental strategies to address air pollution (Dasgupta and Wheeler 1997; Dong et al. 2011).

Last but not the least, curbing air pollution is a double-edged sword. Air pollution is the consequence of China's fast industrialization and economic development. On the one hand, Chinese people are the victim of air pollution; on the other hand, they are also the beneficiaries of fast economic development. Studies found that Chinese believe economic development is more important than environmental protection and such attitudes are rooted in the nation's long history of poverty, resulting in strong desire for material wealth (Harris 2006). China's environmental pollution is deeply intertwined with other problems related to industrial structure, energy consumption structure, and the model of economic development. China still has a long way to go in curbing environmental pollution.

In addition, how Chinese people perceives air pollution is further complicated by the fact that the causal mechanisms between air pollution and health consequences are ambiguous, and the negative effects of air pollution on public health are often long term and chronical (Holdaway 2013). Moreover, previous environmental studies suggest that demographic characteristics such as age, gender, education, residence, and social economic status may also influence people's environmental perceptions (Daneshvary, Daneshvary, and Schwer 1998; Ebreo, Hershey, and Vining 1999; Howell and Laska 1992; Xiao and Hong 2010). Therefore, it is more meaningful to study how Chinese people's perceptions of air pollution affect their political attitudes and actions than studying the static and "objective" air quality indicators.

### *Environmental Transparency*

The central government has considered environmental pollution as its top policy priority due to the devastating and detrimental effects of environmental pollution on environmental sustainability, public health, and political trust and legitimacy (Economy 2010). However, the implementation of environmental protection policy is largely thwarted by China's unique political system characterized by fragmented and decentralized authoritarianism (Lieberthal 1997). Local functional departments and environmental departments need to report to and under the control of both central and local governments. The fragmented and decentralized power of environment protection challenges the efforts of the central government in striking a balance between economic development and environmental protection (Tan 2014).

Given the importance of people's perceptions of air pollution and the difficulties of addressing the "wicked problem" in the short run, it is crucial to manage subjective air quality apart from actually improving air quality.

As a policy tool to break the deadlock, Chinese central government has given high hope to environmental transparency. The Regulations on Open Government Information was enacted by the State Council in 2007 and took effective in 2008. According to the Regulations, governments are required to disclose public information and citizens are entitled to request government information. Almost at the same time, The National Bureau of Environmental Protection (the predecessor of Ministry of Environmental Projection, MEP) promulgated the Measures on Open Environmental Information (OEI) in 2008 to mandate local environmental protection bureaus (EPBs) to disclose environmental information.

For individual citizens, environmental transparency can educate citizens to realize the importance of environmental sustainability and proactively take measures to protect themselves from pollution. Pollution is more than a pure objective fact. It is actually social constructed. The definition of pollution changes with people's understanding of the causes and consequences of environmental changes. With more environmental information, the public is informed and educated to understand the consequences of economic growth and development so as to make an informed decision to adjust their expectations toward pollution. Their attitudes or behaviors would change accordingly. In China's unique historical and cultural context characterized by government secrecy, disclosing environmental information to the public exhibits government's genuine wiliness and commitment in constraining corruption and abuse of power and controlling pollution (Yu 2011).

In China, due to the lack of a vibrant civil society, government monopolies environmental information provision. However, citizens themselves can also acquire information from other channels (Fung, Graham, and Weil 2007). When government information is absent, information from other sources would significantly influence their perceptions on government performance in a way government may not like. In 2009, the U.S. Embassy in Beijing began to provide air quality information to its employees and American expatriate due to the frequent appearance of choking haze. The publicity of air quality assessment went viral in social media. After accusing the Embassy for intervening China's internal affairs, the MEP was forced to release air quality information. However, the continuous inconsistence between the two data sources further jeopardized public confidence and trust in the MEP (Ma and Zhang 2015). What is even worse is that the information released by government and other sources (including citizens themselves) is inconsistent or mutual conflicting, which may result in citizens' discontent and distrust toward government.

## Theoretical Hypotheses

### *Objective and Subjective Air Quality*

In public administration literature, since 1970s, a stream of literature has examined the congruence between objective and subjective performance indicators (Miller and Miller 1991). Due to the differences in conceptualization, measurement, and model specification, the debate has continued and research findings are mixed. A stream of studies tries to empirically show that subjective government performance are influenced by individual demographic characteristics such as gender, age, education and race, and individual use experience and expectation (Brown and Coulter 1983; Stipak, 1979, 1980; Swindell and Kelly 2000; Van Ryzin and Immerwahr 2008).

Stipak (1979) argued that many public services are rarely used by citizens. The relationship between subjective indicators and objective indicators is confounded by many statistical problems. Brown and Coulter (1983) empirically explored the relationship between residents' perceived police performance (e.g., policy response time, police treatment of people, and police service quality) and corresponding objective indicators, finding that subjective indicators are not significantly related to objective indicators. However, other scholars disagreed that previous studies have various methodological problems. At least in public service areas such as park maintenance and street cleaning, if outcome-oriented objective indicators and a more scientific research design are adopted, subjective and objective indicators are significantly related, which means that citizen's perception of government performance can reflect government genuine performance (Andrews et al. 2011). Using New York public school data, Charbonneau and Van Ryzin (2011) found that parents' subjective assessment of education quality corresponds fairly well with objective school performance measures.

Proving public goods and service, and regulating market are two essential functions of government. Air is a pure public good. Without government regulation, due to air pollution's externality and spillover effects, "the tragedy of commons" takes place (Burger and Gochfeld 1998). Air quality is considered as a sign of government regulatory capacity and an essential indicator of government performance (Holzer and Yang 2004). Different from other public services such as park and library; nobody can escape from enjoying the "service." Air pollution is highly sensible and visible. However, it does not necessarily mean subjective perceptional measures are reliable in measuring air quality.

Perception of air pollution is socially constructed by individuals and societal characteristics, as well as their interactions (Li and Li 2012). Human body is an adaptive system, and long-time exposure to pollution could change the psychological and biological formula for evaluating air quality. Objective measures of air pollution are based on a universal set of scientific measures according to a set of universal scientific "truth" of public health. In contrast, regular people do not have professional equipment and knowledge to detect air quality. In addition to their personal experience, their perception of air quality is highly influenced by information provided by mass media and social media (Liu, Dong, and Wang 2011). Therefore, visible and sensible experience of residents is based on but not equivalent to the objective level of air quality.

Hadrich and Wolf (2011) studied the environmental pollution by Michigan's livestock operations and citizen complains. They found that compared with surface water pollution, odor pollution was more difficult to be verified. In China, although after the 2013 pandemic air pollution in Beijing, the municipal government had promised to clean the air, in the summer of 2016 local dwellers' complaints on the hovering haze raged the social media, blaming government's incompetence and inaction. However, according to objective scientific data, air quality in Beijing has significantly improved over years. In order to establish the significant relationship between objective and subjective air quality, we need empirical studies based on solid data, representative sample, and rigorous research design. Although there are no empirical studies specifically testing the relationship, some research on citizen environmental complaints and environmental pollution found that citizens' complaints are significantly related to air pollution (Dasgupta and Wheeler 1997; Dong et al. 2011). Therefore, we develop the first hypothesis as below.

*Hypothesis 1: Subjective air quality significantly correlates with objective air quality.*

### *The Moderating Effect of Environmental Transparency*

In addition to the influence of objective air pollution, subjective air pollution could also be affected by their expectation, knowledge, information availability, and political attitudes, which are influenced by environmental transparency. Transparency refers to the availability and usability of government information to the public, and it is subtly different from openness and information disclosure (Wu, Ma, and Yu 2017). Openness means the disclosure of government information, which might not be equivalent to transparency. For instance, government may discretionarily and selectively disclose some information while keep others (e.g., politically sensitive data) opaque. Government may also purposely distort and manipulate the information disclosed to the public. Transparency, in contrast, means government information is not only disclosed and available to the public, but also citizens can access, understand, interpret, and use the information for private or public purposes (Fung, Graham, and Weil 2007).

Transparency is not only about the disclosure and use of government information, but also reflects the motivations and capacities of the government in addressing air pollution. Given the professionalism of air quality monitoring, the information on air quality is to some extent controlled by the government. Whether citizens can get access to and utilize this information partially depends on government interventions, and local governments who are willing to disclose and share environmental information are favorably supported by citizens. If local governments can honestly disclose environmental information and proactively adopt policies to address environmental pollution, then citizens are more likely to support their policies. Despite environmental pollution might not be substantially reduced, citizens will tolerate their sluggish improvement and perceive environmental quality in a more lenient way.

We argue that citizens' perception of air quality is positively related to objective reading of air quality, and environmental transparency moderates this relationship. Specifically, we expect that the objective–subjective air quality relationship will be attenuated when the level of environmental transparency is higher. Recent studies have consistently found that transparency moderates the relationship between people's perceptions of government performance and other social phenomena such as corruption (Park and Blenkinsopp 2011) and social equity (Wu, Ma, and Yu 2017). Given these considerations, we develop our second hypothesis as follows.

*Hypothesis 2: Environment transparency negatively moderates the relationship between subjective and objective air quality, and the relationship is weaker when environmental transparency is higher.*

## Methods

### *Sample and Data*

We use recent large-scale citizen survey data and external assessments in over 30 Chinese largest cities to empirically examine the interaction effects of objective air quality and environmental transparency on citizens' subjective air quality. We test the two hypotheses by using multisource data from 32 largest cities in China. The sample covers four municipalities (Beijing, Tianjin, Shanghai, and Chongqing), 22 provincial capital cities (e.g., Guangzhou and Hangzhou), five subprovincial cities (Dalian, Qingdao, Ningbo, Xiamen, and Shenzhen), and one prefecture-level city (Suzhou). The administrative structure in China consists of five layers, and cities are at the first three layers (provinces, prefectures, and counties). There are 15 prefecture-level cities granted with subprovincial authorities by the central government, and 10 of them are provincial capital cities (e.g., Harbin and Xi'an). The sample of cities is comparable in administrative ranks while heterogeneous in geography and socioeconomic development. The 32 largest cities are frequent research subject for urban management scholar in studying China (Smyth, Mishra, and Qian 2008).

Data on citizens' perceived air quality are from the 2011 Lien Chinese Cities Service-Oriented Government Survey, which telephone interviewed over 25,000 residents in 32 largest cities by Computer Assisted Telephone Interviewing (CATI) method according to a stratified sampling framework. In each city, 700 residents were interviewed, except for megacities like Beijing and Shanghai, in which 1,000 residents were surveyed. Data on the government information disclosure of air quality (environmental transparency) are from IPE's independent assessment (Lorentzen, Landry, and Yasuda 2014). As IPE did not cover Haikou, our final number of observation is 31 cities. In addition, we collect archival data from the statistics of MEP to measure air quality and other city-level variables.

### *Dependent Variable*

One 10-point Likert scale item is used to gauge the respondents' subjective perceptions of air quality. The respondents were asked to rate their perceptions of air quality in the cities where they live. In the survey, 1 refers to the worst while 10 means the best. Although single-item measure is one of the limitations of this study, it is appropriate to capture respondents' overall assessment of air quality. Roughly 50.5 percent of the respondents scored higher than 6, which is in congruence with the 2010 MEP citizen survey.

### *Independent Variables*

Prior studies on the effect of air quality on life satisfaction usually use annual mean concentration levels or amount of emission per capita of air pollutants (e.g., sulfur dioxide, $SO_2$) as the key measures of air pollution (Luechinger 2009; Smyth, Mishra, and Qian 2008). We measure urban air quality by yearly averaged pollutant concentration indicators reported by the MEP. The key air pollution indicators include $SO_2$, nitrogen dioxide ($NO_2$), and particulate matters ($PM_{10}$), all measured by milligram per cubic meter ($mg/m^3$). As of 2011, the data on $PM_{2.5}$ were unavailable in most cities and we cannot assess whether it correlates with public satisfaction with air quality.

To take all air pollutants into account, we also use an umbrella measure to gauge air quality. We use the proportion of monitored days of air quality equal to or above national secondary standard (grade II) in the whole year (percent). The MEP classifies cities into three grades according to the national air quality standards (air quality index, AQI), with higher grades (e.g., grade I) denoting better air quality. The variable is coded as a dummy, since all sampled cities are either in the status of grade II (0) or grade III (1). Air quality fluctuates seasonally and it is appropriate to concentrate on the period when we collect our survey data. We use the average air pollution measures in the first half of the year,[1] since the survey was conducted from April to August 2011.

IPE developed the Urban Air Quality Information Transparency Index (AQTI) to rate cities' performance in air quality information disclosure (IPE 2012). The initial version of AQTI released in 2010 assessed 20 Chinese cities (e.g., Beijing, Shanghai, Guangzhou) and compared their performance

1 MEP, Air quality of key environmental protection cities in the first half year of 2011, July 22, 2011, http://www.mep.gov.cn/gkml/hbb/bgg/201107/t20110730_215576.htm (accessed January 18,

with those of 10 international cities (e.g., New York, Paris). The 2012 version of AQTI ranked 113 Chinese cities monitored by the MEP as National Key Environmental Protection Cities. The MEP monitors 10 air pollutants (e.g., $PM_{10}$, $PM_{2.5}$, $SO_2$, $NO_2$, and so forth), and their availability, completeness, promptness, and user-friendliness on urban EPBs' websites were assessed. The air pollutants were weighted by potential health impacts and environmental management practices. The total score of AQTI ranges from 0 to 100 points, and higher scores refer to higher levels of environmental transparency.

IPE also developed the Pollution Information Transparency Index (PITI) to assess environmental transparency of urban EPBs (IPE and NRDC 2009). PITI evaluated the performance of EPBs' online disclosure of pollution-intensive enterprises, clean production, environmental impact assessment, and other pollution-related information. In this article, we use AQTI and PITI to measure municipal governments' environmental transparency.

### Control Variables

Individual-level variables such as gender, age, education, and income that may affect people's subjective air quality are included in the model. Gender as a dummy is coded as 1 for male and 0 female. Age is measured by an ordinal variable ranging from 2 (18–29) to 6 (above 60). Education is denoted by an ordinal variable with four categories, ranging from 1 (primary school or below) to 6 (master's degree or above). Monthly income is gauged similarly by an ordinal variable ranging from 0 (no fixed income) to 14 (above 30,000 RMB Yuan).

The demographics of the respondents are similar with the latest census data and our sample is largely representative of the population of the sampled cities. In the sample, 45.19 percent of respondents were female, and 55.89 percent had college and above degrees. The majority of the respondents (45.89 percent) aged between 18 and 29, and those older than 60 accounted for 7.59 percent. In the sample, 62.84 percent of the respondents earned monthly income below 3,000 RMB Yuan, whereas rich residents with income above 6,000 RMB Yuan only accounted for 8.30 percent.

### Analytical Methods

As our data structure is nested or multilevel (individual citizens nested in cities), we adopt MLM to test our hypotheses. MLM is preferable to estimate variances at multiple levels. In the model, individuals are at Level 1 while cities are at Level 2. MLM can simultaneously estimate the variances at both Levels 1 and 2 (Raudenbush and Bryk 2002). We center Level 1 predictors within the cluster (group mean centering) and center Level 2 predictors by grand mean centering, which is appropriate to estimate the same-level and cross-level moderating effects in MLM (Enders and Tofighi 2007).

In order to test our two hypotheses, we need to estimate the moderating

2013)

effects of Level 2 variables (environment transparency) on the relationship between Level 2 independent variable (objective air pollution) and Level 1 dependent variable (subjective air quality). The first case is referred to the moderating effects at the same level (*means as outcomes* model) while the other case is cross-level (*slopes as outcomes* model). We follow standard procedures to detect moderating effects in MLMs and probe graphically, which enables us to intuitively interpret the results (Preacher, Curran, and Bauer 2006). Varying-intercept model is used to estimate the direct effects while varying-intercept, varying-coefficient model is used to estimate the moderating effects.

## Results

### *Descriptive Statistics*

Table 1 reports the descriptive statistics of our key variables. It reveals that the sampled cities vary substantially in objective air pollution indicators. Air grade in 14 sampled cities (or 43.75 percent) was rated as III, with the other 18 cities graded as II. Environmental transparency varies drastically across sampled cities with

**Table 1.** Descriptive Statistics and Correlation Matrices

| Variable | Observations | Mean | Std. Dev. | Min | Max | Correlation |
|---|---|---|---|---|---|---|
| Air quality satisfaction | 25,139 | 6.328 | 2.075 | 1 | 10 | 1 |
| $SO_2$ | 32 | 0.039 | 0.014 | 0.007 | 0.062 | −0.0989* |
| $NO_2$ | 32 | 0.044 | 0.012 | 0.015 | 0.064 | −0.0815* |
| $PM_{10}$ | 32 | 0.094 | 0.022 | 0.044 | 0.145 | −0.182* |
| Air grade (III=1) | 32 | 0.438 | 0.504 | 0 | 1 | −0.153* |
| AQTI | 31 | 32.355 | 19.607 | 9 | 76 | −0.096* |
| PITI | 31 | 50.555 | 17.216 | 23.2 | 83.7 | −0.0051 |
| Gender (male=1) | 25,222 | 0.548 | 0.498 | 0 | 1 | −0.0119 |
| Age | 24,939 | 3.069 | 1.260 | 2 | 6 | 0.117* |
| Education | 24,988 | 3.712 | 1.250 | 1 | 6 | −0.0438* |
| Income | 23,595 | 3.348 | 2.780 | 0 | 14 | −0.0464* |

Note: The last column denotes the correlation coefficients between air quality satisfaction (the dependent variable) and all independent variables. *p<0.05.

AQTI scores ranging from 9 to 76 and PITI scores between 23.2 and 83.7.

The last column in Table 1 shows the correlations between our independent variables and the dependent variable. The results suggest all air pollutants are negatively associated with air quality satisfaction and statistically significant at the 0.05 level. The measures of environmental transparency are both negatively correlated with subjective air quality measures, albeit only the correlation coefficient of AQTI is statistically significant. In addition, we find that age, education, and income of respondents are significant antecedents of subjective correlation, while gender is not significant.

We graphically plot the relationship between subjective and objective air quality measures in Figure 1. The distribution of sampled cities in the plot implies that objective air pollution is negatively related to subjective air quality. For instance, Haikou emitted least $SO_2$ among 32 sampled cities (0.007 mg/m$^3$), and its subjective perception was also highest (8.06 out of 10). In contrast, the midyear emission of $SO_2$ in Taiyuan was among the highest group (0.062 mg/m$^3$) and its subjective air quality was relatively low (5.02). There are also some outliers, however, owing to idiosyncratic geographical and socioeconomic characteristics. Coastal Qingdao had a slightly higher concentration of $SO_2$ (0.059) than inland Lanzhou (0.052), but Qingdao was top in subjective air quality (7.644) while Lanzhou is at the bottom (4.103), mostly because of Qingdao' coastal advantage to quickly disperse air pollutants. Another potential explanation is that subjective air quality measures are influenced by multiple sources of air pollutants. Although Qingdao and Lanzhou were similar in the concentration of $SO_2$ and $NO_2$, Qingdao' $PM_{10}$ emission (0.092 mg/m$^3$) was much lower than Lanzhou

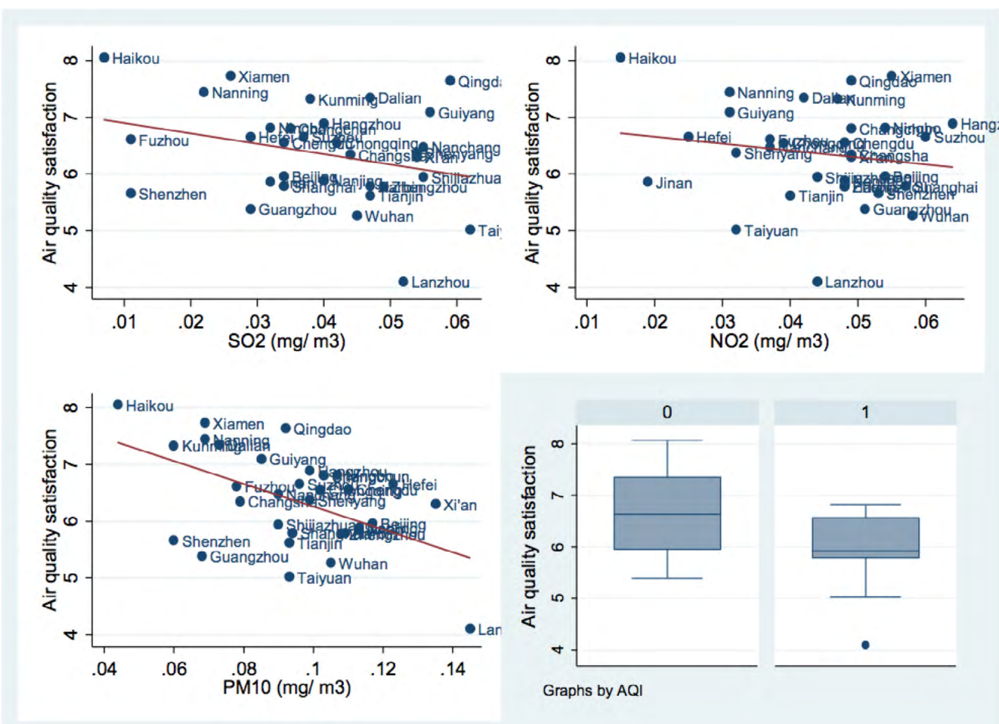



**Figure 1.** The relationship between objective and subjective air quality measures

(0.145 mg/m³). $PM_{10}$ is one of the most visible air pollutants that often lead to citizen complaints.

## Multilevel Model Estimations

We report MLM regression results for both direct and interaction effects of our independent variables (see Table 2). In the first column, the null model with no predictors suggests that 16.39 percent (interclass correlation (ICC) = 0.712/(0.712 + 3.633)) of total variances in air quality satisfaction could be attributable to Level 2 predictors. The explanatory power of Level 2 predictors is remarkably strong, particularly if we consider the small sample size (*N*=32) at Level 2 compared with large sample size at Level 1 (*N*=25,139) (Raudenbush and Bryk 2002). Actually, the impressive ICC, 0.1639, implies that residents nested in each sampled city have very similar perception on the air quality in the city. This finding implies that subjective air quality may significantly correlate with objective air quality. But, whether subjective and objective air quality are correlated need further investigated while controlling other variables. Although the variance at Level 2 (0.712) is statistically insignificant, it is substantially different from the standard error (0.179), suggesting it is essential to use MLM to estimate our models.

