WAYS TO ACHIEVE GREEN ASIA

Edited by Bihong Huang and Eden Yu

ASIAN DEVELOPMENT BANK INSTITUTE
Ways to Achieve Green Asia

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Bihong Huang and Eden Yu
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Abbreviations

AR5  fifth assessment report (IPCC)
ARDL  autoregressive distributed lag
ASI  Annual Survey of Industries
BCA  border carbon adjustment
BIC  Bayesian information criterion
BOD  biochemical oxygen demand
CAC  command and control
CDM  Clean Development Mechanism
CER  certified emissions reduction
CGE  computational general equilibrium
CIPS  cross-sectionally augmented IPS
CO₂  carbon dioxide
COP  Conference of the Parties
Cup-FM  continuously updated fully modified
CVM  contingent valuation method
DDI  distressed and dirty industry
ECO  Ecosystem Vitality Index
EKC  environmental Kuznets curve
EPI  Environmental Performance Index
ETS  emissions trading scheme/system
EU  European Union
EU ETS  European Union Emissions Trading System
FDI  foreign direct investment
GCM  global climate model
GDP  gross domestic product
GEF  Global Environment Facility
GHG  greenhouse gas
GMM  generalized method of moments
HLT  Environmental Health Index
IAM  integrated assessment model
INDC  Intended Nationally Determined Contribution
IPCC  Intergovernmental Panel on Climate Change
IWG  Interagency Working Group
JVETS  Japanese Voluntary Emissions Trading Scheme
Lao PDR  Lao People’s Democratic Republic
NAFTA  North American Free Trade Agreement
NAS  National Academies of Sciences, Engineering, and Medicine
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<td>NGO</td>
<td>nongovernment organization</td>
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<td>NIC</td>
<td>National Industrial Classification</td>
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<td>NO₂</td>
<td>nitrogen dioxide</td>
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<td>OECD</td>
<td>Organisation for Economic Co-operation and Development</td>