In the rest columns of Table 2, we sequentially enter our key independent variables and moderating variables. The results show that Level 2 air pollutants do have statistically significant effects on subjective air quality. When $SO_2$ is used as air pollution measure in Model 2, its regression coefficient is negative and significant ($\beta=-18.37$, $p<0.10$) and statistically significant at the 0.10 level. In Model 5, $NO_2$ is used to gauge air pollution, and we find that its effect is negative albeit insignificant ($\beta=-12.14$, $p>0.10$). Both $PM_{10}$ in Model 8 and air grade in Model 11 have significantly negative effects on subjective air quality ($\beta=-19.98$, $p<0.01$; $\beta=-0.73$, $p<0.01$). In a nutshell, our results suggest that Hypothesis 1 is partially supported and objective air quality is one of the key antecedents of subjective air quality.

We report the results on the moderating effects test in Table 2. Most of our interaction terms are statistically significant and consistent with our hypotheses, and therefore, Hypothesis 2 is partially supported. In Model 3, AQTI has insignificant effect on subjective air quality and its interaction term with objective air pollution (herein $SO_2$) also has little effect of substance ($\beta=0.10$, $p>0.10$). The insignificant effect of AQTI can be attributable to its substantial change over the short term after MEP mandated EPBs to release air quality information, which shrank cities' disparities in transparency (IPE 2012). When we turn to Model 4 with PITI as our moderator, we find its interaction term with air pollution have significantly positive effect on subjective air quality ($\beta=1.51$, $p<0.05$).

The moderating effect of environmental transparency on the relationship between $NO_2$ and subjective air quality was not supported by using

**Table 2.** Multilevel Model Estimates

| Model | 1 | 2 | 3 | 4 | 5 | 6 | 7 | 8 | 9 | 10 | 11 | 12 | 13 |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Pollution variable | | $SO_2$ | | | $NO_2$ | | | $PM_{10}$ | | | Grade | | |
| Moderating variable | | | AQTI | PITI | | AQTI | PITI | | AQTI | PITI | | AQTI | PITI |
| ***Fixed effects*** | | | | | | | | | | | | | |
| Pollution | | −18.37* | −21.40* | −19.73* | −12.14 | 2.06 | −3.73 | −19.98*** | −19.87*** | −15.22** | −0.73*** | −0.64*** | −0.64** |
| | | (10.28) | (12.12) | (11.35) | (12.80) | (16.59) | (14.06) | (5.85) | (5.44) | (6.03) | (0.27) | (0.24) | (0.26) |
| Transparency | | | −0.01 | 0.00 | | −0.00 | 0.01 | | −0.01 | −0.00 | | −0.02** | −0.01 |
| | | | (0.01) | (0.01) | | (0.01) | (0.01) | | (0.01) | (0.01) | | (0.01) | (0.01) |
| Interaction | | | 0.10 | 1.51** | | −0.97 | −1.31 | | 0.80*** | 0.83** | | 0.03** | 0.02 |
| term | | | (0.57) | (0.60) | | (1.13) | (0.96) | | (0.25) | (0.33) | | (0.01) | (0.02) |
| Gender | | −0.03 | −0.03 | −0.03 | −0.03 | −0.03 | −0.03 | −0.03 | −0.03 | −0.03 | −0.03 | −0.03 | −0.03 |
| (male=1) | | (0.03) | (0.03) | (0.03) | (0.03) | (0.03) | (0.03) | (0.03) | (0.03) | (0.03) | (0.03) | (0.03) | (0.03) |
| Age | | 0.20*** | 0.20*** | 0.20*** | 0.20*** | 0.20*** | 0.20*** | 0.20*** | 0.20*** | 0.20*** | 0.20*** | 0.20*** | 0.20*** |
| | | (0.01) | (0.01) | (0.01) | (0.01) | (0.01) | (0.01) | (0.01) | (0.01) | (0.01) | (0.01) | (0.01) | (0.01) |
| Education | | −0.02** | −0.03*** | −0.03*** | −0.02** | −0.03*** | −0.03*** | −0.02** | −0.03*** | −0.03*** | −0.02** | −0.03*** | −0.03*** |
| | | (0.01) | (0.01) | (0.01) | (0.01) | (0.01) | (0.01) | (0.01) | (0.01) | (0.01) | (0.01) | (0.01) | (0.01) |
| Income | | −0.03*** | −0.03*** | −0.03*** | −0.03*** | −0.03*** | −0.03*** | −0.03*** | −0.03*** | −0.03*** | −0.03*** | −0.03*** | −0.03*** |
| | | (0.00) | (0.00) | (0.00) | (0.00) | (0.00) | (0.00) | (0.00) | (0.00) | (0.00) | (0.00) | (0.00) | (0.00) |

| | | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Constant | 6.37*** | 6.37*** | 6.35*** | 6.47*** | 6.37*** | 6.41*** | 6.39*** | 6.37*** | 6.38*** | 6.40*** | 6.69*** | 6.59*** | 6.64*** |
| | (0.15) | (0.14) | (0.15) | (0.14) | (0.15) | (0.18) | (0.15) | (0.13) | (0.11) | (0.12) | (0.18) | (0.16) | (0.17) |
| ***Random effects*** | | | | | | | | | | | | | |
| Variance | 0.71 | 0.64* | 0.55** | 0.51*** | 0.69 | 0.59** | 0.60** | 0.52*** | 0.36*** | 0.43*** | 0.58** | 0.45*** | 0.50*** |
| (L2) | (0.18) | (0.16) | (0.14) | (0.13) | (0.17) | (0.15) | (0.15) | (0.13) | (0.09) | (0.11) | (0.15) | (0.12) | (0.13) |
| Variance | 3.63*** | 3.55*** | 3.56*** | 3.56*** | 3.55*** | 3.56*** | 3.56*** | 3.55*** | 3.56*** | 3.56*** | 3.55*** | 3.56*** | 3.56*** |
| (L1) | (0.03) | (0.03) | (0.03) | (0.03) | (0.03) | (0.03) | (0.03) | (0.03) | (0.03) | (0.03) | (0.03) | (0.03) | (0.03) |
| AIC | 103,938 | 95,487 | 92,775 | 92,773 | 95,489 | 92,777 | 92,778 | 95,480 | 92,762 | 92,767 | 95,484 | 92,769 | 92,772 |
| BIC | 103,962 | 95,552 | 92,855 | 92,853 | 95,554 | 92,858 | 92,858 | 95,545 | 92,843 | 92,847 | 95,548 | 92,849 | 92,853 |
| N | 25,139 | 23,221 | 22,543 | 22,543 | 23,221 | 22,543 | 22,543 | 23,221 | 22,543 | 22,543 | 23,221 | 22,543 | 22,543 |

Note: Standard errors in parentheses. ***p<0.01, **p<0.05, *p<0.1.

either AQTI or PITI as the moderator (see Models 6 and7). In terms of $PM_{10}$, the moderating effects of both AQTI ($\beta=0.80$, $p<0.01$) and PITI ($\beta=0.83$, $p<0.05$) are supported (see Models 9 and 10). In the case of AQI, only the moderating effect of AQTI is significant and positive ($\beta=0.03$, $p<0.05$) (see Models 12 and 13). In sum, the substantially similar results among the four air pollutants corroborate Hypothesis 2.

To visually illustrate the moderating effect of environmental transparency on the relationship between objective and subjective air quality measures, we draw the marginal changes of simple slopes of air pollution as the function of the two moderators in Figure 2. In Panel A of Figure 2, we reveal that air pollution (measured by $SO_2$) is negatively associated with subjective air quality when PITI is at the −1 standard deviation (SD) condition, but its sign turns to be slightly positive at the +1 SD condition. The result suggests the negative effect of air pollution (herein $SO_2$) on subjective air quality is attenuated with the increment of environmental transparency (PITI). When the value of centered PITI (ranging from −27.355 to 33.145) is larger than the lower confidence bound of region at the 0.05 level (−2.2323), the slopes of air pollution turn to be statistically insignificant. Pan-

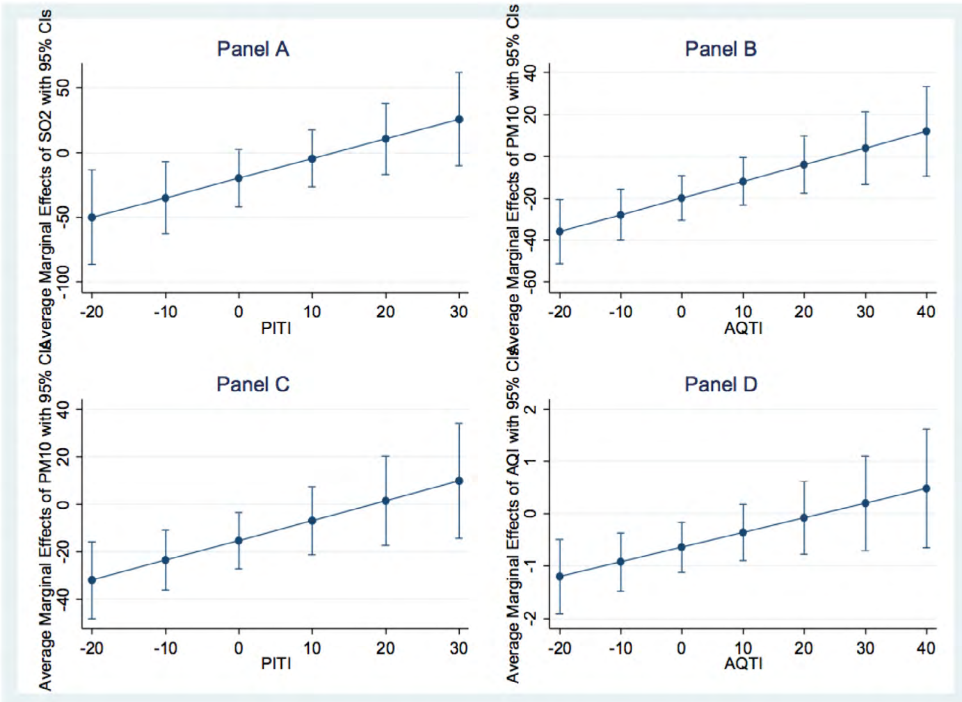



**Figure 2.** The moderating effects of transparency on the objective–subjective air quality relationship

el B–D in Figure 2 reveals similar patterns by using different measures of air pollution and transparency.

With regard to individual-level control variables, we find that gender has insignificant effect, albeit females have slightly better air quality perception than males. Age has significantly positive effect on subjective air quality, suggesting that elder residents are more tolerant of air pollution. Education and income are both significantly and positively associated with subjective air quality, which implies that high-educated and high-earned residents have relatively higher expectations toward air quality and they are consequently more discontent.

## Discussions

How to strike a balance between economic development and environmental sustainability has been one of the top policy priorities in contemporary urban governance, particularly for developing countries such as China (Economy 2010). In China, economic slowdown and pandemic environmental pollution have led to social protest and unrest (Albert and Xu 2016). It is imperative for government to respond in a more transparent manner. In China, failures in managing environmental pollution and citizen's perceptions are threatening the ruling party's political trust and legitimacy.

In this study, we use the data from a recent national citizen survey and a third-party assessment in over 30 Chinese major cities to empirically examine the congruence of subjective air quality and objective air quality measured by government archive data and the moderating effects of environmental transparency on the relationship. Independent and interaction effects of air pollution and environmental transparency on public satisfaction with air quality are quantitatively analyzed by MLM technique. We find that given the same level of environmental pollution, citizens actually would perceive a better level of air quality with a higher level of environmental transparency. Our quantitative analysis reveals the independent and interaction effects of air pollution and environmental transparency on subjective air quality.

Contributing to the literature on transparency, which has been debating whether transparency is good for public administration, this study suggests that environmental transparency is of crucial importance in addressing environment challenges. Other than other benefits such as better decision making for government, private businesses, and individual citizens, one of the most important benefits for democracy is environmental information provided by environmental transparency regime can of help in constructing citizens' perceived reality, air quality in this case. So far, most transparency studies were conducted in the western democratic context (Cucciniello, Porumbescu, and Grimmelikhuijsen 2017). This study focuses on transparency in the context of authoritarian China and examine its effects in a specific policy area, environmental protection. Our findings imply that government and nonprofit organizations can leverage various information-based policy instruments such as information disclosure, openness, and policy campaign to inform, educate, and empower the public to curb air pollution collectively (Li 2012).

In the literature of public performance measurement, the debate on whether and to what extent subjective performance measures echoes objective ones centers on the conceptualization issues of subjective and objective measurements and methodological complications (Kelly 2003). In this study, we explore the relationship in a new policy area, environmental protection, which has not been examined by public administration scholars. We compare subjective and objective air pollution measures to avoid the notorious inputs versus outcome comparison problems (Parks 1984). We also adopt a more sophisticated and advanced research design and data analytic technique to address potential statistical complications such as common method biases. Our findings suggest that public administrators should realize that citizens do have sufficient capacity to detect government performance and their perception of government performance is not only normatively important but also technically essential (Swindell and Kelly 2000; Van Ryzin, Immerwahr, and Altman 2008). Furthermore, our study emphasizes the role government can play in influencing citizen's perceptions, highlighting the potential of environmental transparency for global environmental governance (Li 2012; Tan 2014).

Given the escalating air pollution in urban China, central and local governments have been endeavoring in spending more resources in environmental regulations. Despite the government proclaims that air quality has been steadily improving, citizens do not resonate and buy, generating an enlarging gap between government rhetoric and citizen perceptions. Given the lingering air pollution, citizens may be hopeless of and habituate themselves to air pollution (Menz 2011), because it has been an essential part of their lives (Johnson et al. 2017). How to mitigate citizens' discontent with air pollution? Our findings suggest that government should be more transparent in disclosing information and engaging the public, which helps to retain social legitimacy and support.

Information-based policy instrument is among lots of policy instruments available to policy-makers, and incentive-based and mandatory instruments are equivalently and even more powerful in improving environmental performance. Despite information-based instrument is not the most powerful policy instrument, it is a promising and cost-efficient one. Lots of recent developments in environmental governance use information disclosure and transparency to nudge residents and industrial enterprises to reduce environmental pollution. It is thus important to highlight the value of information-based policy instruments in improving air quality, both objectively and subjectively.

Environmental transparency may be undermined by giant industrial enterprises due to interest entrenchment (Lorentzen, Landry, and Yasuda 2014), and government should leverage the power of social accountability and citizen participation. The ubiquitous data manipulation must be addressed by introducing third-party engagement, since it is common to find interrupted points or discontinuities of air quality monitoring, especially when air pollution is heavy. It is also relevant to expand the comprehensiveness of coverage and the density of monitoring stations, particularly in rural areas, which help citizens more precisely perceive and respond to air pollution.

The limitations of the study are threefold, and we hope future studies can replicate and extend our findings. First, the measurements of our key variables could be improved in future research. The measurement of subjective air quality is based on a single item from a national citizen survey, and we will address the issue in our future study. Although air quality is highly sensible and easier to be detected and felt, a subjective AQI consisting of multiple questions would be better. Citizens' perceptions and satisfaction are conceptually different, and we cannot distinguish their fine-grained differences due to data limitations. Our measurement of objective air quality could also be improved by using more recent data, since the Chinese government revised the national Ambient Air Quality Standards and included the concentration level of $PM_{2.5}$ in 2013. It should be noted that information disclosure is different from information availability or access, and we use the former as a proxy of the latter. In the survey, we asked the respondents about their access to government information, but it is not specifically about environmental information. We encourage future studies to collect more fine-tuned data to measure environmental transparency.

Second, our data are cross-sectional and we cannot infer causal mechanisms through which objective air quality measures affect subjective perceptions. It would be promising to use experimental design (e.g., natural and quasi-experiments) to examine the interaction effects of air pollution and transparency on citizens' perceptions. Third, we only surveyed residents in large cities in China, and our findings reported here should not be overgeneralized to small and medium-sized cities. Given China's large geographical disparity in economic development, energy structure, and environmental pollution, we call for future research to replicate and extend our investigation in other contexts (e.g., small and medium-size cities) and employ more solid measures and data analysis technique to answer the research questions.

## Acknowledgement

An earlier version of this manuscript entitled "Government Accountability and Citizens' Satisfaction with Air Quality in Chinese Cities" was presented at the Annual Conference of The Hong Kong Educational Research Association 2012-13, February 22, 2013, Hong Kong Institute of Education, Hong Kong, China.

We want to thank Hua Wang, Xu Wu, and Yuan Xu for organizing this workshop. We also thank workshop participants and two anonymous reviewers for helpful comments on an earlier version of this paper. We are grateful to financial support from the National Natural Science Foundation of China (NSFC) (No.: 71774164; 1633004) and the Key Project of Ministry of Education (18JZD048).

## References

Albert, E., and B. Xu. 2016. "China's Environmental Crisis." Council on Foreign Relations. http://www.cfr.org/china/chinas-environmental-crisis/p12608.

Andrews, R., G. Boyne, and R. Walker. 2011. "The Impact of Management on Administrative and Survey Measures of Organizational Performance." *Public Management Review* 13 (2): 227-55.

Bawa, K.S., L.P. Koh, T.M. Lee, J. Liu, P.S. Ramakrishnan, and D.W. Yu. et al. 2010. "China, India, and the Environment." *Science* 327 (5972): 1457-59.

Brown, K., and P.B. Coulter. 1983. "Subjective and Objective Measures of Police Service Delivery." *Public Administration Review* 43 (1): 50-58. http://doi.org/10.2307/975299?ref=no-x-route:3ef10db3b8163ecfda4036a41f390e01.

Burger, J., and M. Gochfeld. 1998. "The Tragedy of the Commons 30 Years Later." *Environment: Science and Policy for Sustainable Development* 40 (10): 4-13.

Charbonneau, É., and G.G. Van Ryzin. 2011. "Performance Measures and Parental Satisfaction with New York City Schools." *The American Review of Public Administration* 42 (1): 54-65. http://doi.org/10.1177/0275074010397333.

Chen, R., P. Yin, X. Meng, C. Liu, L. Wang, and X. Xu et al. 2017. Fine Particulate Air Pollution and Daily Mortality: A Nationwide Analysis in 272 Chinese Cities. *American Journal of Respiratory and Critical Care Medicine,* 196 (1) :73.

CIA Worldfact Book. 2017. Accessed February 2, 2017. https://www.cia.gov/library/publications/the-world-factbook/geos/print_ch.html.

Cucciniello, M., G.A. Porumbescu, and S. Grimmelikhuijsen. 2017. "25 Years of Transparency Research: Evidence and Future Directions." *Public Administration Review* 77 (1): 32-44. http://doi.org/10.1111/puar.12685.

Daneshvary, N., R. Daneshvary, and R.K. Schwer. 1998. "Solid-Waste Recycling Behavior and Support for Curbside Textile Recycling." *Environment and Behavior* 30: 144-61.

Dasgupta, S., and D. Wheeler. 1997. "Citizen Complaints as Environmental Indicators: Evidence from China." *World Bank Policy Research Working Paper Series* (1704).

Dong, Y., M. Ishikawa, X. Liu, and S. Hamori. 2011. "The Determinants of Citizen Complaints on Environmental Pollution: An Empirical Study from China." *Journal of Cleaner Production* 19 (12): 1306-14.

Ebreo, A., J. Hershey, and J. Vining. 1999. "Reducing Solid Waste: Linking Recycling to Environmentally Responsible Consumerism." *Environment and Behavior* 31: 107-35.

Economy, E.C. 2010. *The River Runs Black: The Environmental Challenge to China's Future*. London: Cornell University Press.

Enders, C.K., and D. Tofighi. 2007. "Centering Predictor Variables in Cross-Sectional Multilevel Models: A New Look at an Old Issue." *Psychological Methods* 12 (2): 121-38.

Fung, A., M. Graham, and D. Weil. 2007. *Full Disclosure: The Perils and Promise of Transparency*. Cambridge: Cambridge University Press.

Gupta, A. 2010. "Transparency in Global Environmental Governance: A Coming of Age?" *Global Environmental Politics* 10 (3): 1-9.

Hadrich, J.C., and C.A. Wolf. 2011. "Citizen Complaints and Environmental Regulation of Michigan Livestock Operations." *Journal of Animal Science* 89 (1): 277-286.

Harris, P. G. 2006. Environmental Perspectives and Behavior in China. *Environment and Behavior* 38 (1): 5-21.

He, G., M. Fan, and M. Zhou 2016. "The Effect of Air Pollution on Mortality in

China: Evidence from the 2008 Beijing Olympic Games." *Journal of Environmental Economics and Management* 79: 18-39.

Holdaway, J. 2013. "Environment and Health Research in China: The State of the Field." *The China Quarterly* 214: 255-82.

Holzer, Marc, and Kaifeng Yang. 2004. "Performance Measurement and Improvement: An Assessment of the State of the Art." *International Review of Administrative Sciences* 70 (1): 15-31.

Howell, S.E., and S.B. Laska. 1992. "The Changing Face of the Environmental Coalition: A Research Note." *Environment and Behavior* 24: 134-44.

IPE, & NRDC. 2009. *Breaking the Ice on Environmental Open Information: The 2008 Pollution Information Transparency Index (PITI) First Annual Assessment of Environmental Transparency in 113 Chinese Cities*. Beijing: Institute of Public & Environmental Affairs (IPE) & Natural Resources Defense Council (NRDC).

IPE. 2012. *Small Particles, Big Breakthrough: 2012 Urban Air Quality Information Transparency Index*. Beijing: Institute of Public & Environmental Affairs (IPE).

Johnson, T., A.P.J. Mol, L. Zhang, and S. Yang. 2017. "Living Under the Dome: Individual Strategies Against Air Pollution in Beijing." *Habitat International* 59: 110-17.

Kelly, J.M. 2003. "Citizen Satisfaction and Administrative Performance Measures: Is There Really a Link?" *Urban Affairs Review* 38 (6): 855-66. http://doi.org/10.1177/1078087403038006005.

Li, W., D. Li. 2012. "Environmental Information Transparency and Implications for Green Growth in China." *Public Administration and Development* 32 (3): 324-34.

Li, Z., H. Folmer, and J. Xue. 2016. "Perception of Air Pollution in the Jinchuan Mining Area, China: A Structural Equation Modeling Approach." *International Journal of Environmental Research and Public Health* 13 (7): 735-18. http://doi.org/10.3390/ijerph13070735.

Lieberthal, Kenneth. 1997. "China's Governing System and Its Impact on Environ mental Policy Implementation." *China Environment Series*. Washington, DC: The Woodrow Wilson Center.

Liu, J., and J. Diamond. 2005. "China's Environment in a Globalizing World." *Nature* 435 (7046): 1179-86.

Liu, X., Y. Dong, and C. Wang. 2011. "Citizen Complaints about Environmental Pollution: A Survey Study in Suzhou, China." *Journal of Current Chinese Affairs*, 40 (3):193-219.

Lorentzen, P., P. Landry, and J. Yasuda. 2014. "Undermining Authoritarian Innovation: The Power of China's Industrial Giants." *The Journal of Politics* 76 (1): 182-94.

Luechinger, S. 2009. "Valuing Air Quality Using the Life Satisfaction Approach." *The Economic Journal* 119 (536): 482-515.

Ma, L., and Z. Zhang. 2015. "Microblogging (weibo) and Environmental Nonprofit Organizations in China: The Case of Urban Air Pollution Monitoring Campaign." In *Cases on Strategic Social Media Utilization in the Nonprofit Sector*, edited by H. Asencio and R. Sun, 94-121. Hershey, PA: IGI Global.

Menz, T. 2011. "Do People Habituate to Air Pollution? Evidence from International Life Satisfaction Data." *Ecological Economics* 71: 211-19.

Miller, T.I., and M.A. Miller. 1991. "Standards of Excellence: U.S. Residents' Evaluations of Local Government Services." *Public Administration Review* 51 (6): 503-14.

Park, H., and J. Blenkinsopp. 2011. "The Roles of Transparency and Trust in the Relationship Between Corruption and Citizen Satisfaction." *International Review of Administrative Sciences* 77 (2): 254-74.

Parks, R.B. 1984. "Linking Objective and Subjective Measures of Performance." *Public Administration Review* 44 (2): 118-27.

Preacher, K.J., P.J. Curran, and D.J. Bauer. 2006. "Computational Tools for Probing Interactions in Multiple Linear Regression, Multilevel Modeling, and Latent Curve Analysis." *Journal of Educational and Behavioral Statistics* 31 (4): 437-48.

Qin, Y., and H. Zhu. 2018. *Run Away? Air Pollution and Emigration Interests in China. Journal of Population Economics* 31 (1):235-266.

Raudenbush, S.W., and A.S. Bryk. 2002. *Hierarchical Linear Models: Applications and Data Analysis Methods* 2nd edition. Thousand Oaks, CA: Sage Publications, Inc.

Rittel, Horst, and Webber Melvin. 1973. "Dilemmas in a General Theory of Planning." *Policy Sciences* 4 (2): 155-69.

Schachter, H.L. 2010. "Objective and Subjective Performance Measures." *Administration & Society* 42 (5): 550-67.

Shi, X. 2015. "Factors Influencing the Environmental Satisfaction of Local Residents in the Coal Mining Area, China." *Social Indicators Research* 120 (1): 67-77. http://doi.org/10.1007/s11205-014-0584-z.

Shingler, J., M.E. Van Loon, and T.R. Alter. 2008. "The Importance of Subjective Data for Public Agency Performance Evaluation." *Public Administration Review* 68 (6):1101-1111.

Smyth, R., V. Mishra, and X. Qian. 2008. "The Environment and Well-Being in Urban China." *Ecological Economics* 68 (1–2): 547-55.

Stipak, B. 1979. "Citizen Satisfaction with Urban Services: Potential Misuse as a Performance Indicator." *Public Administration Review* 39 (1): 46-52.

Stipak, B. 1980. "Local Governments' Use of Citizen Surveys." *Public Administration Review* 40 (5): 521-25.

Swindell, D., and J.M. Kelly. 2000. "Linking Citizen Satisfaction Data to Performance Measures: A Preliminary Evaluation." *Public Performance & Management Review* 24 (1): 30-52.

Tan, Y. 2014. "Transparency Without Democracy: The Unexpected Effects of China's Environmental Disclosure Policy." *Governance* 27 (1) :37-62 .

Van Ryzin, G.G., S. Immerwahr, and S. Altman. 2008. "Measuring Street Cleanliness: A Comparison of New York City's Scorecard and Results from a Citizen Survey." *Public Administration Review* 68 (2): 295-303.

Wang, B.Z., and Z. Cheng. 2017. "Environmental Perceptions, Happiness and Pro-Environmental Actions in China." *Social Indicators Research* 132 (1): 357-75.

World Bank. 2007. "Cost of Pollution in China." Accessed February 2, 2017. http://siteresources.worldbank.org/INTEAPREGTOPENVIRONMENT/Resources/China_Cost_of_Pollution.pdf.

Wu, W., L. Ma, and W. Yu. 2017. "Government Transparency and Perceived Social Equity." *Administration & Society* 49 (6): 882-906.

Xiao, C.Y., and D.Y. Hong. 2010. "Gender Differences in Environmental Behaviors in China." *Population Environment* 32: 88-104.

Xinhua. 2016. "China's Car Ownership Reaches 172 Million." Accessed February 10, 2017. http://www.chinadaily.com.cn/business/motoring/2016-01/26/content_23253925.htm.

Yu, W. 2011. "Open Government Information: Challenges Faced by Public Human Resource Management in China." *International Journal of Public Administration* 34 (13): 879-88.

Zhang, Q., and R. Crooks. 2012. *Toward an Environmentally Sustainable Future: Country Environmental Analysis of the People's Republic of China*. Manila: Asian Development Bank.

# Environmental Performance Rating and Disclosure: An Empirical Investigation in Jiangsu, China

Yanhong Jin[1], Hua Wang[2*], David Wheeler[3]

## Abstract

Environmental performance rating and disclosure (PRD) has emerged as an alternative or complementary approach to conventional pollution regulation, especially in developing countries. However, little systematic research has been conducted on the effectiveness of this emerging environmental policy instrument. This paper investigates the impact of a PRD program, Green Watch in Jiangsu, China, which has been operating for more than 10 years. This study assesses the impact of Green Watch by using panel data on pollution emissions from rated and unrated firms, before and after implementation of the program. Controlling for the characteristics of firms and locations, time trend, and the initial level of environmental performance, we draw two main findings: (1) firms covered by Green Watch improved their environmental performance more than non-covered firms and (2) the program effects varied by the initial level of environmental performance: bad performers improved more than good performers and moderately non-compliant firms improve more than firms that are significantly out of compliance. The reasons for these different responses seem to be that the strengths of incentives that the disclosure program provides to the polluters at different levels of compliance are different and the abatement costs of achieving desired levels of ratings are different for different firms. Further investigation is merited for a large scale or even a national level in most recent years.

[1] *Rutgers, The State University of New York, New Brunswick, NJ*

[2] *Renmin University of China, Beijing*

[3] *Independent Scholar*

[*] *Corresponding author:* huawangbeijing@qq.com



doi: 10.18278/cpj.1.1.5

***Keywords:*** Environmental performance rating; Public disclosure; Environmental policy instrument; Developing country; China; Green Watch; Policy evaluation

# Calificación de desempeño ambiental y divulgación: una investigación empírica en Jiangsu, China


## Resumen

La calificación y divulgación del desempeño ambiental (PRD) ha surgido como un enfoque alternativo o complementario a la regulación de la contaminación convencional, especialmente en los países en desarrollo. Sin embargo, se ha realizado poca investigación sistemática sobre la efectividad de este instrumento emergente de política ambiental. Este documento investiga el impacto de un programa de PRD, Green Watch en Jiangsu, China, que ha estado operando durante más de 10 años. Este estudio evalúa el impacto de Green Watch mediante el uso de datos de panel sobre emisiones de contaminación de empresas calificadas y no calificadas, antes y después de la implementación del programa. Al controlar las características de las empresas y las ubicaciones, la tendencia temporal y el nivel inicial de desempeño ambiental, extraemos dos conclusiones principales: (1) las firmas cubiertas por Green Watch mejoraron su desempeño ambiental más que las firmas no cubiertas y (2) los efectos del programa variaron según el nivel inicial de desempeño ambiental: los de bajo desempeño mejoraron más que los de buen desempeño y las firmas que no cumplen con las normas mejoran más que las firmas que están significativamente fuera de cumplimiento. Las razones de estas respuestas diferentes parecen ser que las fortalezas de los incentivos que el programa de divulgación proporciona a los contaminadores en diferentes niveles de cumplimiento son diferentes y los costos de reducción de alcanzar los niveles deseados de calificaciones son diferentes para diferentes empresas. Se merece una investigación adicional a gran escala o incluso a nivel nacional en los últimos años.


**Palabras clave:** Calificación de desempeño ambiental; Revelación pública; Instrumento de política ambiental; País en desarrollo; China; Reloj verde; Evaluación de políticas

# 企业环境绩效评级公开：针对中国江苏省的实证研究


摘要

企业环境绩效评级公开已成为常规污染控制政策手段的替代或补充手段， 在发展中国家尤为如此。然而，学术界对这一新出现的环境政策手段的有效性的研究则很少。本研究评估了中国江苏的企业环境绩效评级公开工作，这一运作十多年的环境政策手段， 在污染控制方面所带来的影响。本研究采用了来自被评级和未被评级的公司在该政策实施之前和之后关于污染排放的面板数据。给定公司和所在地的特点、时间趋势和环境绩效的初始水平，作者分析得出以下两个主要结论：（1） 被评级公开的公司的环境绩效比没有被评级公开的公司的环境绩效好；（2）评级公开的影响效果因环境绩效的初始水平而异：初始绩效差的公司比绩效好的公司提高得多，而中等合规的公司其改善程度比严重违规的公司高。这些不同反应的原因可能在于，评级公开工作对处于不同合规水平的污染者所提供的激励力度不同；对于不同的公司，达到期望的评级水平的成本不同。


关键词：环境绩效评级；信息公开；环境政策工具；发展中国家；中国绿色观察；政策评估

## 1. Introduction

Environmental performance rating and disclosure (PRD) has emerged as a substitute or complement for traditional pollution regulation, especially in developing countries (Bennett, James, and Klinkers 2017; Dasgupta, Wang, and Wheeler 2006; Kurniawan 2017; Meng et al. 2014; Portney 2000; Tietenberg 1998). Indonesia's PROPER (Program for Pollution Control, Evaluation and Rating), initiated in June 1995, was the first PRD program in developing countries. Because of its perceived overall success, as measured by reduced emissions at a lower regulatory cost, many countries have established similar programs for a variety of industry sectors and pollutants in diverse economic, institutional, and cultural settings. These programs include the Philippines' EcoWatch, India's Green Rating Project, China's Green Watch, Vietnam's Green Bamboo, Ghana's EPRD, and Ukraine's PRIDE. PRD programs are particularly attractive for developing countries because institutional weaknesses hinder conventional monitoring and enforcement of environmental laws, regulations, and standards (Foulon, Lanoie, and Laplante 2002), and because PRD programs have lower regulatory costs (Dasgupta, Wang, and Wheeler 2006).

The literature on the effectiveness of PRD programs is very limited and falls into two groups. The first group compares the environmental performance ratings of firms before and after a program is implemented and ascribes any ratings improvements to the program (Afsah, Laplante, and Wheeler 1997). However, this approach may be confounded by time-varying factors such as technology improvements. The second group compares polluting emissions from rated and unrated firms and credit performance improvements by rated firms to the program. However, this approach may be confounded by selection bias (e.g., firms with better environmental performance may be more likely to be rated).

It is rare to have pollution data for both rated and unrated firms before and after implementation of a PRD program. Garcia, Sterner, and Afsah (2007) and Garcia, Afsah, and Sterner (2009) assessed the effectiveness of Indonesia's PROPER using measured pollution from rated and unrated firms, both ex ante and ex post. Their 2007 study suggested that PROPER reduced emissions intensity, with a particularly rapid and strong impact on firms that had poor initial compliance records. Their 2009 study found a strong reactive response during the first six months of disclosure, followed by a more moderate, but still significant, longer-run response as management adjusts to the new regime.

This study extends PRD assessment to China, using panel data on pollution from rated and unrated firms, before and after implementation of the Green Watch program in Jiangsu province. It offers the following two main contributions to the literature. First, we exploit the panel structure of the data to control for confounding factors such as time-variant technology improvement and selection bias between rated and unrated firms. Second, we go beyond a single measure of environmental performance by considering the impact of ratings disclosure on several measures, including emissions intensity and effluent concentrations for a variety of air and water pollutants.

The remainder of the paper is organized as follows. Section 2 reviews the relevant literature, focusing on the role of PRD programs in developing countries. Section 3 describes China's Green Watch program, while Section 4 describes the survey instruments and provides descriptive statistics for major variables. Section 5 presents the estimation model and results, and Section 6 summarizes and concludes the paper.

## 2. Previous Research

The literature on pollution control policies consists of extensive work on command-and-control, and market-based and information-based instruments (Dasgupta, Wang, and Wheeler 2006; Keohane and Olmstead 2016). Command-and-control instruments are often inefficient and ineffective in developing countries, because firms may fail to report adequately, regulators may lack the technical and administrative capacity for effective monitoring and enforcement, and judicial systems may be weak and/or corrupt. These weaknesses limit regulators' ability to employ market-based instruments, which also work less effectively in countries where market failures are common and legal and institutional supports for formal market activities are weak.

Information-based instruments can be effective in developing countries where strong regulatory institutions and/or well-developed markets are absent, but where enough information can be reliably obtained to provide credible performance ratings. In practice, diverse information programs have served as complements to command-and-control and market-based instruments (Kleindorfer and Orts 1998). Information programs reduce the information asymmetry between polluters and environmental stakeholders (consumers, communities, NGOs, and investors), empowering these stakeholders to pressure polluters for improved environmental performance (Bui and Mayer 2003; Chaklader and Gulati 2015; Kennedy, Laplante, and Maxwell 1994; Meng et al. 2014; Oberholzer-Gee and Mitsunari 2006).When implemented appropriately, information instruments promote better interaction and dialogue among firms, stakeholders, and regulators (Cormier and Magnan 2015; Garcia, Sterner, and Afsah 2007).

Information instruments also leverage markets in significant ways. An extensive empirical literature suggests that disclosure of firms' bad environmental performance reduces their stock prices both in developed countries (Foulon, Lanoie, and Laplante 2002; Hamilton 1995; Kiel 1995; Konar and Cohen 1997; Lonoie and Roy 1998; Lyon and Shimshack 2015) and developing countries such as Argentina, Chile, Mexico, and the Philippines (Dasgupta, Laplante, and Mamingi 2001). Jackson (2001) and Boyle and Kiel (2001) review the impacts of disclosure on U.S. housing prices, which are found to be lower near Superfund sites (Kohlase 1991; Reichert 1997), hazardous waste sites (Thayer, Albers, and Rahmatian 1992), non-hazardous landfills (Michaels and Smith 1990), nuclear radiation sources (Gamble and Downing 1982), and polluting manufacturing plants (Garcia, Afsah, and Sterner 2009). Housing prices also negatively respond to publicized environmental contamination incidents (Kiel 1995; Kiel and McClain 1995).

Information instruments have diverse forms, including reports of measured pollution, environmental accident reports, and environmental performance ratings. In the United States, for example, the Toxics Release Inventory (TRI) discloses toxic chemical releases and waste management activities by significant toxic polluters and federal facilities. However, regulatory institutions in developing countries face significant challenges to implement such emission inventories due to the weak monitoring and enforcing power. In addition, despite an emerging literature on stakeholders' role in improving firms' environmental performance (Arora and Cason 1998; Blackman and Banniseter 1998; Pargal and Wheeler 1996; Wheeler et al. 1997), concerns about the public's ability to understand and utilize complex emissions reports remain. For example, Bui and Mayer (2003) find that the release of TRI's highly detailed information on facilities' toxic emissions has virtually no effect on housing prices in neighboring areas, even when the release of such information is unexpected. The dual problems of emission inventories in developing countries—technical feasibility and public understanding—have led to a preference for programs that condense complex information into environmental performance ratings that are disclosed to the public.

The literature finds a significant, positive impact of PRD programs on regulatory compliance (Afsah, Laplante, and Wheeler 1997; Dasgupta, Wang, and Wheeler 2006; Garcia, Sterner, and Afsah 2007; Garcia, Afsah, and Sterner 2009; Wang et al. 2004). Dasgupta, Wang, and Wheeler (2006) summarize the changes in compliance rates for several PRD programs in Asia. During the first and second years after inception, compliance rates among covered firms increased from 37% to 61% in Indonesia, 8% to 58% in the Philippines, 10% to 24% in Vietnam, 75% to 85% in Zhenjiang, China, and 23% to 62% in Hohhot, China. Several empirical studies also find that PRD programs have improved firms' environmental performance in Indonesia (Afsah, Laplante, and Wheeler 1997; Garcia, Sterner, and Afsah 2007; Garcia, Afsah, and Sterner 2009) and China (Wang et al. 2004). However, data constraints generally limit these studies to comparisons of environmental ratings before and after program implementation, or comparisons of compliance status between rated and unrated firms. Unfortunately, intertemporal rating comparisons are subject to confounding effects from time-varying factors such as technology change, while cross-sectional comparisons can be subject to significant selection bias.

## 3. China's Green Watch Program

Despite long-standing efforts to control pollution with traditional regulatory instruments, China continues to have severe pollution problems. This has led China's State Environmental Protection Administration (SEPA) to test the effectiveness of environmental PRD program supported by the World Bank. In 1999, SEPA launched its Green Watch program in Zhenjiang City, Jiangsu Province, and Hohhot City, Inner Mongolia Autonomous District. Zhenjiang implemented a relatively complex rating system, as shown in Figure 1, while Hohhot used a simpler rating system that was suited to its lower level of economic and institutional development (Wang et al. 2004). As shown in Figure 1, Green Watch in Jiangsu rated firms' environmental performance from best to worst in five colors—green for superior performance; blue for full compliance; yellow for meeting major compliance standards but violating some minor requirements; red for violating important standards; and black for more extreme non-compliance.

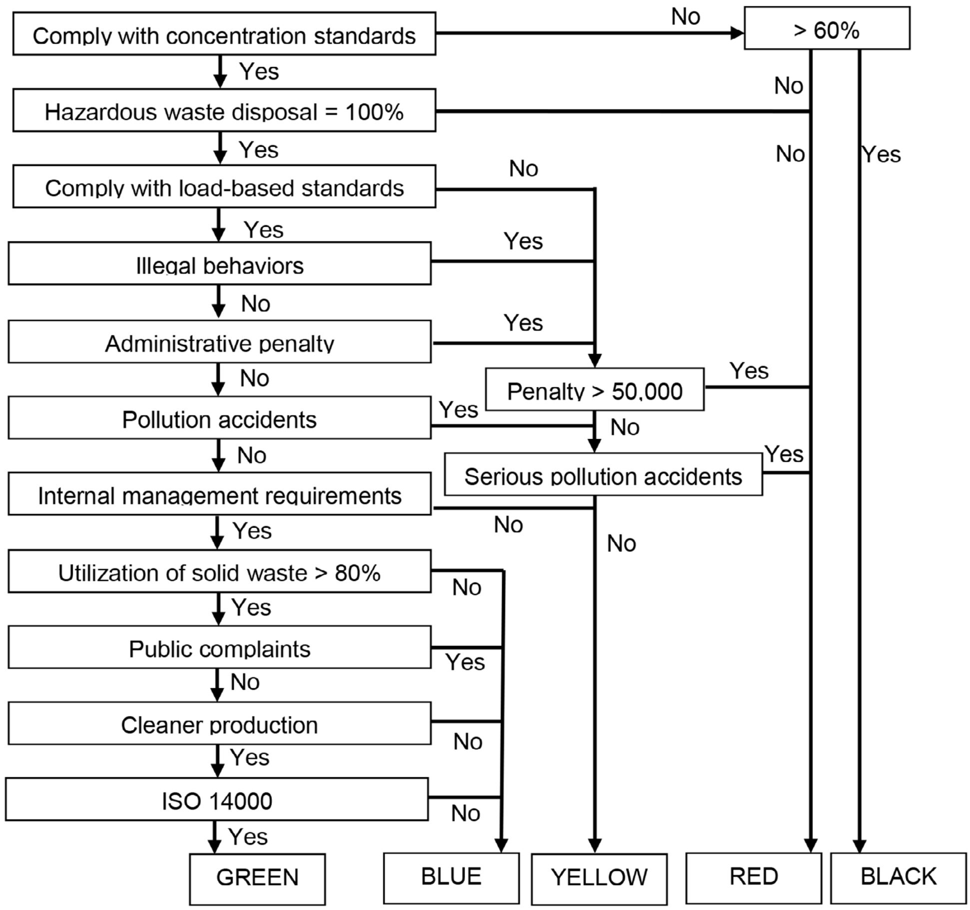



**Figure 1**. Rating Criteria of the Green Watch Program in Jiangsu

Source: Revised based on Figure 1 in Wang et al. (2004).

Green Watch ratings provided incentives for firms to improve their environmental performance in a comprehensive way. The primary benchmarks for ratings were China's emission and discharge standards that specify effluent concentration limits. Firms violating any of these standards were rated red, and firms violating standards in more than 60% of inspections were rated black. The secondary benchmarks were China's load-based emission and discharge standards. Firms that satisfied the primary benchmarks but violated the secondary standards were rated yellow. The ratings system also incorporated other performance indicators, including hazardous waste disposal practices, solid waste recycling, pollution accidents, public complaints, internal management requirements, China cleaner production certificates, ISO 14000 certificates, administrative penalties, and other citations for illegal activity. For each indicator, the system specified a link to ratings that was clear, unambiguous, and publicly available.

The first Green Watch ratings were disclosed through the media in 1999. The program was extended from Zhenjiang to all of Jiangsu Province in 2001, and to eight other provinces during 2003-2005. Nationwide implementation of the Green Watch rating has been promoted since 2005. Overall, the evidence suggests a positive impact for the program, at least in the beginning years. Table 1-1 shows that in Zhenjiang, the percentage of firms with positive ratings (green, blue, and yellow) increased from 75% in 1999 to 85% in 2000. The most significant changes were in the extremely noncompliant black group, whose percentage dropped from 11% in 1999 to 2% in 2000, and a major shift from the partially compliant yellow group (44%–22%) to the fully compliant blue group (27%–61%).

Evidence for the Green Watch program in Jiangsu Province indicated both increasing participation by firms and improvement in their compliance rates. As shown in Table 1-1, the number of rated firms increased more than 10-fold, from 1,059 in 2001 to 11,215 in 2006; and the percentage of firms with positive ratings (green, blue, and yellow) increased from 83% in 2001 to 90% in 2006. Furthermore, Table 1-1 suggests that Green Watch ratings pro-

**Table 1-1**. Firms' Environmental Performance Ratings by the First Green Watch Program in Jiangsu Province in the First Eight Years (% Representation in Parentheses)

| | Year | Green | Blue | Yellow | Red | Black | Total |
|---|---|---|---|---|---|---|---|
| Pilot program in Zhenjiang City | 1999 | 3 (3.30) | 25 (27.47) | 40 (43.96) | 13 (14.29) | 10 (10.99) | 91 |
| | 2000 | 2 (2.10) | 58 (61.05) | 21 (22.11) | 12 (12.63) | 2 (2.10) | 95 |
| Province-wide program | 2001 | 77 (7.27) | 512 (48.35) | 288 (27.20) | 141 (13.31) | 41 (3.87) | 1,059 |
| | 2002 | 182 (7.26) | 1,196 (47.69) | 655 (26.12) | 398 (15.87) | 77 (3.07) | 2,508 |
| | 2003 | 267 (8.69) | 1,545 (50.26) | 789 (25.67) | 367 (11.94) | 106 (3.44) | 3,074 |
| | 2004 | 329 (6.46) | 2,659 (52.20) | 1,467 (28.80) | 525 (10.31) | 114 (2.24) | 5,094 |
| | 2005 | 530 (6.62) | 4,016 (50.17) | 2,614 (32.65) | 702 (8.77) | 143 (1.79) | 8,005 |
| | 2006 | 702 (6.26) | 5,414 (48.27) | 3,944 (35.17) | 1,000 (8.92) | 155 (1.38) | 11,215 |

Sources: Pilot program in Zhenjiang: Wang et al. (2004). Province-wide program: the Legislative Affairs Office of the China State Council (2007) (http://www.chinalaw.gov.cn/article/dfxx/dffzxx/js/200706/20070600021431.shtml; last accessed on May 19, 2009).

vided a strong improvement incentive for noncompliant (red and black) firms, with stronger effects on firms with red ratings (moderate noncompliance) than those with black ratings (extreme noncompliance). However, more rigorous analyses are needed to further identify the effect of the Green Watch ratings to controlling for potential confounding factors.

In 2013, the central government modified the Green Watch program design, in response to the call for building a credit society by the Party. In 2013, the Ministry of Environmental Protection issued a policy document "Trial Procedure for Firm's Environmental Credit Rating," jointly with the National Development and Reform Commission, People's Bank of China, and the Commission of Bank Monitoring and Supervision. The new program, or the Second Green Watch program, keeps the nature of color-coding environmental performance, but focus more on compliance with government regulations. Indicators and scores are recommended by the central government, but provinces can modify them and have their own design. Up to now, most of the provinces have started their second generation of Green Watch programs.

Jiangsu Province started its new Green Watch program in the year of 2012, with the firms marked as state-controlled pollution sources at the beginning. Table 1-2 provides information about the new Green Watch ratings in Jiangsu in the first five years of the program from 2012 to 2016. The coverage was stable, with the green ratings increasing along the years. However, there has been no serious research conducted to evaluate the new program.

**Table 1-2**. Firms' Environmental Performance Ratings by the New Green Watch Program in Jiangsu Province in the First Five Years (% for State Controlled Firms)

| Years | 2012 | 2013 | 2014 | 2015 | 2016 |
|---|---|---|---|---|---|
| Green | 15.60% | 20.40% | 31.90% | 32.00% | 38.90% |
| Blue | 60.60% | 53.30% | 47.30% | 49.80% | 46.80% |
| Yellow | 17.20% | 17.70% | 13.40% | 11.00% | 8.80% |
| Red | 5.40% | 6.80% | 4.90% | 4.80% | 3.70% |
| Black | 1.20% | 1.90% | 2.50% | 2.40% | 1.70% |
| Total % | 100% | 100% | 100% | 100% | 100% |
| Total number of firms | 921 | 961 | 878 | 944 | 1,064 |

Source: Website of Jiangsu Environmental Protection Department.

## 4. Data

A dataset for both rated and unrated firms in four cities of Jiangsu province (Huaian, Wuxi, Yangzhou, and Zhenjiang) for the years before and after the start of the Green Watch programs (during the period of 1996-2001) was collected from the municipal governments and could be utilized to evaluate the effectiveness of the Green Watch program in the years of beginning. The dataset consisted of detailed information on the firms' characteristics, pollution, and environmental performance ratings.

**Table 2**. City Comparisons for Industrial Pollution and Socioeconomic and Environmental Conditions, 2001

| | Huaian | Wuxi | Yangzhou | Zhenjiang |
|---|---|---|---|---|
| **Socioeconomic conditions** | | | | |
| GDP per capita (Yuan) | 14,359 | 37,700 | 21,311 | 18,852 |
| Economic growth rate (%) | 11.05 | 12.20 | 7.30 | 11.10 |
| Unemployment rate (%) | 3.84 | 3.62 | 3.60 | 2.30 |
| Population (1,000) | 558 | 2,131 | 1,097 | 628 |
| **Environmental conditions** | | | | |
| TSS: total suspend solids (mg/m$^3$) | 0.158 | 0.144 | 0.237 | 0.105 |
| $SO_2$: sulfur dioxide (mg/m$^3$) | 0.037 | 0.056 | 0.023 | 0.024 |
| $NO_2$: nitrogen dioxide (mg/m$_3$) | 0.027 | 0.034 | 0.035 | 0.038 |
| % of drinking water meeting standards | 93.00 | 97.96 | 98.80 | 96.43 |
| % of surface water meeting standards | 83.00 | 91.67 | 62.00 | 88.89 |
| Noise (dB(A)) | 55.80 | 56.90 | 53.20 | 55.50 |
| **Total industrial pollution emissions** | | | | |
| Waste water (10,000 tons) | 1,674 | 14,010 | 3,774 | 4,544 |
| COD: chemical oxygen demand (tons) | 1,708 | N.A. | 6,787 | 25,200 |
| Waste gas (100 million m$^3$) | 129 | 471 | 461 | 1,895 |
| Smoke (tons) | 11,063 | 8,611 | 5,385 | 47,421 |
| $SO_2$ (tons) | 8,863 | 21,492 | 35,765 | 96,377 |
| Solid waste (10,000 tons) | 1 | 8 | N.A. | 265 |

Sources: Municipal governments of the four cities.

Following the success of the pilot program in Zhenjiang in 1999, Huaian, Wuxi, and Yangzhou adopted the same program in 2001. Table 2 provides information on socioeconomic and environmental conditions in the four cities, as well as polluting emissions in 2001. Wuxi had the largest population as well as the highest GDP per capita, while Huaian was the poorest. Wuxi and Yangzhou had the lowest readings for air quality, measured by $SO_2$ (sulfur

**Table 3**. Distributions of Sample Firms During 1997-2001

***Total Number of Rated (R) and Nonrated (NR) Firms by City and Year***

| Year | Status | Huaian | Wuxi | Yangzhou | Zhenjiang | Total |
|---|---|---|---|---|---|---|
| 1997[a] | Unrated | 42 | 33 | 64 | 89 | 228 |
| 1998[a] | Unrated | 46 | 26 | 71 | 81 | 224 |
| 1999[a] | Unrated | 46 | 32 | 76 | 12 | 166 |
| | Rated | | | | 57 | 57 |
| 2000 | Unrated | 54 | 43 | 68 | 13 | 178 |
| | Rated | | | | 78 | 78 |
| 2001 | Unrated | 16 | 1 | 13 | 131 | 161 |
| | Rated | 39 | 69 | 59 | 91 | 258 |
| 1997-2001 | Unrated | 204 | 135 | 292 | 326 | 957 |
| | Rated | 39 | 69 | 59 | 226 | 393 |
| | Total | 243 | 204 | 351 | 552 | 1,350 |

***Distribution of Rated Firms by Rating Colors****

| | Green | Blue | Yellow | Red | Black | Rated |
|---|---|---|---|---|---|---|
| ZhenJiang (1999) | 1 (1.75) | 34 (59.65) | 17 (29.82) | 4 (7.02) | 1 (1.75) | 57 |
| Zhenjiang (2000) | 2 (2.56) | 48 (61.54) | 19 (24.36) | 8 (10.26) | 1 (1.28) | 78 |
| Huai'an (2001) | 2 (5.13) | 29 (74.36) | 4 (10.26) | 3 (7.69) | 1 (2.56) | 39 |
| Wuxi (2001) | 15 (21.74) | 20 (28.99) | 20 (28.99) | 8 (10.59) | 6 (8.70) | 69 |
| Yangzhou (2001) | 2 (3.39) | 52 (88.14) | 5 (8.47) | 0 (0.00) | 0 (0.00) | 59 |
| Zhenjiang (2001) | 2 (2.20) | 60 (65.93) | 20 (21.98) | 7 (7.69) | 2 (2.20) | 91 |
| All cities (1998-2001) | 24 (6.11) | 243 (61.83) | 85 (21.63) | 30 (7.63) | 11 (2.80) | 393 |

*Figures in parentheses represent the percent of firms by rating colors.

[a]Green Watch began in Zhenjiang in 1999 and in the other three cities in 2001. Thus, no firms were rated in 1997 and 1998, and only some firms in Zhenjiang were rated in 1999 and 2000.

dioxide) and $NO_2$ (nitrogen dioxide), and water quality measured by TSS (total suspended solids) and regulatory compliance percentage. Their pollution monitoring, inspection, and environmental information systems were well developed and well managed, primarily because of their long-standing experience with pollution registration requirements and China's pollution charge system.[1]

Table 3 shows that 36.7% of the firms in the sample were rated by the Green Watch program. The majority of the firms were assigned blue (60.38%) and yellow (22.37%); only a few earned the best (green) rating (2.96 %) or the worst (black—2.96%). The rating distributions were similar across cities, with the majority of firms rated blue and yellow, and very few green and black.

## 5. Multivariate Analysis

The pollution data collected for this study were sufficiently detailed to permit assessment of Green Watch for both water and air pollution, measured by intensity and effluent concentration. Pollution intensity was calculated as the total emissions divided by the gross value of output. We use TSS, chemical oxygen demand (COD), and generated waste water to measure water pollution, and sulfur dioxide ($SO_2$), waste gas, dust, and smoke to measure air pollution.

The dependent variables in our multivariate analyses are changes in pollution intensity and concentration for different pollutants. Let pollution intensity be specified as $Y_{it}$ for firm $i$ in year $t$. The dependent variable for the intensity equation is the first difference, $Y_{i,t} - Y_{i,t-1}$. The reduced-form fixed effects model for $Y_{i,t} - Y_{i,t-1}$ is

$$(1)\ Y_{it} - Y_{it-1} = \beta_0 + \boldsymbol{\alpha}_1 \mathbf{F}_{it} + \boldsymbol{\alpha}_2 \mathbf{C}_{it} + \boldsymbol{\alpha}_3 \mathbf{R}_{it} + \beta_1 t + \mu_i + \varepsilon_{it.}$$

where $\mathbf{F}_{it}$ and $\mathbf{C}_{it}$ are vectors of characteristics of the firm and the city; $\mathbf{R}_{it}$ is a vector that incorporates both rating status (rated or unrated) and color-category assignments for rated firms; t is a time trend; $\mu_i$ represents unobserved firm effects; and $\varepsilon_{it}$ is a random error term. Specifically, firm characteristics in $\mathbf{F}_{it}$ include the number of establishment years, firm size (e.g., large, medium, and small), ownership structure (e.g., state-owned enterprise, collectively owned enterprise, private companies, foreign companies, and companies with limited shares), and industry sectors (e.g., chemicals, fiber/rubber/plastic, food and beverage, machinery manufacture, medical, mining, pulp and paper, smelting, textile and leather, transportation, and utilities). City dummies are incorporated in $\mathbf{C}_{it}$ to reflect differences between cities. Endogeneity is not a serious problem in this case, because ratings released in year t were based on multi-dimensional performance observations during year $t$ – 1.

If sample firms were randomly assigned to rated and unrated groups, we would not expect a statistical difference in intergroup pollution at $t$ −1, before the first Green Watch disclosure in period $t$.

1 For discussion of firm-level pollution data in China, see Wang and Wheeler (2006).

**Table 4**. Pollution Intensities and Effluent Concentrations for Unrated Firms and Firms Rated for the First Time

| | Pollution Intensity | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| | Water | TSS | COD | $SO_2$ | Dust/ Smoke | Gas | TSS | COD | $SO_2$ | Dust/ Smoke |
| ***Zhenjiang launched its pilot Green Watch Program in 1999*** | | | | | | | | | | |
| Unrated | 50.06 (103.13) | 68.05 (190.49) | 63.44 (177.37) | 0.58 (1.70) | 0.38 (0.81) | 6.38 (13.39) | 115.44 (51.90) | 170.43 (97.95) | 183.52 (343.55) | 199.82 (166.58) |
| Rated | 6.93 (19.94) | 26.71 (63.85) | 23.82 (67.41) | 0.04 (0.11) | 0.01 (0.06) | 2.30 (5.68) | 93.54 (79.60) | 179.51 (227.12) | 413.45 (522.20) | 485.60 (709.19) |
| Equal mean test | 3.64* | 0.13 | 0.82 | 0.42 | 3.21* | 0.52 | 3.77** | 1.30 | 0.73 | 0.67 |
| Equal median test | 5.14** | 0.15 | 2.73* | 0.17 | 0.48 | 0.10 | 6.71*** | 3.25* | 1.99 | 2.58* |
| ***Three cities (Huaian, Wuxi and Yangzhou) launched Green Watch programs in 2001*** | | | | | | | | | | |
| Unrated | 76.74 (245.71) | 8.79 (28.67) | 1.89 (6.67) | 0.16 (0.49) | 0.11 (0.31) | 24.17 (102.11) | 95.83 (56.86) | 278.24 (304.49) | 967.66 (999.83) | 253.96 (218.00) |
| Rated | 143.92 (492.41) | 30.34 (94.04) | 119.72 (556.27) | 0.13 (0.68) | 0.06 (0.28) | 28.81 (148.89) | 120.27 (335.57) | 229.84 (650.73) | 1284.55 (2476.77) | 295.01 (784.40) |
| Equal mean test | 2.93** | 0.06 | 6.83*** | 1.67 | 1.25 | 2.42 | 5.06** | 8.71*** | 0.01 | 0.22 |
| Equal median test | 7.54*** | 1.23 | 2.58* | 0.05 | 0.83 | 1.52 | 7.02*** | 10.57*** | 0.14 | 0.23 |

Note: *is for a significance level of 10%, ** for 5%, and ***for 1%, respectively.

Assessing prior randomness is complicated in this case by the distributions of pollution intensity and effluent concentration. Both are highly skewed, with skewness coefficients ranging from 3 to 9. In this case, the traditional student t test for equality of pre-rating group means is not appropriate. We employ the nonparametric Wilcoxon–Mann–Whitney test for equal means and the K-sample test for equal medians. Our results, reported in Table 4, show that significant differences in means and medians were common in the sample. In Zhenjiang, where Green Watch began in 1999, we find significant differences in mean and/or median pollution intensities for waste water, COD, and dust and smoke as well as significant differences in mean and/or median effluent concentrations for TSS, COD, and dust and smoke. Table 4 reports similar findings for the other three sample cities (Huaian, Wuxi, and Yangzhou), where the first public disclosure of ratings occurred in early 2001.[2] In light of these results, it is appropriate to introduce controls for pre-program pollution in our estimating equation:

(2) $Y_{it} - Y_{it-1} = \beta_0 + \boldsymbol{\alpha_1} \mathbf{F_{it}} + \boldsymbol{\alpha_2} \mathbf{C_{it}} + \boldsymbol{\alpha_3} \mathbf{R_{it}} + \beta_1 t + \beta_2 Y_{it-1} + \mu_{it} + \varepsilon_{it}$

To determine the appropriate estimator, we employ Breusch and Pagan Lagrangian multiplier (BPLM) tests for random effects. We reject the null hypothesis in favor of the random effects model for air pollution intensities, and for air and water effluent concentrations. We assume that $\varepsilon_{it}$ was correlated across firms within a city but uncorrelated across firms in different cities.

Tables 5/6 and 7/8 present estimation results for changes in pollution intensity and effluent concentration, respectively. In Tables 5 and 7, we test whether a firm reduces pollution simply because it was rated. A priori, it was possible that self-scrutiny by a rated firm resulted in better environmental management and reduced pollution, even if the firm had a good rating. Our results for the regression variable PRD were consistent with this hypothesis: PRD rating had a negative impact on pollution for all equations in Tables 5 and 7, and a statistically significant impact on TSS and $SO_2$ for pollution intensity, and dust and smoke for effluent concentration.

Tables 6 and 8 provide more insights, by identifying specific color ratings for firms. We find strong results for water pollution intensity (TSS and COD) and dust-and-smoke intensity in Table 6, with highly significant reductions for the poorly rated firms that were much larger than reductions for the firms with better ratings. Intensities generally declined more among the rated firms for the other pollutants as well, but without the striking differential for the poorly rated firms. The same general pattern holds in Table 8, with generally declining effluent concentrations across all rated firms and the largest impacts among the poorly rated firms. Although some concentration results were highly

2 We also conducted the equal mean and median tests for each of the three cities. The results are qualitatively similar.

significant, the overall significance level was somewhat lower than for pollution intensities. Across both tables, the red-rated firms exhibited stronger responses than the black-rated firms.

**Table 5**. Estimation Results for Pollution Intensity Increases: Rated Versus Unrated Firms

| | | Water Pollution | | Air Pollution | | |
|---|---|---|---|---|---|---|
| | Waste Water | TSS | COD | $SO_2$ | Waste Gas | Dust and Smoke |
| PRD | −−41.62 | −15.47** | −18.23 | −0.05 | −11.06 | −0.02 |
| | (29.35) | (7.82) | (21.85) | (0.06) | (12.96) | (0.03) |
| Lagged pollution | −0.56* | −0.94*** | −1.01*** | −1.00*** | −0.75*** | −0.80*** |
| intensity | (0.31) | (0.15) | (0.17) | 0.00 | (0.27) | (0.16) |
| City dummies (base = Wuxi) | | | | | | |
| Huanan | −277.03*** | −11.29*** | −25.94 | −0.02 | −6.27* | 0.03*** |
| | (26.99) | (3.79) | (23.11) | (0.01) | (3.24) | (0.01) |
| Yangzhou | −259.84*** | −6.37 | −24.28 | −0.09*** | −14.56* | −0.02*** |
| | (24.50) | (8.66) | (42.11) | (0.02) | (7.48) | (0.01) |
| Zhengjiang | −287.32*** | 0.07 | −6.66 | −0.07*** | −11.41* | 0.00 |
| | (36.43) | (5.31) | (37.96) | (0.01) | (6.51) | (0.01) |
| Firm size (base = small) | | | | | | |
| Large | 16.31** | −14.18 | −29.13 | −0.07*** | −6.53*** | −0.02 |
| | (7.20) | (10.29) | (37.82) | (0.02) | (1.50) | (0.02) |
| Medium | 30.02 | −14.73** | −22.42 | −0.02 | −0.01 | −0.01 |
| | (30.30) | (6.01) | (25.97) | (0.03) | (6.48) | (0.02) |
| Ownership structure (base = private) | | | | | | |
| State-owned | −114.86** | 14.66* | 23.14 | (0.03) | (9.15) | (0.02) |
| | (52.68) | (8.27) | (19.18) | (0.09) | (6.44) | (0.05) |
| Collectively | −113.49* | 12.91*** | 1.66 | −0.06 | −16.09** | −0.02 |
| owned | (61.29) | (3.53) | (19.67) | (0.08) | (6.73) | (0.05) |
| HK, Macao, and | −169.5 | 46.81*** | −10.71 | −0.04 | −13.75* | −0.04 |
| Taiwan investor | (158.32) | (17.24) | (19.96) | (0.08) | (7.60) | (0.04) |
| Foreign investor | −17.23 | 9.4 | −2.85 | −0.06 | −21.77*** | 0.01 |
| | (46.86) | (12.93) | (5.59) | (0.07) | (8.24) | (0.05) |
| Companies with | −113.59* | 23.82* | 51.00 | (0.06) | −17.30*** | (0.03) |
| limited shares | (62.37) | (13.86) | (45.32) | (0.07) | (6.11) | (0.04) |
| Others | −110.31*** | 7.27 | 2.67 | (0.08) | −19.37** | (0.04) |
| | (33.46) | (7.43) | (7.98) | (0.07) | (9.22) | (0.04) |

| | | | | | | |
|---|---|---|---|---|---|---|
| Firm age (years) | 0.36* | −0.21** | −0.12** | 0.00 | (0.08) | 0.00 |
| | (0.21) | (0.10) | (0.05) | 0.00 | (0.13) | 0.00 |
| Industry (base = mining) | | | | | | |
| Food and beverages | 41.43 | 18.64 | 17.37 | −0.26*** | 4.98 | −0.32*** |
| | (84.23) | (16.99) | (10.80) | (0.03) | (7.14) | (0.07) |
| Textiles and leather | −57.54 | 11.97 | 25.70* | −0.19** | 3.85 | −0.30*** |
| | (48.65) | (9.28) | (14.86) | (0.08) | (3.96) | (0.06) |
| Pulp and paper | −39.94 | 47.03** | 95.62 | −0.26*** | 3.27 | −0.33*** |
| | (69.45) | (19.59) | (72.80) | (0.04) | (3.70) | (0.07) |
| Chemicals | −18.07 | 27.17 | 86.95** | −0.15** | 8.61* | −0.26*** |
| | (55.47) | (18.02) | (43.88) | (0.07) | (4.93) | (0.07) |
| Medical | −28.76 | 7.18 | 46.21** | −0.28*** | 2.59 | −0.33*** |
| | (50.67) | (8.67) | (18.42) | (0.02) | (3.73) | (0.06) |
| Fiber, rubber and | −33.66 | 19.83 | 9.34 | −0.26*** | 20 | −0.33*** |
| plastic | (65.93) | (26.68) | (15.45) | (0.04) | (12.85) | (0.07) |
| Smelting | (52.41) | 4.56 | 11.52 | −0.29*** | 0.81 | −0.33*** |
| | (66.49) | (12.51) | (14.89) | (0.03) | (5.51) | (0.07) |
| Machinery | (100.65) | 11.87 | 12.53** | −0.28*** | 0.50 | −0.33*** |
| manufacture | (89.26) | (11.87) | (6.02) | (0.02) | (1.63) | (0.06) |
| Utilities | 145.69* | 5.53 | 13.66 | −0.12*** | 33.50 | −0.27*** |
| | (74.49) | (16.52) | (39.68) | (0.03) | (35.80) | (0.08) |
| Transportation | (58.63) | 20.67* | (2.78) | −0.30*** | (1.72) | −0.32*** |
| | (67.32) | (11.46) | (8.49) | (0.06) | (3.95) | (0.08) |
| Others | −24.54 | 9.74 | 12.69 | −0.24*** | 1.12 | −0.31*** |
| | (25.03) | (16.18) | (8.75) | (0.04) | (3.19) | (0.06) |
| Time trend | 13.04 | 1.67 | 2.46 | 0.00 | 3.28 | 0.00 |
| | (8.84) | (2.50) | (6.27) | (0.02) | (4.18) | (0.01) |
| Constant | 377.03*** | (0.18) | 2.11 | 0.43*** | 18.97*** | 0.36*** |
| | −107.85 | −20.45 | −34.02 | −0.08 | −6.75 | −0.07 |
| No. of obs | 1,320 | 1,128 | 1,296 | 1,158 | 1,229 | 1,104 |
| within $R^2$ | 0.15 | 0.61 | 0.73 | 0.99 | 0.49 | 0.82 |
| between $R^2$ | 0.16 | 0.52 | 0.84 | 0.97 | 0.33 | 0.61 |
| overall $R^2$ | 0.14 | 0.39 | 0.77 | 0.99 | 0.33 | 0.62 |
| Breusch and Pagan Lagrangian Multiplier test for random effects | | | | | | |
| Test statistics: $\chi^2(1)$ | 0.04 | 0.19 | 0.05 | 75.13*** | 5.27** | 21.68*** |

Numbers in parentheses are standard errors of the estimated coefficients. ***, **, and * represent significance at the 1%, 5%, and 10% levels, respectively.

**Table 6.** Estimation Results for Pollution Intensity Increases: Five-Color Ratings

| | | Water | | Air | | |
|---|---|---|---|---|---|---|
| | Waste Water | TSS | COD | $SO_2$ | Waste Gas | Dust/ Smoke |
| Rating dummies (base = not rated) | | | | | | |
| Green | −−46.24 | −4.37 | −0.51 | −0.05** | −6.06** | 0.01 |
| | (53.17) | (6.63) | (4.01) | (0.02) | (2.45) | (0.01) |
| Blue | −12.35 | −9.18*** | −5.1 | −0.03** | −4.26 | 0.00 |
| | (8.56) | (2.39) | (4.58) | (0.02) | (6.20) | (0.01) |
| Yellow | −16.32 | −13.69* | −30.51 | −0.04* | −5.84 | −0.03** |
| | (18.19) | (7.32) | (23.78) | (0.02) | (3.88) | (0.01) |
| Red | 24.03 | −35.14*** | −44.40*** | −0.05** | −5.48* | −0.03* |
| | (30.88) | (5.76) | (10.95) | (0.02) | (2.87) | (0.01) |
| Black | −11.4 | −19.72*** | −25.06*** | −0.15 | −5.3 | −0.16* |
| | (32.89) | (2.67) | (8.34) | (0.12) | (5.81) | (0.09) |
| Lagged pollution | −0.56* | −0.94*** | −1.00*** | −1.00*** | −0.75*** | −0.80*** |
| intensity | (0.31) | (0.15) | (0.16) | 0.00 | (0.27) | (0.16) |
| City dummies (base = Wuxi) | | | | | | |
| Huanan | −270.10*** | −11.20*** | −27.55 | −0.02* | −5.79** | 0.02** |
| | (23.91) | (3.13) | (22.27) | (0.01) | (2.32) | (0.01) |
| Yangzhou | −251.97*** | −6.64 | −26.67 | −0.10*** | −14.27** | −0.03*** |
| | (23.46) | (7.87) | (40.37) | (0.02) | (6.37) | (0.01) |
| Zhengjiang | −275.58*** | −0.24 | −6.82 | −0.07*** | −12.54* | 0.00 |
| | (30.74) | (4.41) | (33.79) | (0.01) | (7.21) | (0.00) |
| Firm size (base = small) | | | | | | |
| Large | 13.66** | −15.44 | −30.23 | −0.07*** | −8.09*** | −0.02* |
| | (6.36) | (11.08) | (39.11) | (0.02) | (2.57) | (0.01) |
| Medium | 24.35 | −16.17** | −23.81 | −0.03 | −2.02 | −0.01 |
| | (30.51) | (7.55) | (29.53) | (0.02) | (4.53) | (0.01) |
| Ownership structure (base = private) | | | | | | |
| State-owned | −113.99** | 15.31* | 23.5 | −0.03 | −8.77 | −0.02 |
| | (47.38) | (9.17) | (20.86) | (0.09) | (7.29) | (0.05) |
| Collectively owned | −106.28** | 14.33*** | 2.7 | −0.06 | −15.24** | −0.02 |
| | (47.70) | (3.97) | (17.08) | (0.09) | (6.85) | (0.05) |

| | | | | | | |
|---|---|---|---|---|---|---|
| HK, Macao and Taiwan investor | −146.61 | 48.76*** | −8.14 | −0.04 | −10.38* | −0.04 |
| | (178.46) | (17.67) | (27.85) | (0.07) | (5.78) | (0.04) |
| Foreign investor | 19.3 | 11.12 | −0.84 | −0.06 | −19.02*** | 0.01 |
| | (56.03) | (11.37) | (7.19) | (0.07) | (6.22) | (0.05) |
| Companies with limited shares | −99.66** | 25.36* | 52.34 | −0.05 | −16.26** | −0.03 |
| | (48.54) | (14.97) | (50.35) | (0.07) | (6.31) | (0.04) |
| Others | −92.33*** | 11.51* | 10.45 | −0.07 | −17.10** | −0.03 |
| | (29.16) | (6.84) | (16.47) | (0.06) | (7.77) | (0.04) |
| Firm age (years) | 0.32*** | −0.21** | −0.07 | 0 | −0.1 | 0 |
| | (0.11) | (0.10) | (0.06) | 0.00 | (0.14) | 0.00 |
| Industry (base = mining) | | | | | | |
| Food and beverages | 50.26 | 19.02 | 16.87* | −0.26*** | 3.8 | −0.31*** |
| | (89.18) | (16.31) | (9.34) | (0.03) | (5.06) | (0.08) |
| Textiles and leather | −64 | 13.86* | 27.70* | −0.19** | 3.33 | −0.29*** |
| | (49.92) | (7.39) | (14.67) | (0.09) | (2.74) | (0.07) |
| Pulp and paper | −45.68 | 47.79** | 94.92 | −0.27*** | 2.27 | −0.33*** |
| | (77.88) | (20.04) | (69.00) | (0.04) | (2.70) | (0.07) |
| Chemicals | −25.34 | 27.33 | 85.24* | −0.15** | 7.33* | −0.26*** |
| | (62.05) | (18.48) | (46.95) | (0.07) | (4.23) | (0.07) |
| Medical | −32.38 | 8.39 | 47.90*** | −0.28*** | −0.05 | −0.33*** |
| | (49.98) | (8.36) | (17.31) | (0.02) | (2.53) | (0.07) |
| Fiber, rubber and plastic | −38.3 | 19.67 | 5.93 | −0.26*** | 18.71 | −0.32*** |
| | (65.87) | (26.08) | (20.49) | (0.04) | 13.91) | (0.07) |
| Smelting | −49.93 | 4.12 | 10.07 | −0.29*** | 0.03 | −0.33*** |
| | (62.06) | (11.75) | (14.23) | (0.03) | (4.38) | (0.07) |
| Machinery manufacture | −102.24 | 11.2 | 9.34*** | −0.29*** | −0.87 | −0.32*** |
| | (87.87) | (12.63) | (3.44) | (0.02) | (1.05) | (0.06) |
| Utilities | 154.48* | 4.44 | 10.57 | −0.12*** | 31.06 | −0.27*** |
| | (90.40) | (16.83) | (36.76) | (0.04) | 32.86) | (0.08) |

| | | | | | | |
|---|---|---|---|---|---|---|
| Transportation | −42.63 | 23.17*** | −0.19 | −0.29*** | 3.27 | −0.32*** |
| | (60.82) | (6.49) | (10.26) | (0.04) | (2.84) | (0.07) |
| Others | −22.62 | 10.17 | 13.15* | −0.24*** | 0.85 | −0.30*** |
| | (32.06) | (15.22) | (7.11) | (0.03) | (2.27) | (0.06) |
| Time trend | 12.82 | 1.67 | 2.52 | 0.00 | 3.28 | 0.00 |
| | (8.56) | (2.52) | (6.32) | (0.02) | (4.22) | (0.01) |
| Constant | 404.84*** | 4.3 | 10.24 | 0.45*** | 30.09*** | 0.37*** |
| | (114.78) | (25.46) | (48.67) | (0.09) | (9.50) | (0.09) |
| No. of obs | 1,320 (4) | 1,128 (4) | 1,296(4) | 1,158 (4) | 1,229 (4) | 1,104 (4) |
| within $R^2$ | 0.15 | 0.61 | 0.73 | 0.99 | 0.49 | 0.82 |
| between $R^2$ | 0.16 | 0.52 | 0.84 | 0.97 | 0.33 | 0.61 |
| overall $R^2$ | 0.14 | 0.39 | 0.77 | 0.99 | 0.33 | 0.62 |
| Breusch and Pagan Lagrangian Multiplier test for random effects | | | | | | |
| Test statistics: $\chi^2(1)$ | 0.06 | 0.14 | 0.05 | 71.64*** | 5.17** | 23.77*** |

Numbers in parentheses are standard errors of the estimated coefficients. ***, **, and * represent significance at the 1%, 5%, and 10% levels, respectively.

---

**Table 7.** Estimation Results for Pollution Concentration Increase: Rated Versus Unrated Firms

| | Water | | Air | |
|---|---|---|---|---|
| | TSS | COD | $SO_2$ | Dust/Smoke |
| PRD | −21.76 | −31.45 | −32.72** | −29.94** |
| | (17.67) | (28.59) | (14.69) | (14.63) |
| Lagged pollution intensity | −0.50* | −0.57*** | −1.02*** | −1.00*** |
| | (0.28) | (0.03) | (0.00) | (0.00) |
| City dummies (base = Wuxi) | | | | |
| Huanan | −44.52 | −32.72*** | −136.48* | −133.30* |
| | (37.69) | (8.98) | (78.80) | (72.49) |
| Yangzhou | 10.99 | −85.95** | −3 | −29.81 |
| | (29.41) | (36.22) | (21.63) | (29.67) |
| Zhengjiang | 9.11 | −50.03* | −40.24 | −44.08 |
| | (37.23) | (25.89) | (32.61) | (28.44) |

| | | | | |
|---|---|---|---|---|
| Firm size (base = small) | | | | |
| Large | 46.17 | −11.1 | 91.68 | 89.42 |
| | (30.65) | (25.74) | (115.56) | (105.23) |
| Medium | 31.62** | 38.21 | 68.88** | 74.29*** |
| | (16.11) | (36.70) | (33.95) | (23.99) |
| Ownership structure (base = private) | | | | |
| State-owned | −7.97 | −48.77** | 2.79 | −10.12 |
| | (31.43) | (22.30) | (81.34) | (99.86) |
| Collectively owned | −24.28 | −81.77 | −75.84*** | −78.17 |
| | (31.73) | (50.92) | (29.33) | (51.18) |
| HK, Macao and Taiwan investor | −71.65** | −123.66*** | −150.52 | −146.23 |
| | (35.61) | (40.66) | (95.66) | (90.76) |
| Foreign investor | −23.82 | −78.31** | −70.96 | −74.38 |
| | (41.25) | (35.54) | (48.88) | (52.14) |
| Companies with limited shares | −30.64 | −68.33 | −104.19*** | −87.09** |
| | (23.55) | (48.15) | (40.15) | (43.20) |
| Others | −46.89** | −103.96** | −33.63 | −31.56 |
| | (18.73) | (47.75) | (55.95) | (69.48) |
| Firm age (years) | 0.17 | 0.7 | 0.43 | 0.38 |
| | (0.42) | (0.87) | (0.64) | (0.68) |
| Industry (base = mining) | | | | |
| Food and beverages | 31.87 | −100.11* | 70.79 | 3.7 |
| | (27.39) | (51.64) | (47.65) | (26.30) |
| Textiles and leather | 14.72* | −71.75 | 18.11 | −25.07 |
| | (8.15) | (60.39) | (22.11) | (23.43) |
| Pulp, paper and print | 41.38* | −48.51 | 53.41* | 6.48 |
| | (23.19) | (30.64) | (31.95) | (43.91) |
| Chemicals | 68.33*** | −24.58 | 155.20*** | 96.44*** |
| | (22.19) | (60.20) | (37.76) | (35.55) |
| Medical | 272.51 | 79.85 | 681.55* | 571.31* |
| | (190.54) | (108.69) | (364.86) | (345.14) |
| Fiber, rubber and plastic | 28.79 | −54.47* | 41.76 | −18.84 |
| | (28.23) | (28.32) | (73.15) | (17.63) |
| Smelting | 15.55 | −28.35* | 41.75 | −21.98 |
| | (21.69) | (15.67) | (32.05) | (30.01) |

| | | | | |
|---|---|---|---|---|
| Machinery manufacture | 6.16 | −54.18 | −26.72 | −59.85 |
| | (12.24) | (47.44) | (21.91) | (41.71) |
| Utilities | 20.87** | 2.2 | 0.62 | −46.04 |
| | (10.09) | (63.66) | (48.36) | (70.90) |
| Transportation | 144.51* | 95.16*** | 439.38*** | 379.11*** |
| | (84.83) | (10.56) | (94.84) | (58.23) |
| Others | 40.96*** | −10.94 | 29.16 | −20.65 |
| | (15.68) | (44.81) | (70.07) | (38.23) |
| Time | 3.84 | 6.78 | 3.14 | 1.20 |
| | (8.03) | (7.90) | (18.50) | (17.48) |
| Constant | −3.13 | 129.43** | 70.99 | 130.67 |
| | (73.45) | (51.03) | (124.84) | (103.53) |
| No. of obs | 967 | 914 | 659 | 664 |
| within $R^2$ | 0.53 | 0.01 | 0.14 | 0.96 |
| between $R^2$ | 0.11 | 0.76 | 0.98 | 0.95 |
| overall $R^2$ | 0.24 | 0.57 | 0.96 | 0.96 |
| Breusch and Pagan Lagrangian Multiplier test for random effects | | | | |
| Test statistics: $\chi^2(1)$ | 9.05*** | 8.91*** | 8.61** | 8.52*** |

Numbers in parentheses are standard errors of the estimated coefficients. ***, **, and * represent significance at the 1%, 5%, and 10% levels, respectively.

**Table 8.** Estimation Results for Pollution Concentration Increase: Five-Color Ratings

| | Water Pollution | | Air Pollution | |
|---|---|---|---|---|
| | TSS | COD | $SO_2$ | Dust/Smoke |
| Rating dummies (base = not rated) | | | | |
| Green | −38.95 | −55.95*** | −46.42 | −40.46 |
| | (25.97) | (9.49) | (70.40) | (75.08) |
| Blue | −7.09 | −3.83 | −12.14 | −7.70 |
| | (18.04) | (19.48) | (23.17) | (23.33) |
| Yellow | −41.46 | −63.43 | −54.54*** | −53.79*** |
| | (26.54) | (39.15) | (12.15) | (10.35) |
| Red | −68.47** | −77.8 | −101.34* | −100.56 |
| | (34.07) | (59.15) | (59.88) | (62.64) |

| | | | | |
|---|---|---|---|---|
| Black | 9.02 | −109.21** | 33.02 | 23.29 |
| | (11.58) | (48.53) | (28.47) | (19.68) |
| Lagged pollution intensity | −0.50* | −0.57*** | −1.01*** | −1.00*** |
| | (0.28) | (0.03) | (0.00) | (0.00) |
| City dummies (base = Wuxi) | | | | |
| Huanan | −47.82 | −40.70*** | −137.09* | −134.01* |
| | (38.77) | (8.29) | (79.30) | (73.40) |
| Yangzhou | 8.14 | −93.54*** | −4.91 | −31.37 |
| | (30.45) | (35.12) | (24.20) | (31.73) |
| Zhengjiang | 8.39 | −54.57** | −39.73 | −43.68 |
| | (37.45) | (24.41) | (33.35) | (29.12) |
| Firm size (base = small) | | | | |
| Large | 47.88 | −9.47 | 92.21 | 90.07 |
| | (29.90) | (24.87) | (116.47) | (105.35) |
| Medium | 31.91** | 37.46 | 69.04** | 74.09*** |
| | (15.63) | (37.44) | (34.67) | (24.31) |
| Ownership structure (base = private) | | | | |
| State-owned | −7.00 | −45.87** | 4.27 | −7.18 |
| | (32.37) | (19.54) | (84.70) | (102.78) |
| Collectively owned | −21.71 | −77.1 | −73.43** | −74.79 |
| | (31.05) | (47.51) | (32.93) | (52.46) |
| HK, Macao and Taiwan investor | −65.87* | −116.33*** | −140.2 | −135.37 |
| | (35.41) | (37.48) | (96.19) | (90.21) |
| Foreign investor | −17.34 | −67.09** | −59.55 | −59.53 |
| | (38.43) | (32.29) | (46.26) | (51.74) |
| Companies with limited shares | −28.13 | −62.52 | −101.31** | −83.70* |
| | (23.66) | (48.72) | (41.87) | (45.07) |
| Others | −42.68** | −88.13** | −30.49 | −26.83 |
| | (21.08) | (43.05) | (65.66) | (79.01) |
| Firm age (years) | 0.2 | 0.71 | 0.46 | 0.42 |
| | (0.46) | (0.95) | (0.67) | (0.70) |
| Industry (base = mining) | | | | |
| Food and beverages | 33.51 | −100.96* | 70.56 | 1.46 |
| | (30.03) | (55.52) | (49.40) | (26.47) |
| Textiles and leather | 18.65** | −67.9 | 22.38 | −22.09 |
| | (8.78) | (61.76) | (18.45) | (17.09) |
| Pulp and paper | 50.24* | −44.35 | 69.28* | 20.58 |
| | (27.29) | (31.55) | (36.85) | (49.40) |

| | | | | |
|---|---|---|---|---|
| Chemicals | 72.18*** | −27.88 | 160.67*** | 100.26*** |
| | (24.02) | (60.28) | (36.49) | (31.95) |
| Medical | 279.24 | 79.76 | 686.83* | 573.04* |
| | (194.64) | (111.22) | (366.71) | (346.37) |
| Fiber, rubber and plastic | 29.2 | −59.85* | 40.74 | −22.71 |
| | (28.77) | (32.48) | (74.32) | (17.33) |
| Smelting | 18.62 | −30.11* | 46.8 | −20.37 |
| | (23.73) | (17.80) | (34.14) | (29.70) |
| Machinery manufacture | 7.39 | −60.81 | −24.94 | −61.55 |
| | (12.60) | (50.83) | (19.93) | (41.75) |
| Utility | 21.58** | −4.32 | 0.34 | −49.59 |
| | (9.66) | (68.11) | (47.24) | (71.06) |
| Transportation | 147.44* | 93.82*** | 442.96*** | 380.16*** |
| | (86.75) | (11.30) | (95.04) | (57.75) |
| Others | 46.08*** | −11.19 | 36.24 | −16.45 |
| | (17.25) | (47.13) | (71.49) | (39.33) |
| Time trend | 3.5 | 5.87 | 2.79 | 0.95 |
| | (8.26) | (7.28) | (18.68) | (17.70) |
| Constant | −6.66 | 134.60*** | 65.27 | 126.36 |
| | (70.41) | (46.86) | (122.78) | (99.61) |
| No. of obs (clusters) | 967 (3) | 914 (4) | 659 (4) | 664 (4) |
| within $R^2$ | 0.53 | 0.01 | 0.14 | 0.96 |
| between $R^2$ | 0.12 | 0.77 | 0.98 | 0.95 |
| overall $R^2$ | 0.24 | 0.57 | 0.96 | 0.96 |
| Breusch and Pagan Lagrangian Multiplier test for random effects | | | | |
| Test statistics: $\chi^2(1)$ | 9.05*** | 8.91*** | 8.61** | 8.52*** |

Numbers in parentheses are standard errors of the estimated coefficients. ***, **, and * represent significance at the 1%, 5%, and 10% levels, respectively.

## 6. Conclusions

This study has employed a new panel data set to test the impact of environmental PRD on polluting firms in Jiangsu province, China. The data consist of ex ante and ex post pollution measures for both rated and unrated firms, enabling us to control for confounding factors such as time-variant technology improvement and selection bias. Our results strongly suggest that Green Watch had significantly reduced pollution for the rated firms, with particularly strong impacts on firms with poor ratings. Among the poorly rated red and black firms, the impact was generally greater on the red-rated firms that were closer to compliance with regulations. The reasons for these responses could be that the incentive for improvement that the Green Watch generates was stronger for firms with poor ratings than those with good ratings, and that the abatement costs for the red-rated firms to achieve compliance were lower than those black-rated firms, even though the pressure for improvement could be stronger with the black-rated firms than the red-rated firms.

This research also adds some insights to the growing comparative literature on PRDs. After studying PRD experiences in Indonesia (PROPER) and the Philippines (EcoWatch), Dasgupta, Wang and Wheeler (2006) argue that PRD programs are most effective in moving moderately noncompliant firms into compliance with regulations, but may provide insufficient incentives to induce significant improvements by the worst performers or firms with good ratings. However, our results for Green Watch indicate significant impacts for firms with good (green and blue) ratings. The stronger result for the four cities in Jiangsu Province may stem from two additional benefits for the green-rated firms: **(a)** enterprises awarded green in a particular year could be given priority consideration in the selection of enterprises with the best economic and social performance records and (b) an enterprise rated green for three consecutive years was given preferential status by provincial environmental regulators. The Jiangsu experience suggests that PRD programs could effectively improve environmental performance even for good performers if the programs could target highly rated firms for additional benefits beyond reputational improvement.

We envision several directions for future research. First, this study focuses on the Green Watch programs adopted in four cities in Jiangsu province. A large scale or even national level study is merited for the generation of the findings. Secondly, the whole Green Watch program had been revised in 2013. The question how the effectiveness has been changed along with the program revision naturally arises.

## References

Afsah, S., B. Laplante, and D. Wheeler. 1997. *Regulation in the Information Age: Indonesian Public Information Program for Environmental Management*. Research Paper. Washington, DC: World Bank.

Arora, S., and T. Cason. 1998. "Do Community Characteristics Influence Environmental Outcomes? Evidence from the Toxics Release Inventory." *Journal of Applied Economics* 1: 413-53.

Bennett, M., P. James, and L. Klinkers. 2017. "Environmental Performance Evaluation and Reporting in Developing Countries: The Case of Indonesia's Programme for Pollution Control, Evaluation and Rating (PROPER)." In *Sustainable Measures: Evaluation and Reporting of Environmental and Social*

*Performance,* edited by Bennett, James and Klinkers . 185-201., London: Routledge.

Blackman, A., and G.J. Banniseter. 1998. "Community Pressure and Clean Technology in the Informal Sector: an Econometric Analysis of the Adoption of Propane by Traditional Mexican Brick Makers." *Journal of Environmental and Economic Management* 35: 1-21.

Boyle, M.A., and K.A. Kiel. 2001. "A Survey of House Price Hedonic Studies of the Impact of Environmental Externalities." *Journal of Real Estate Literature* 9: 117-44.

Bui, L.T., and C.J. Mayer. 2003. "Regulation and Capitalization of Environmental Amenities: Evidence from the Toxic Release Inventory in Massachusetts." *Review of Economics and Statistics* 85: 693-708.

Chaklader, B., and P.A. Gulati. 2015. "A Study of Corporate Environmental Disclosure Practices of Companies Doing Business in India." *Global Business Review* 16 (2): 321-35.

Cormier, D., and M. Magnan. 2015. "The Economic Relevance of Environmental Disclosure and its Impact on Corporate Legitimacy: An Empirical Investigation." *Business Strategy and the Environment* 24 (6): 431-50.

Dasgupta, S., H. Wang, and D. Wheeler. 2006. "Disclosure Strategies for Pollution Control." In *The International Yearbook of Environmental and Resource Economics 2006/2007: A Survey of Current Issues*, edited by T. Tietenberg, and H. Folmer, 93-119. Northampton, MA: Edward Elgar Publishers.

Dasgupta, S., B. Laplante, and N. Mamingi. 2001. "Pollution and Capital Markets in Developing Countries." *Journal of Environmental Economics and Management* 42: 310-35.

Foulon, J., P. Lanoie, and B. Laplante. 2002. "Incentives for Pollution Control: Regulation or Information?" *Journal of Environmental Economics and Management* 44: 169-87.

Gamble, H.B., and R.H. Downing. 1982. "Downing Effects of Nuclear Power Plants on Residential Property Values." *Journal of Regulatory Science* 22: 457-78.

Garcia J.H., T. Sterner, and S. Afsah. 2007. "Public Disclosure of Industrial Pollution: the PROPER Approach for Indonesia?" *Environment, Development and Sustainability* 12: 739-56.

Garcia, J.H., S. Afsah, and T. Sterner. 2009. "Which Firms are More Sensitive to Public Disclosure Schemes for Pollution Control? Evidence from Indonesia's PROPER Program." *Environmental and Resource Economics* 42: 151-68.

Hamilton, J. 1995. "Pollution as News: Media and Stock Market Reaction to the Toxics Release Inventory Data." *Journal of Environmental Economics and Management* 28: 98-113.

Jackson, T.O. 2001. "The Effects of Environmental Contamination on Real Estate: A Literature Review." *Journal of Real Estate Literature* 9: 93-116.

Kurniawan, R. 2017. "Effect of Environmental Performance on Environmental Disclosures of Manufacturing, Mining and Plantation Companies Listed in Indonesia Stock Exchange." *Arthatama Journal of Business Management and Accounting* 1 (1): 6-17.

Keohane, N.O., and S.M. Olmstead. 2016. "Principles of Market-Based Environmental Policy." In *Markets and the Environment*, 139-67 Washington, DC: Island Press.

Kennedy, P.W., B. Laplante, and J. Maxwell. 1994. "Pollution Policy: The Role for Publicly Provided Information." *Journal of Environmental Economics and Management* 26: 31-43.

Kiel, K.A. 1995. "Measuring the Impact of the Discovery and Cleaning of Identified Hazardous Waste Sites on House Value." *Land Economics* 71: 428-35.

Kiel, K.A., and K.T. McClain. 1995. House Prices During Sitting Decision Stages: The Case of an Incinerator from Rumor Through Operation." *Journal of Environmental Economics and Management* 28: 241-55.

Kleindorfer, P.R., and E.W. Orts. 1998. "Information Regulation of Environmental Risks." *Risk Analysis* 18: 155-70.

Kohlase, J.E., 1991. "The Impact of Toxic Waste Site on Housing Values." *Journal of Urban Economics* 30: 1-26.

Konar, S., and M. Cohen. 1997. "Information as Regulation: The Effect of Community Right to Know Laws on Toxic Emissions." *Journal of Environmental Economics and Management* 32: 109-24.

Lonoie, P., B. Laplante, and M. Roy. 1998. "Can Capital Markets Create Incentives for Pollution Control?" *Ecological Economics* 26: 31-41.

Lyon, T.P, and J.P. Shimshack. 2015. "Environmental Disclosure: Evidence from Newsweek's Green Companies Rankings." *Business & Society* 54 (5): 632-75.

Meng, X.H., S.X. Zeng, J.J. Shi. et al. 2014. "The Relationship Between Corporate Environmental Performance and Environmental Disclosure: An Empirical Study in China." *Journal of Environmental Management* 145: 357-67.

Michaels, P.G., and V.K.Smith. 1990. "Market Segmentation and Value Amenities with Hedonic Models: The Case of Hazardous Waste Sites." *Journal of Urban Economics* 28: 223-42.

Oberholzer-Gee, F., and M. Mitsunari. 2006. "Information Regulation: Do the Victims of Externalities Pay Attention?" *Journal of Regulatory Economics* 30: 141-58.

Pargal, S., and D. Wheeler. 1996. "Informal Regulation of Industrial Pollution in Developing Countries: Evidence From Indonesia." *Journal of Political Economy* 104: 1314-27.

Portney, P. 2000. "Environmental Problems and Policy 2000-2050." *Journal of Economic Perspectives* 14: 199-206.

Reichert, A.K. 1997. "Impact of a Toxic Waste Superfund Site on Property Values." *Appraisal Journal* 65: 381-92.

Thayer, M., H. Albers and M. Rahmatian. 1992. "The Benefits of Reducing Exposure to Waste Disposal Sites: A Hedonic Housing Value Approach." *Journal of Real Estate Research* 7: 265-82.

Tietenberg, T. 1998. Disclosure Strategies for Pollution Control." *Environmental and Resource Economics* 11: 587-602.

Wang, H., J. Bi, D. Wheeler, J. Wang, D. Cao, G. Lu, and Y. Wang. 2004. "Environmental Performance Rating and Disclosure: China's Green Watch Program." *Journal of Environmental Management* 71: 123-33.

Wheeler, D., H. Hettige, S. Pargal, and M. Singh. 1997. "Formal and Informal Regulation of Industrial Pollution: Evidence From Indonesia and the U.S." *World Bank Economic Review* 11: 433-50.

*China Policy Journal • Vol. 1, No. 1 • Fall 2018*

# Chronic Noncompliance and Ineffective Enforcement in Guangzhou

Lin Peng[2], Yuan Xu[1,3 *], Hua Zhong[4],
Tai Wei Lim[5], Fengshi Wu[6]

## Abstract

Industrial pollution is the most important cause of China's current environmental crisis. This article concentrates on those firms that continue to seriously deviate from compliance even though they have been repeatedly caught and punished for breaching environmental regulations. This chronic noncompliance is explained through examining formal and informal enforcement activities by the state and by civil society, respectively. Guangzhou, a metropolis located in the heart of the Pearl River Delta, has been selected for this study because of data availability, its relatively more mature civil society, and its political status as a provincial capital. We employ both publicly available government law enforcement data and fieldwork-based first-hand data from individual firms. Special attention is given to the period 2007–2015, during which market fluctuations after the 2008 global financial crisis could have affected the firms' compliance decisions. Formal monitoring and enforcement by the local environmental agencies are found to have

[1] *Department of Geography and Resource Management, The Chinese University of Hong Kong, Hong Kong, China*

[2] *Department of Government and Laws, Guangzhou Academy of Social Sciences, Guangzhou, China*

[3] *Institute of Environment, Energy and Sustainability, The Chinese University of Hong Kong, Hong Kong, China*

[4] *Department of Sociology, The Chinese University of Hong Kong, Hong Kong, China*

[5] *East Asian Institute, National University of Singapore & Singapore University of Social Sciences, Singapore*

[6] *Asia Institute, University of Melbourne, Melbourne, Australia*

[*] *Corresponding author:* yuanxu@cuhk.edu.hk; *Tel: +852-3943 6647*

 doi: 10.18278/cpj.1.1.6

improved compliance, but the regulatory effectiveness was still constrained by the low probability of catching noncompliance and/or insufficient penalty upon conviction. Enforcement activities by civil society played an increasingly visible, yet still complementary, role. Overall, the existing enforcement activities are still inadequate to fundamentally reverse this pattern of chronic noncompliance.

***Keywords:*** Enforcement and compliance; Environmental pollution; Governance; Crime and punishment; China

# Incumplimiento crónico y cumplimiento ineficaz en Guangzhou

## Resumen

La contaminación industrial es la causa más importante de la actual crisis ambiental de China. Este artículo se concentra en aquellas empresas que continúan desviándose seriamente del cumplimiento a pesar de que han sido atrapadas y castigadas repetidamente por infringir las regulaciones ambientales. Este incumplimiento crónico se explica a través del examen de las actividades de cumplimiento formales e informales del estado y la sociedad civil, respectivamente. Guangzhou, una metrópolis ubicada en el corazón del delta del río Perla, ha sido seleccionada para este estudio debido a la disponibilidad de datos, su sociedad civil relativamente más madura y su estatus político como capital provincial. Empleamos tanto datos públicos de aplicación de la ley del gobierno como datos de primera mano basados en el trabajo de campo de empresas individuales. Se presta especial atención al período 2007-2015, durante el cual las fluctuaciones del mercado después de la crisis financiera mundial de 2008 podrían haber afectado las decisiones de cumplimiento de las empresas. Se encuentra que el monitoreo formal y la aplicación por parte de las agencias ambientales locales han mejorado el cumplimiento, pero la efectividad regulatoria aún se vio limitada por la baja probabilidad de que no se detecte el cumplimiento y / o la sanción insuficiente de la condena. Las actividades de cumplimiento por parte de la sociedad civil desempeñaron un papel cada vez más visible, pero aún así complementario. En general, las actividades de cumplimiento existentes aún son inadecuadas para revertir fundamentalmente este patrón de incumplimiento crónico.

***Palabras clave:*** Aplicación y cumplimiento; Contaminación ambiental; Gobernanza; Crimen y castigo; China

# 长期环保违规与广州政府执法的研究

摘要

工业污染是中国当前环境危机最重要的原因。本文聚焦于那些多次因违反环境法规而受到惩罚然而却持续不改的公司。我们研究政府的正式和公民社会的非正式执法活动，从而来解释这种长期违规现象。位于珠江三角洲腹地的大都市广州由于其数据的可获得性、相对成熟的公民社会和作为省会的政治地位，被选为本次研究的对象。由于2008年全球金融危机后的市场波动可能特别影响到公司的环保合规决定，2007-2015年成为研究的重点时段。我们采用了公开的政府执法数据和公司层面基于实地调查的第一手数据。我们的研究发现，尽管当地环境机构的正式监测和执法已经改善了合规情况，但发现违规现象可能性低和(或)定罪后处罚不严仍然限制了监管的有效性。虽然只是辅助，公民社会的执法活动发挥了日益明显的作用。总之，现有的执法活动还不足以从根本上扭转这一长期违规局面。

关键词：执法与合规；环境污染；治理；犯罪与处罚；中国

## 1. Introduction

China is facing environmental crises on several fronts (Wu and Edmonds 2017). Numerous countermeasures have been actively implemented, including increasingly more stringent standards and laws, shutting down more polluting plants, and initiating more ambitious pollution-control programs. However, the overall impacts on the behavior of polluting firms may be less positive than what the regulations intend. Despite increasingly sophisticated and rigorous environmental standards and regulations, illegal pollution and other forms of violation remain rampant. The rapid growth of citizen complaints and even of protests triggered by pollution in recent years also show the ineffectiveness of China's overall environmental law enforcement (Steinhardt and Wu 2016).

The existing literature on polluting behavior at plant level mainly focuses on the economic and institutional aspects of the firms, assuming polluting firms are rational agents and they make decisions by calculating the expected costs of compliance and the expected penalties for noncompliance. If the former exceeds the latter, an individual firm will be more likely to choose noncompliance. Weak laws and/or law enforcement could result in low expected penalties and, therefore, a high probability of noncompliance by firms.

Many studies on environmental compliance and enforcement, including those concentrating on China, adopt this approach and they tend to focus on the formal rules and the mechanisms for implementing them. On the government's part, considerable academic attention has been given to explaining the "enforcement gaps"; these are mainly caused by high costs, limited budgets, shortages of personnel and necessary expertise, and a variety of institutional problems (Arguedas 2008; Blackman and Harrington 2000; McAllister et al. 2010; Pan, Wang, and Wang 2005; Russell and Vaughan 2003). On the regulatees' part, existing literature suggests that polluting firms' behavior is mainly shaped by environmental laws and policies and, more importantly, by their enforcement, as polluting firms are driven by utility maximization rationales and their managers make decisions based on comparing various costs and benefits (Xu 2011).

Another strand of scholarship has proved that neither formal nor informal enforcement mechanisms can alone ensure consistent compliance. Empirical evidence from both industrialized and developing countries suggests that effective government regulation and law enforcement and public participation are mutually supportive (Gunningham 2009; Thornton, Gunningham, and Kagan 2005).

A third strand of literature suggests that noncompliance with environmental regulations is a consequence of comprehensive factors that include not only the incentives and sanctions created by the formal regulations, but also by a variety of determinants such as the design of policy instruments, the political consensus on law enforcement intensity, the degree of heterogeneity and the capacity of regulatees, and the pressures from private intervention (Pargal et al. 1997; Weaver 2014). Specialized literature on environmental regulation and compliance in China also suggests that corporate compliance behavior can be affected by informal mechanisms such as the political connections (of the polluting firms), features of ownership, general developmental modes, and public monitoring and participation (Van Rooij 2010; Wang et al. 2003, 2008; Xie, Yuan, and Huang 2017).

Building upon the above literature, this study will explore how the formal and informal factors work together to affect the rational polluting behaviors of different firms and how various factors determine the effectiveness and efficiency of China's environmental regulatory activities. Formal factors mainly include the official regulatory system, the monitoring instruments, the administrative structures, and the alternative regulatory mechanisms voluntarily adopted by polluting firms. Informal factors refer to the political and social construction of the seriousness of compliance or noncompliance, the attitudes and beliefs of the regulatees, and the social pressures on the polluting firms to comply with the regulatory standards.

In this research, we chose to focus on one of the least environmentally friendly behaviors of polluting firms, namely, chronic noncompliance. These firms have been caught and punished numerous times for noncompliance with the environmental rules, but they still proceed without any significant changes to their polluting behavior. Guangzhou, the capital city of Guangdong province and a major site of pollution in the Pearl River Delta (PRD), is selected for an intensive case study because of data availability and representativeness in terms of pollution regulations and the polluting behavior of local firms. This paper is organized as follows. Section 2 introduces and explains the data and the analytical framework. Section 3 explains the features of chronic noncompliance with environmental regulation in Guangzhou. Section 4 examines the nature and effectiveness of the formal regulations related to pollution in Guangzhou. Section 5 examines the emerging informal approaches to monitor corporate pollution behavior in the case. Section 6 discusses the findings and concludes.

## 2. Data and Analytical Framework

### *2.1 Data*

For this research, we have compiled a comprehensive dataset on the chronic offenses of corporate polluters in Guangzhou based on official environmental, provincial, and municipal enforcement data covering the period 2007–2015 (Table 1). The official data on inspections, penalties, and citizen reporting were acquired from the Annual Report on the State of Environment in Guangzhou (RSE) and other open-source documents available on the Guangzhou Environmental Protection Bureau (EPB) website. We also used the websites of the Institute of Public and Environmental Affairs (IPE) and the Guangdong Provincial EPB for supplementary sources of data. IPE is a Beijing-based nongovernment organization (NGO) specialized in creating interactive maps/data of industrial pollution to facilitate public participation in pollution control. The IPE data focuses on large polluting sources monitored by the Ministry of Environmental Protection (MEP) and is a useful source for researchers to find information about sanctions related to the state-monitored sources (SMSs). In addition, the first author of the paper conducted six interviews with district-level EPB officers and staff of environmental NGOs (ENGOs), which provides contextual data for our interpretation and analysis.

To identity cases of chronical polluting firms, we created a set of criteria that can be easily applied with open-source data: (1) the firms that have committed three or more violations, or received three or more specific penalties (*i.e.*, fines, suspension, or shutdown); (2) the firms that are subjected to one-year interagency supervisions jointly conducted by the EPBs and Bureaus of Discipline Inspection (BODI) at the municipal or provincial level (*gua pai du ban*); and (3) the firms are included in the Environmental Noncompliance Blacklist (ENCB) disclosed by the Guangzhou EPB. The ENCB is an important supplement to the formal regulatory system focusing on large pollution sources, as it strengthens law enforcement on the small-sized polluters in scattered locations on the periphery of cities and the areas where the reach of state regulation is scarce. (More details of the ENCB and its implementation in Guangzhou will be provided in Section 4.)

Applying the criteria to our dataset, we have identified 65 out of the 250 polluting firms as chronic offenders. Among them, 37 were SMSs, mostly large state-owned enterprises (SOEs), monitored directly by the MEP. The remaining 28 cases were mainly small-sized plants subjected to prioritized monitoring by local EPBs and included in the ENCB for enhanced law enforcement. There are at least 18 blacklisted factories included in our dataset and, by the time of writing this paper, none of these polluting firms have been removed from the prioritization lists.

## 2.2. Analytical Framework

To analyze the formal and informal enforcement activities and how they might reverse chronic noncompliance, we adopt the framework based on the economic calculation of costs and benefits responding to enforcement activities, as explained in the introduction section. The benefits of noncompliance are mainly the saved costs to comply with certain environmental laws and policies, such as those for mitigating emissions. The costs of noncompliance are the expected penalty, resulting of two factors namely, the probability of catching noncompliance and the penalty for noncompliance. A key research question is how to effectively deter environmental noncompliance. With different data and methodology, other studies have concluded that detection probability is more important (Grogger 1991), or that punishment severity is more important (Friesen 2009), or that both are important (Earnhart and Friesen 2012).

Environmental noncompliance is one of the most important causes of China's current environmental crises (Xu 2011, 2013). Noncompliance cannot be deterred without a high-enough proportion of cases being caught and punished (Guo et al. 2014; Xu 2011). Catching noncompliance by firms is predominantly reliant on effective techniques of monitoring, reporting, and verification (MRV) designed, deployed, and executed by government regulatory agencies at various levels. However, the positive correlation between the deployment of MRV techniques and

**Table 1.** Data Sources

| Types of Data | Sources of Data | |
|---|---|---|
| Regular administrative penalties (2007–2015) | Annual Report on the State of Environment in Guangzhou (2008–2015) (Guangzhou EPB 2008–2016) | |
| | The Guangzhou Yearbook (2008–2015) (Editing Committee 2008–2016) | |
| | The Institute of Public & Environmental Affairs, Database on the environmental performance of enterprises (IPE 2018) | |
| Enforcement campaigns (2007–2015) | Report on the Implementation of the Listed Supervision of the Prioritized Environmental Pollution Problems (Guangzhou EPB and Guangzhou Bureau of Discipline and Inspection 2008–2016) | |
| | The Guangzhou Yearbook (2008–2015) (Editing Committee 2008– 2016) | |
| Information about citizen reporting (2007–2015) | Annual Report on the State of Environment in Guangzhou (2008–2015) (Guangzhou EPB 2008–2016) | |
| | The Guangzhou Yearbook (2008–2015) (Editing Committee 2008–2016) | |
| First-hand information from fieldwork | 1 officer from a district-level EPB in Guangzhou | Interview conducted by Lin Peng on August 14, 2015 |
| | 1 staff from an ENGO, Guangzhou Environmental Protection (GEP) | Interviews conducted by Lin Peng on April 13, 2016, May 3, 2016, and May 8, 2018 |
| | 2 staffs from an ENGO, Liu Xi He Ecological Protection Center (LAUKAI) | Interviews conducted by Lin Peng on April 13, 2016, May 7, 2016, and April 5, 2017 |
| | A public hearing hosted by the Guangzhou Municipal People's Congress | Participatory observation conducted by Lin Peng on April 12, 2018 |
| | A project evaluation session hosted by a district-level water bureau in Guangzhou | Participatory observation conducted by Lin Peng on May 9, 2018 |

the effective deterring and sanctioning of environmental violations might be hindered by inadequate regulatory capacity and other institutional factors (Lo et al. 2012; McAllister et al. 2010). The "enforcement gap" caused by weak bureaucratic capacity is particularly significant in industrializing countries like China. For instance, continuous emission monitoring systems (CEMSs) have been widely used in the United States to provide accurate data on $SO_2$

emissions and to monitor compliance (Stranlund and Chavez 2000), while the poorer quality of CEMSs in China often fails to achieve such accuracy and they are mainly used to provide guidance for occasional site inspections (Pan, Wang, and Wang 2005; Xu 2011).

Weaver (2014) has criticized the conventional scholarship on compliance and enforcement gaps that predominantly focused on the motivations of the regulatees. He then proposed a comprehensive framework to analyze noncompliance with public policies by adding factors related to the willingness and capacity of the regulatees. He also suggests that the technical and social aspects of policy instrument designs, such as the heterogeneity of regulation targets and the political constructs of seriousness of noncompliance, can have important effects on the compliant behavior of regulatees. Weaver's conceptualization of "compliance and enforcement regimes" provides a useful framework for this paper, although it falls short by focusing narrowly on formal rules made and implemented by governments and by failing to capture the widespread informal enforcement initiated by nongovernmental actors against noncompliant behavior.

This article intends to evaluate the instruments of formal and informal compliance regimes and their impact on chronic noncompliance behavior. As shown in Figure 1, the Y-axis shows compliance regimes measured in terms of formality. The most formal compliance regime mainly includes institutional and policy instruments set by the formal regulatory bodies. The less formal regime includes the instruments of voluntary regulation established by enterprises and professional civil society organizations. The least formal one refers to more spontaneous and sporadic efforts made by local communities. The X-axis measures the instruments compliance regime by the degree of enforcement coerciveness. A combination of both formal compliance regime and coercive enforcement instrument lead to highest level of compliance.

## 3. Chronic Offenders in Guangzhou

Located in the heartland of the prosperous PRD, Guangzhou is a main growth engine of China's manufacturing industries and consequently, yet it suffers from serious industrial pollution. Textile manufacturing, paper making, electronics, and metal processing are the main sources of pollution. In addition, Guangzhou has a high concentration of power stations and industrial waste treatment facilities, and these can produce high levels of pollution if their emissions are not properly treated. Chronic environmental infringers discussed in below are all from the abovementioned sectors.

Chronic environmental infringers can be further divided into two broad categories based on the variation in types and frequencies of penalties imposed by the environmental regulators (Figure 1). The first group of chronic environmental offenders are often termed "nail polluters" (*wuran*

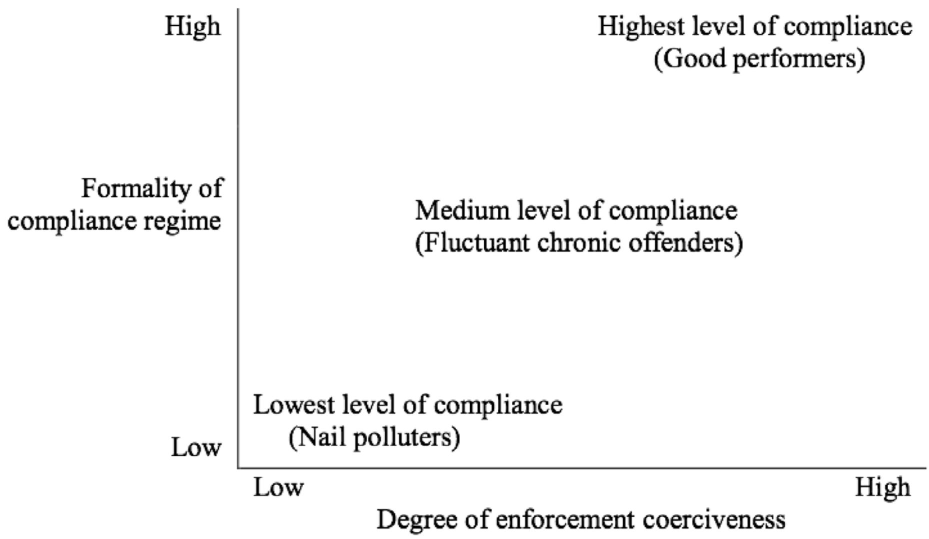



**Figure 1.** The analytical framework on the relationship between formality of compliance regime, degree of enforcement coerciveness, and compliance rates

*dingzi hu*). These polluting firms have received the highest frequency (once or more than once per year between 2007 and 2015) and the highest intensity (measured by the amount of fines and the number of coercive sanctions such as suspension, shutdown, and relocation) of penalties and yet have continued with new pollution emissions. The second type of chronic violators' noncompliant behavior is more fluctuating, with obvious ups and downs over time. In good years, polluting firms falling into this category would avoid any records of noncompliance in one year and then receive multiple warnings and penalties in the next because of their illegal emissions.

It is noteworthy that the enterprises with governmental and foreign ownerships can all be serious violators of environmental standards. Neither do SOEs and private companies demonstrate any distinctive patterns in terms of their chronic noncompliant behavior. The diversity of ownership of polluting firms not only challenges the conventional proposition that foreign companies have better environmental performance or that private companies tend to do a better job in environmental compliance than SOEs, but it also shows a high degree of heterogeneity of targets of environmental enforcement (Wang and Jin 2007).

Another important feature of chronic corporate noncompliance in Guangzhou is that a large number of the violators are small plants. Many of these small polluters are scattered throughout the city outskirts or in "urban villages"—this is a peculiar phenomenon in the PRD due to the rapid uneven urban expansion into the rural areas, and this has left patches of formerly rural villages and their land enclaved in the cities. These polluters often operate in the absence of governmental regulation and they can easily evade government inspections and sanctions by operating at night, secretly shutting off treatment facilities, concealing outlets, or simply abandoning the old factories and moving into new sites.

## 4. Formal Compliance Regime and Coercive Enforcement

China's environmental regulators mainly rely on formal administrative and legal enforcement instruments to deter and sanction those firms which breach the pollution limits. Most of the regulatory responsibilities, including daily monitoring, inspections, and administrative penalties, are decentralized downwards to the EPBs—these take charge mainly at municipality and county levels. The Guangzhou EPB is responsible for the regulation of the SMSs administered by the MEP and prioritized polluting sources at the municipal level. As grassroots level regulators, the district-level EPBs focus their law enforcement mainly on small polluters and they also assist the municipal EPB to regulate the prioritized sources.

### *4.1. Formal Monitoring and Inspection to Catch Noncompliance*

Most administrative and financial resources for monitoring pollution in China in general or in Guangzhou in particular are devoted to prioritized areas identified by governments at all administrative levels, i.e. centrally, provincially, and municipally prioritized sources. The centrally prioritized polluting sources, or the SMSs, are usually large polluters directly managed by the MEP under the Automatic Monitoring Management Program (AMMP). The Guangzhou EPB then generates a more inclusive monitoring lists by adding polluting sources prioritized at the provincial and municipal levels to the MEP SMS catalog. In practice, the local EPBs do not always differentiate between the locally prioritized sources and the SMSs. All the SMSs under the AMMP are required by the MEP to install automatic monitoring and reporting systems that can submit real-time pollution data and serve as the primary source of compliance-monitoring information for the environmental regulators. In Guangzhou, the automatic monitoring systems installed at the SMSs became fully operational in 2009, and similar devices were gradually deployed to the locally prioritized sources in the following years.

Besides formal monitoring, inspection is another important administrative enforcement instrument for China's environmental regulators. Before the deployment of the automatic monitoring systems, inspection was actually the most important method used by local EPB officers to collect information on pollution and execute sanctions. Even after the diffusion of the automatic monitoring systems, inspection is still an important instrument by which the local environmental regulators can detect and catch the small polluting firms insufficiently covered by the formal monitoring system.

Inspection mainly takes two forms, namely regular monitoring inspections and surprise field inspections or inspection sweeps. Regular inspections are conducted by the EPBs at municipal and district levels to confirm real-time monitoring information, locate key pollution sources, and spot environmental violations. The municipal EPB has been responsible for inspecting the centrally supervised SMSs and the prioritized pollution sources at the municipal level. The EPBs in various districts mainly inspect small factories within their jurisdictions.

Surprise field inspections are randomly conducted and often prompted by enforcement campaigns, environmental disasters, and citizen reporting. Inspections associated with enforcement campaigns are more likely to lead to tougher punishments such as suspension of production, temporary or even permanent closure, relocation, or detention of the owners of the polluting firms. For instance, in 2012, the municipal EPB coordinated several interagency field inspections targeting industrial pollution along the Liuxi River, an important drinking water source of the city suffering from serious environmental deterioration. Official statistics show that in 2012, 905 factories were penalized after these inspection operations, including 14 cases of permanent closures and six cases of forced relocation (Guangzhou EPB; Guangzhou BODI 2008–2016).

However, there are deep-rooted problems that have affected the effectiveness of the inspections such as weak regulatory capacity of the enforcement authorities, and political pressures from the pro-development sectors of local governments. Inspection requires substantial inputs of financial resources and manpower to ensure the effective detection of noncompliant behavior. Even in the relatively more affluent regions, like the PRD, local EPBs are still handicapped by persistent shortage of funding and personnel to carry out necessary inspections. According to the information we collected from the interviews, a typical district-level EPB in Guangzhou (with no more than 70 full-time staff) is responsible for monitoring more than 10,000 plants, most of which are small private firms. Because many small factories exhibit polluting activities irregularly or evade inspections, it is extremely challenging for the undermanned law enforcers to detect and catch these small offenders.

The second persistent challenge for effective inspections stems from the pressures from the local governments that prioritize economic development over environmental protection. According to the interviews we conducted with district-level environmental protection officers on August 14, 2015, too frequent and too strict inspections might be considered as "interfering" with the enterprises' operation and not conducive for "building business friendly environment."

### *4.2. Administrative Penalties on Noncomplying Polluting Firms*

We have identified two major types of penalties on noncompliance. The first type utilizes administrative laws and mainly targets the corporate interests of the polluting firms which would make noncompliance costly and therefore compel them toward compliance in the future. The second type activates civil criminal laws and addresses the personal interests of the decision makers or managers of the polluting firms. Criminal prosecution could possibly result in the imprisonment of the managers, and such a consequence could deter their intentions of noncompliance.

In China, environmental regulators have traditionally been more reliant on administrative penalties to deter and punish violations of environmental standards committed by the polluting firms. The forms of administrative penalties, which have degrees of coerciveness from low to high, mainly include orders requiring correction within a time limit, fines, suspension of operation, closure, relocation, and detention.

The most frequent form of administrative penalties against environmental violations is pollution fine. But, scholars have long criticized this form of penalty for being too weak to inflict sufficient financial hardship on the polluting firms and to thereby deter noncompliance with environmental regulations. China used to adopt a "per event" fine system, under which the financial penalties had a maximum limit and, according to China's Administrative Penalty Law, polluting firms were fined only once even if they violated the same environmental standards over an extended period. For both large polluters (such as SOEs) and smaller private companies, the monetary penalties caused by the traditional pollution fines were so small that it made economic sense for the polluters to continue their illegal discharges rather than to invest in pollution abatement.

Alarmed by the unacceptable environmental pollution and widespread ecological deterioration, the Chinese government began to strengthen its environmental law enforcement in the 11$^{th}$ Five-Year-Plan (2006–2010) by adopting more stringent standards and applying more intrusive policy instruments, which did lead to tougher sanctions. The official statistics of Guangzhou show that the value of pollution fines has increased significantly since 2007 (Figure 2).

More progress in China's environmental law enforcement has taken place since 2014, when the National People's Congress (NPC) introduced important amendments to the Environmental Protection Law and significantly increased the financial penalties for environmental infringements. The updated legislation, coming into force in January 2015, canceled the cap on pollution fines and allowed the environmental regulators to fine infringers on a daily basis. The modified EPL also gave enforcement officers more coercive power by allowing them to seize and confiscate production equipment and even detain the owners of the polluting firms. Although it is still too early to make any judgment about the impacts of the new environmental legislation on pollution control, the new legal measures have begun to inflict greater hardship on the polluting firms which breach the environmental standards. In 2015, the first year of the implementation of the new EPL, the Guangzhou

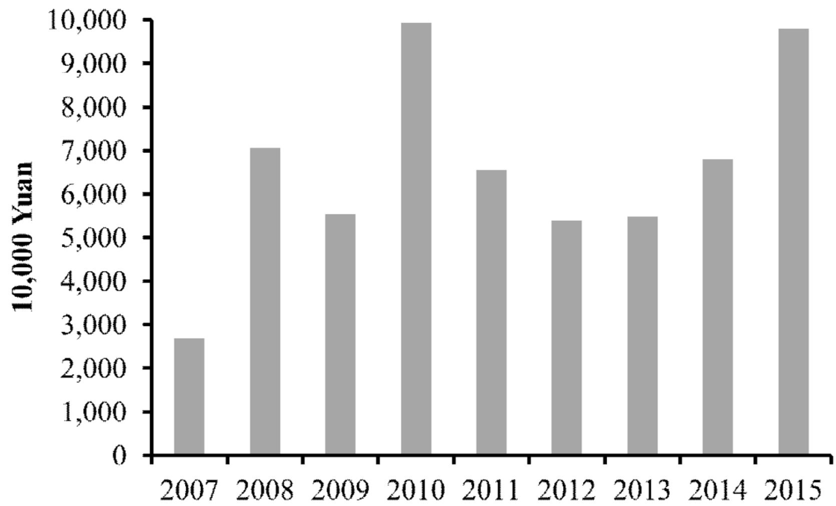



**Figure 2.** Annual total value of pollution fines (2007–2015) (Guangzhou EPB 2008–2016)

EPB levied continuing fines on four polluting firms. In 2016, the Yuehua power station, an SOE and a major electricity generator in the city, became the first large polluting firm to have continued fines imposed upon it; these fines for illegal air pollution had record-breaking value totaling 5.4 million yuan, which is the largest ever in China.

In recent years, the Chinese environmental authorities have also established a set of innovative policy instruments, such as interagency supervisions, to supplement the formal environmental laws and regulations by adding extra costs on the chronic noncompliant behavior of polluting firms. The PEPA is essentially an environmental information disclosure and rating system promoted in the early 2000s by the then Department of Science and Technology (DoST) and the State Environmental Protection Agency (SEPA) with direct assistance from an expert team from the World Bank. After several years' pivotal experiments in two cities in Zhejiang and Inner Mongolia, the project began to be implemented nationwide by the end of the 11th Five-Year-Plan (2006–2010) (Li 2012; Wang et al. 2003). In this incentive-based pollution control project, the environmental performance of firms is rated by local EPBs from best to worst by using five colors—green, blue, yellow, red, and black—and the rating results would be disclosed via mass media and the Internet. Guangzhou adopted the PEPA system in 2007 and adopted a four-color rating system (green, blue, yellow, and red). Polluting firms coded in red would be subjected to more intense supervisions and sanctions jointly conducted by the local EPBs and BODIs and barred from deposit-refunds, performance bonds, and various green loans. As a complement to the PEPA system mainly covering the big polluting sources, the Guangzhou EPB also established a municipal environmental performance disclosure system called the ENCB. Initiated in 2014, the ENCB specifically targets polluting firms which commit continued violations, and it focuses more on the small polluters which are difficult to detect and monitor by the formal regulatory system.

The interagency environmental supervision, conducted by the EPBs in conjunction with the BODIs, was initiated in 2006 as an instrument of political implementation aimed at adding political and economic costs on chronic noncompliance. The Rules for the Listed Supervision of the Cases of Environmental Infringements enacted by the MEP in 2009 specifically stated that the polluting firms listed on supervision by the environmental protection agencies and departments of discipline include polluters "failed to stop offences despite repeated investigations and penalties." The involvement of the BODIs in environmental law enforcement is a method for not only enhancing the enforcement of environmental standards and rules, but it also enables the authorities to punish the owners of the polluting firms who serve in public office or who are members of the local People's Congress or Political Consultative Conference, specifically addressing the barrier for effective environmental regulation created by political connections and protectionism. The implementation of interagency environmental supervisions has been associated with public reporting, severe and continued infringements disclosed by the PEPA, and pollution control priorities set by local EPBs. Supervised polluting firms would usually be given six months to reverse their noncompliant behaviors by, for instance, reducing emissions to the levels required by the pollution control laws or investing sufficient money in improving their abatement facilities. If the supervised polluters failed to reduce illegal emissions within the given time limit and initiate new offenses, they could be forced to shut down or relocate.

These additional policy instruments have led to the growing number and severity of penalties against environmental violations in Guangzhou. For instance, the interagency environmental supervisions have resulted in a greater number of permanent shutdowns and relocations of polluting plants in Guangzhou (Figure 3). However, most of the harsh penalties were imposed on small factories without significant economic and social impacts. Large polluters, such as the SOEs and big foreign companies, targeted by the supplementary enforcement mechanisms, have rarely been forced to shut down or relocate even when their infringements have been more serious than those of the small polluters.

### *4.3. Criminal Prosecutions Against Decision Makers in Polluting Firms*

Effective legal measures are largely absent in China to criminalize individuals who are found to be responsible for serious harm both to personal interests and to public resources. Prior to 1997, China's pollution victims could initiate legal actions against polluters only for personal economic compensation and

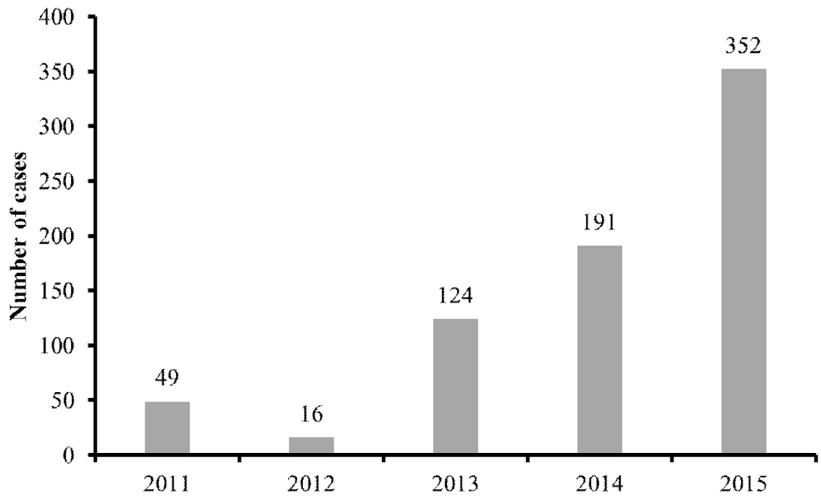



**Figure 3.** Cases of closure and relocation associated with interagency environmental enforcement in Guangzhou (Guangzhou EPB 2008–2016)

they can be easily frustrated by legal processes which are not only time-consuming and expensive but also technically challenging in the presentation of evidence. The 1997 Criminal Law for the first time included articles of "crime of major environmental pollution accidents," but it has not become an enforceable instrument by which to punish the decision makers of the polluting firms mainly due to the lack of accurate definition of crimes, limited coverage of behaviors, and a high burden of proof for the plaintiffs. Even when the environmental offenders were sued, China's legal authorities, such as the courts and procuratorates, did not have sufficient professionally trained personnel and the will to deal with environmental cases. The absence of enforceable laws and the weak capacity and will of the legal authorities have led to a very low number of criminal prosecutions against the owners of polluting firms. In Guangzhou, there was (before 2013) only one criminal prosecution relating to the serious illegal emission of pollutants; the person responsible for the infringement was sentenced to three years in prison.

Major breakthroughs in China's environmental criminal enforcement came from a special judicial interpretation made by the Supreme People's Court (SPC) in 2013 that specified 14 circumstances that should considered as "serious environment pollution" and five criteria for conviction without requiring evidences of specific harm or injuries caused by the violation, respectively. These new legal measures have granted new powers to environmental law enforcers to deter and sanction environmental infringements by specifically targeting the decision makers of the polluting firms.

In Guangzhou, the first conviction for environmental pollution crime was made about four months after the SPC's new decision. By the end of 2013, 11 cases of environmental violations had been subject to criminal prosecution. In 2014, the cases handed over by the environmental authorities to the local police departments had doubled. However, the number of criminal sentences had not increased substantially (Table 2).

**Table 2.** Cases of Criminal Prosecution in Guangzhou (Editing Committee 2008–2016; Guangzhou EPB 2008–2016)

| | 2013 | 2014 | 2015 |
|---|---|---|---|
| Cases handed over to police | 11 | 26 | 9 |
| Number of convictions | 3 | 2 | 2 |

So far, most of the prosecutions and sentences were associated with violations committed by small private enterprises and no SOEs and their managers were sued for serious illegal pollution. But, it deserves particular notice that two staff members, including a Korean citizen, from a Korean company, were jailed in 2015 because of the illegal discharge of a large amount of contaminants into a local river. This was not only the first criminal penalty against a foreign corporation, but it was also first criminal prosecution against foreign infringers in Guangzhou. Given the fact that enterprises with foreign ownerships have been enjoying favors from the local governments, the criminal penalties on the Korean company signaled the strength of the new legislation and the increasing determination of the local enforcement agencies.

However, the improvement of environmental legislation and the implementation of the newly established legal measures do not mean the disappearance of the "enforcement gap"—this is characterized by the disparity between the increased law enforcement tasks and the insufficient regulatory resources and capacity. Although the new laws enable the authorities to sue and criminalize individuals responsible for illegal pollution emissions, the environmental authorities' unwillingness to transfer cases to the legal authorities, the weak coordination between the environmental and legal authorities, and the lack of professional training and qualified staffs are important barriers to the effective enforcement of the new legislation.

To overcome such barriers for criminal prosecution against environmental offenders, in January 2014, the MEP and Ministry of Public Security issued a specific circular calling for the enhanced coordination between the environmental authorities and the public security forces in environmental law enforcement. Soon after the statement made by the central government, the

Guangzhou EPB established a specific coordinative mechanism with the local public security departments. The municipal Public Security Bureau also established a special division specializing in investigating environmental crimes. Despite all these efforts made by the local authorities, it still takes time to effectively implement these new mechanisms for environmental criminal enforcement. Our fieldwork found that the police forces were still poorly trained in the investigation of environmental cases and they were therefore unwilling to receive cases transferred from the environmental protection agencies.

## 5. Informal Compliance Regime and Noncoercive Citizen Enforcement

The informal compliance regime is characterized by active intervention in unregulated polluting behavior and enforcement of environmental standards by private citizens. Informal regulation takes many forms. In the context with well-established regulatory environment, citizen enforcement mainly takes the forms of citizen-initiated law suits and reporting. In developing countries, where environmental regulation is generally weak, typical citizen environmental enforcement usually happens outside of the institutional framework, such as public campaigns, resistance and boycotts led by NGOs and community leaders.

In China, citizen enforcement is profoundly structured and influenced by the authoritarian state, and public participation in environmental law enforcement mainly takes on cooperative instead of confrontational approaches. The past decade has witnessed the growing impact of citizen enforcement on corporate environmental noncompliance (Johnson et al. 2018). As a response to increasing environmental pollution and inadequate regulatory resources, the Chinese government has, over the past decade, been trying to encourage public participation in reporting environmental offences and monitoring polluting plants. A variety of laws, institutions, and polices have been established by governments at various levels to enable and encourage ordinary citizens to monitor and report illegal discharges from polluting firms. On the other hand, the growth of the public's environmental awareness and the continued development of ENGOs during the past decade have also given rise to more vigorous citizen enforcement activities and they have also provided supplementary instruments to the formal environmental regulatory system.

### *5.1. Informal Reporting and Monitoring by Citizens*

The major forms of informal enforcement of environmental regulation in China include citizen reporting and monitoring. Ordinary Chinese citizens can report illegal discharges to local EPBs through a variety of channels, such as telephone hotlines, official websites and the microblogs of EPBs, and conventional letter-and-visit systems. More importantly, the formal law enforcement activities conducted by the environmental regulators have become increasingly interconnected with public reporting. To be more specific, citizen reporting has become an important cause for environmental inspections and sanctions. For instance, an important criterion for the environmental performance rating under the PEPA system is the frequency of public complaints. Only those firms receiving fewer than three public complaints can be coded "green," the best score in the rating system. Many chronic polluters targeted by the interagency supervisions and ENCBs were also those polluters who were frequently reported by the local residents.

However, the effectiveness of citizen monitoring and reporting in China has been constrained by the general weakness of bureaucratic capacity of the local environmental authorities. The official statistics show that more than 90% of public complaints have been "properly handled" (*tuo shan chu li*) over the past five years (Figure 4). However, according to our interview on April 13, 2016, less than 70% of citizen reports of environmental offenses could be processed by the EPBs at the district level, these bodies being seriously undermanned and also preoccupied with other regulatory priorities. Furthermore, the official management of citizen reporting lacks transparency, and this makes it difficult for the complainants to verify the law enforcement activities and their real impacts on the reported polluting firms.

In the dataset compiled for this research, we find only a few chronic polluters with clear records of citizen reporting and thereafter connect these reports of offenses with specific environmental sanctions. For instance, Meiye Textile, a Hong Kong-based textile company and an important supplier to many brand-name multinational corporations, was reported by an anonymous citizen on the website of the Guangzhou EPB for polluting the air of the local community in January 2013 (Guangzhou EPB 2013). Two months later, the Guangzhou EPB publicized a very specific response to the netizen who earlier complained through the same platform on its website. In this online statement, the Guangzhou EPB declared that the inspectors dispatched to the scene had not detected illegal emissions carried out by the reported enterprise. However, the enforcement agency also stated that the reported plant would be relocated as planned in June 2013 by the municipal government in a massive industrial relocation project (Guangzhou EPB 2013).

Another rare case that shows direct linkage between citizen reporting and specific law enforcement actions against illegal polluting behavior was a bloc prosecution of polluting firms surrounding several high and elementary schools in the District of Baiyun, in Guangzhou, a suburban area with a high density of small enterprises that was particularly weak in environmental legal enforcement. At least eight polluting factories were stormed and clamped down by the local EPB in an enforcement campaign as a result of four years of petitioning by teachers and concerned parents in that area, who might have been affected by the pollution

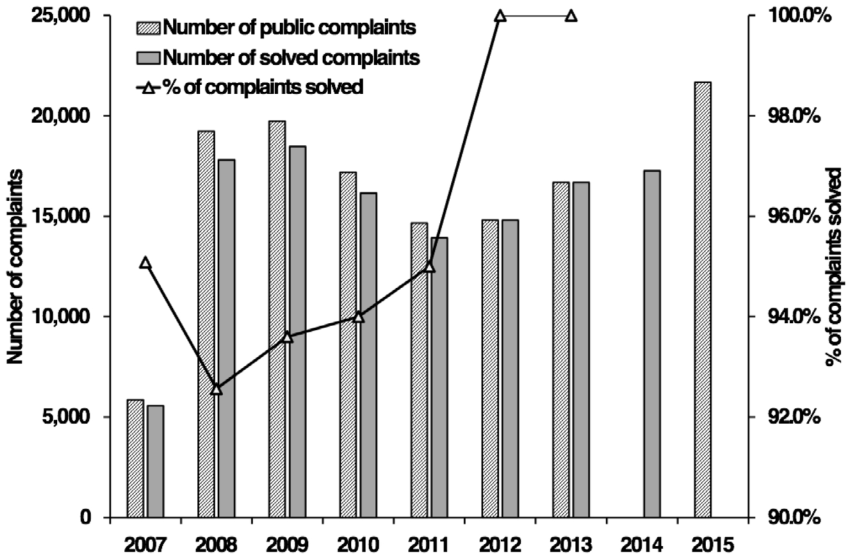



**Figure 4.** The number of public complaints on pollution in Guangzhou and the proportion of resolved cases (2007–2015) (Guangzhou EPB 2008–2016) (Data for 2014 and 2015 are not completely available)

(Wang 2014). Actually, the local EPB involved in the case did respond to the citizens' reports of air pollution by fining and suspending production of some of the polluters. However, these sanctions were too weak to deter the environmental offenses and the continuing illegal air pollution was causing numerous respiratory symptoms among the students and teachers near these polluting plants. It was not until the frustrated petitioners brought the cases directly to the Ministry Education (and not the MEP) and attracted media attention in 2014, that tougher enforcement actions were implemented.

Compared with citizen reporting, which is usually individual-based, post facto and lack of systemic data, the monitoring of industrial pollution conducted by NGOs is relatively more organized and preemptive and has more impacts on EPB actions and policy change. A representative example of such social initiatives in environmental enforcement is a nationwide pollution monitoring campaign led by IPE, a Beijing-based ENGO focusing on production chain environmental performance supervision. IPE uses the monitoring data of the SMSs disclosed by the EPBs at various levels to create Green Audit reports of the suppliers of multinational corporations in China and it has developed interactive web maps to encourage ordinary citizens to monitor pollution sources and report illegal discharges. Due to its success in facilitating public participation in environmental law enforcement by applying information and communication technologies (ICTs), IPE was in 2015 invited by the environmental authorities to participate in mobilizing citizen monitoring over water pollution in urban areas (Wu 2017). IPE coordinated dozens of grassroots ENGOs across the country to join this campaign and form a network to facilitate the exchange of information and collaborative action.

In Guangzhou, some local NGOs also joined this network and became active participants in pollution control campaigns and private enforcement activities focusing on local environmental violations. One of the active Guangzhou-based ENGOs participating in the private monitoring of industrial pollution is Guangzhou Environmental Protection (GEP). Stemming from a volunteer group organized by several ordinary citizens concerned about the environmental deterioration in their home city, GEP has turned into a professional Green NGO specifically focusing on detecting and reporting illegal water pollution over the past few years. Besides carrying out independent investigations itself and also reporting on illegal pollutant emissions, GEP also relies on mobilizing local communities to detect and report illegal polluting sources. GEP's independent monitoring and mobilization of community participation have led to a growing number of detections of illegal emissions made by small polluting sources that are poorly covered by the formal regulatory system. To further enhance its capacity to deter environmental violations, GEP has been consciously trying to nurture and maintain cooperative relationships with the local environmental authorities and it has established regular coordinative mechanisms with the municipal and provincial EPBs and Water Service Bureaus (WSBs). The GEP's proactive engagement with the government has been rewarded because the environmental authorities are becoming more responsive to citizen reporting and supportive of public participation in environmental law enforcement. For instance, in 2016, GEP and its community partners reported 37 cases of illegal emissions to the local environmental authorities and most of them were investigated. In the same year, GEP, along with several other grassroots and official social organizations, was invited by the municipal government to train citizens who had volunteered to serve as "citizen river chiefs," an innovative strategy designed to mobilize public participation in a top-down enforcement campaign aiming to curb urban water pollution. The rise of grassroots NGOs like GEP improved the capacity of mobilizing public participation and establishing partnership among potential pollution victims, NGOs and governmental regulators.

### *5.2. Civil Lawsuits Led by NGOs*

NGO-led environmental public interest litigations have proven to be effective instruments to deter corporate environmental offenses and to enforce environmental standards in the industrialized countries. In China, however, "citizen suits" against environmental violations had been lacking because the laws did not provide articles for public participation in environmental litigation and for prosecution for infringing the public interests associated with the illegal polluting behavior of firms.

The newly-modified Environmental Protection Law passed in 2014 marked a watershed moment for China's private environmental enforcement by allowing NGOs to bring public interest environmental lawsuits. Although the new legislation still sets strict legal and even political limitations on the participation of nongovernmental prosecutors, such as registration at municipal administration and no records of any administrative and legal violations in three years, some professional ENGOs that had been advocating for public interest litigation immediately seized the newly-acquired opportunity to file lawsuits against the enterprises responsible for pollution incidents that severely harmed the local environment and communities. In 2015, among the 48 public interest cases in China, 41 of them were initiated by ENGOs and the government-sponsored Environmental Protection Federations (EPFs) (Li 2016). In Guangzhou, the first public interest case took place soon after the implementation of the updated EPL and the litigant was the All-China Federation of Environmental Protection (ACFEP).

Although the revised EPL did enable the NGOs to enforce environmental laws and deter noncompliant behavior of polluting firms with public interest litigation, effective implementation of the new legislation still needs to address numerous barriers such as, to name a few, the lack of professional knowledge and resources on the part of nongovernmental litigants and the weak capacity and will of the legal authorities to catch the infringers. The limitation of the newly-empowered litigation was clearly shown in the first public interest case in Guangzhou. Although the infringers and illegal emissions were local, the plaintiff was the ACFEP, based in Beijing, and therefore, neither the local public procurator nor the municipal FEP were technically and politically prepared to file the environmental public interest litigation. More importantly, as an actor from the political capital of China, the involvement of ACFEP in this case showed the MEP's determination to set a precedent in the regulation of illegal industrial waste dumping and strengthen the law enforcement in this field. But as with enforcing the law on those small polluters operating on the periphery of the formal regulatory system, the local courts found it difficult to enforce the verdicts because the polluters responsible for the illegal dumping simply disappeared.

## 6. Conclusion and Discussion

Chronic noncompliance by polluting firms is one of the most serious challenges for China's policymakers and law enforcement agencies in the face of environmental degradation and widespread pollution in China. With rich empirical firm-level data derived from Guangzhou, we have examined both formal and informal environmental enforcement actions and their effects on corporate environmental behavior.

We found that combined efforts by both state agencies and civil society actors, represented by environmental NGOs, have not been able to fundamentally discourage or reverse some firms' repeated decisions to illegally pollute the environment. For large polluting firms (mostly SOEs, as well as some privately owned ones), although they are more visible and thus receive more monitoring and inspections by the formal law enforcement agencies, insignificant penalties, relative to their compliance costs or economic scale, makes it more attractive for them to evade the regulations. As for small polluting firms, although the penalties for them are significantly heavy relative to their sizes, the probability of being caught in their noncompliance remains low, given the inadequate bureaucratic capacity of the frontline environmental authorities for effective enforcement. Thus, many small firms would simply take risks in conducting polluting activities because, even when they are caught, it is not difficult or financially devastating to simply abandon and close down the business to evade punishment. New administrative and economic instruments have been introduced such as the repeated fine system and criminal prosecution against firm managers, but the impacts of these measures are yet to be clearly seen and be visible to all.

However, the general weakness of compliance and enforcement regimes in China's pollution control does not disguise the complexity of compliant behavior of firms (or lack of it) and the practices of environmental regulation in China. The experience of Guangzhou shows that barriers to the effective performance of compliance and enforcement regimes are complicated and that they vary across different types of firms and violations.

For large polluters, like the SOEs and some high-profile privately-owned companies, the main barriers for environmental compliance include not only weak sanctions due to the problematic environmental legislation and policies, but also factors such as the political connections of the polluting firms or their owners and the local government's fear of increasing unemployment and the loss of revenue caused by the shutdown or relocation of big companies.

For small polluters, a persistent barrier to ensuring long-term compliance has been the passiveness of the frontline enforcement agencies. In practice, the catching and sanctioning of small polluters are not cost-efficient for the frontline law enforcers because of their continuing shortage of staff and lack of financial resources and the absence of reliable monitoring information about these small polluting sources. However, recent development and combination of both citizen and official enforcement mechanisms have improved the probability and efficiency of catching and punishing the small polluters. Statistics from both official and NGOs' sources have shown a growing number of detections and shutdowns of small factories which have repeatedly not complied with the environmental laws.

The effectiveness of compliance and enforcement regimes also varies with the types of contaminants discharged by the polluting firms. As shown by the experience of Guangzhou, illegal water pollution and the discharge of untreated industrial solid waste were more easily caught and harsh penalties thereafter imposed. The reporting of illegal air pollution, on the other hand, was relatively more difficult to penalize. The main reason for such differences might be explained by the low visibility of air pollution and the difficulty of detecting and verifying it.

Our findings have confirmed the arguments in the regulatory literature derived from the experiences of both industrialized and industrializing countries, namely, that the sole reliance on either the formal government enforcement or the civilian enforcement cannot effectively ensure corporate compliance with the environmental regulations. Only with an effective combination of formal and informal enforcement mechanisms can any environmental compliance regime ensure the compliant behavior of firms. As shown by the case of Guangzhou, the central government's increased emphasis on pollution control and the continued strengthening of environmental legislation have led to enhanced law enforcement and tougher penalties against illegal polluters at the local level. From 2010, more polluting firms were subjected to political implementation of environmental rules jointly conducted by the EPBs and BODIs and forced to permanently shut down. The past few years have also witnessed the development of public participation in environmental regulation and this has led to the increased detection of environmental violations and more intrusive law enforcement activities launched by the government.

However, the positive correlation between enhanced law enforcement and the decrease in chronic noncompliance with environmental rules was more likely to be observed among small polluters. The sheer scale and the elusive nature of the environmental violations by these polluters used to create major obstacles for effective law enforcement. Illegal emissions committed by small factories were extremely difficult to catch and punish by government enforcement agencies which were seriously undermanned and more preoccupied by the regulation of the larger pollution sources. For large polluting firms (mostly SOEs, as well as some privately owned ones), although they are more visible and thus subjected to more frequent and intense monitoring and inspections conducted by the government law enforcement agencies, insignificant penalties, relative to their compliance costs or economic scale, make it more attractive for them to evade the various laws. As for small polluting firms, although the penalties for them are significant relative to their size and scale, the probability of being caught in noncompliance remains low mainly due to the shortage of manpower and resources of the grassroots regulatory forces, and many would simply choose to take the risk; and, even when they are caught, it is also neither difficult nor economically devastating to simply abandon the business and evade the punishment.

The opening-up and expansion of political space for more involvement by public monitoring and NGO-led litigation is expected to make noncompliance by polluting firms more costly. Through the experience of Guangzhou, where civil society has developed to a higher degree than in the rest of the country, it has been shown that there remain several obstacles preventing social pressure becoming effective in reversing the polluting firms' behaviors. The number of institutions and agencies involved in public monitoring and reporting are increasing, and they are mostly handled by the relevant EPBs. However, this rarely results in serious penalties. Environmental litigation led by NGOs has also just emerged and, if successfully carried out, can lead to significant penalties and consequences for the polluting firms. Nevertheless, the experiences of NGO-led public interest litigation against environmental violators in a Guangzhou lawsuit against illegal industrial pollution suggest that only a handful of experienced and well-funded NGOs are capable of environment litigation. Complex political relations can also impede the normal functioning of the legal process in an environmental pollution case.

Another important political factor for an increasing number of relocations of penalized polluting firms is the grand industrial restructuring being promoted by the local government. This project is aiming to attract high-end service industries and push out the polluting industries and it reflects a long-term policy being pursued by the Guangzhou municipal government. This project has been backed by sustained and strong political will from the municipal government and it serves as an important driving force behind many environmental law enforcement campaigns that have led to permanent shutdowns and relocations of polluting factories.

Other than policy design, firm-specific, and other political reasons, there are other factors which are not directly related to environmental law enforcement but indirectly sustain the chronic noncompliance decisions at the firm level, such as market fluctuation and competition as well as the changing costs of compliance. For polluting firms, the resources for compliance consist nearly entirely of economic costs without generating revenues and profits. In the economic hardship that emerged after the financial crisis, those firms that are fighting for survival may especially give pollution mitigation a lower priority. Fierce market competition could also dilute their profit margin. Firms that can successfully evade compliance could get favorable standings which are largely dependent on pricing. In addition, compliance costs are also evolving, but generally in a decreasing trend due to learning and innovation.

As the Chinese central government is increasingly keen and vocal about reducing industrial pollution, and Chinese society is more aware of the health impacts of pollution, policy and institutional design for effective environmental law enforcement are the common pursuit and goal for all stakeholders. However, establishing such effective enforcement institutions and procedures will take time due to the diverse nature of firms, the uneven development of local EPB capacity, and the still-nascent stage of environmental NGOs and public participation in environmental governance in China.

## Acknowledgment

Funding for this research was provided by the Hong Kong Research Grants Council through the General Research Fund (14654016).

## References

Arguedas, C. 2008. "To Comply or Not to Comply? Pollution Standard Setting Under Costly Monitoring and Sanctioning." *Environmental & Resource Economics* 41 (2): 155-68.

Blackman, A., and W. Harrington. 2000. "The Use of Economic Incentives in Developing Countries: Lessons from International Experience with Industrial Air Pollution." *The Journal of Environment Development* 9 (1): 5-44.

Earnhart, D., and L. Friesen. 2012. *Environmental Management Responses to Punishment: Specific Deterrence and Certainty versus Severity of Punishment*. Working Paper.

Editing Committee. 2008–2016. "*Guangzhou Yearbook*." Guangzhou, China: Guangzhou Yearbook Press.

Friesen, L. 2009. *Certainty of Punishment versus Severity of Punishment: An Experimental Investigation*. Working Paper.

Grogger, J. 1991. "Certainty vs Severity of Punishment." *Economic Inquiry* 29 (2): 297-309.

Guangzhou EPB. 2008–2016. "Annual Report on the State of the Environment in Guangzhou (2007–2015)". Retrieved May 14, 2018. http://www.gzepb.gov.cn/zwgk/hjgb/.

Guangzhou EPB. 2013. "Complaint about Frequent Emission of Black Smokes from the Chimneys of Guangzhou Meiye's Plant." Retrieved May 14, 2018. http://www.gzepb.gov.cn/gzhd/zxrdhb/201303/t20130301_63828.htm.

Guangzhou EPB and Guangzhou Bureau of Discipline and Inspection. 2008–2016. "Report on the Implementation of the Listed Supervision of the Prioritized Environmental Pollution Problems (2007–2015)". Retrieved May 14, 2018. http://www.gzepb.gov.cn/root43/gov/146/1114/1121/1123/list.htm.

Gunningham, N. 2009. "Environmental Law, Regulation and Governance: Shifting Architectures." *Journal of Environmental Law* 21 (2): 179-212.

Guo, M.Y., Y. Xu, and Y.Q.D. Chen. 2014. "Fracking and Pollution: Can China Rescue Its Environment in Time?" *Environmental Science & Technology* 48 (2): 891-92.

IPE. 2018. "Database on the Environmental Performance of Enterprises". Retrieved May 14, 2018. http://www.ipe.org.cn/IndustryRecord/Regulatory_DD.aspx.

Johnson, T., A. Lora-Wainwright, & J.X. Lu. 2018. The quest for environmental justice in China: citizen participation and the rural-urban network against Panguanying's waste incinerator. *Sustainability Science*, 13(3), 733-746.

Li, D. 2016. *Review of Public Interest Litigation in Environment Protection*. China Law Press. Beijing, China.

Li, W.X. 2012. "Advocating Environmental Interests in China." *Administration and Society* 44 (6): 26-42.

Lo, C.W.H., G.E. Fryxell, B. van Rooij, W. Wang, and P.H. Li. 2012. "Explaining the Enforcement Gap in China: Local Government Support and Interna Agency Obstacles as Predictors of Enforcement Actions in Guangzhou." *Journal of Environmental Management* 111: 227-35.

McAllister, L.K., B. van Rooij, and R.A. Kagan. 2010. "Reorienting Regulation: Pollution Enforcement in Industrializing Countries." *Law & Policy* 32 (1): 1-13.

Pan, L., Z. Wang, and Z. Wang. 2005. "Present Status and Countermeasure Suggestion for Thermal Power Plants CEMS in China." *Research of Environmental Sciences* 18 (4): 42-45.

Pargal, S., H. Hettige, M. Singh, and D. Wheeler. 1997. "Formal and Informal Regulation of Industrial Pollution: Comparative Evidence from Indonesia and the United States." *The World Bank Economic Review* 11 (3): 433-50.

Russell, C.S., and W.J. Vaughan. 2003. "The Choice of Pollution Control Policy Instruments in Developing Countries: Arguments, Evidence and Suggestions." In Folmer, H., Tietenberg, T. (Eds.), *International Yearbook of Environmental and Resource Economics*. Cheltenham, UK: Edward Elgar. VII.

Stranlund, J.K., and C.A. Chavez. 2000. "Effective enforcement of a transferable emissions permit system with a self-reporting requirement." *Journal of Regulatory Economics* 18 (2): 113-31.

Steinhardt, H.C., and F.S. Wu, 2016. "In the Name of the Public: Environmental Protest and the Changing Landscape of Popular Contention in China". *China Journal* 75, 61-82.

Thornton, D., N.A. Gunningham, and R.A. Kagan. 2005. "General Deterrence and Corporate Environmental Behavior." *Law and Policy* 27 (2): 262-88.

Van Rooij, B. 2010. "The People vs. Pollution: Understanding Citizen Action against Pollution in China." *Journal of Contemporary China* 19 (63): 55-77.

Wang, J. 2014. "Introduction of Sanctions of Polluting Factories around Schools in the District of Baiyun." *Yangcheng Evening News*, December 31.

Wang, H., and Y.H. Jin, 2007. "Industrial ownership and environmental performance: Evidence from China". *Environmental & Resource Economics* 36, 255-273.

Wang, H., N. Mamingi, B. Laplante, and S. Dasgupta. 2003. "Incomplete Enforcement of Pollution Regulation: Bargaining Power of Chinese Factories." *Environmental & Resource Economics* 24 (3): 245-62.

Wang, M., M. Webber, B. Finlayson, and J. Barnett. 2008. "Rural Industries and Water Pollution in China." *Journal of Environmental Management* 86 (4): 648-59.

Weaver, R.K. 2014. "Compliance Regimes and Barriers to Behavioral Change." *Governance—an International Journal of Policy Administration and Institutions* 27 (2): 243-65.

Wu, F. 2017. "China's Effective Greens and Gaps in Climate Activism," *Diplomat Magazine*, Issue 34.

Wu, F. and R. Edmonds. 2017. "Environmental degradation in China", in Czes Tubilewicz (ed) *Critical Issues in Contemporary China: Unity, Stability and Development* (London and New York: Routledge), pp. 105-119.

Xie, R.H., Y.J. Yuan, and J.J. Huang. 2017. "Different Types of Environmental Regulations and Heterogeneous Influence on "Green" Productivity: Evidence from China." *Ecological Economics* 132: 104-12.

Xu, Y. 2011. Improvements in the Operation of $SO_2$ "Scrubbers in China's Coal Power Plants." *Environmental Science & Technology* 45 (2): 380-85.

Xu, Y. 2013. "Comparative Advantage Strategy for Rapid Pollution Mitigation in China." *Environmental Science & Technology* 47 (17): 9596-603.

This publication is available open access at:
http://www.ipsonet.org/publications/open-access

Thanks to the generosity of the American Public University System

APU American Public University

AMU American Military University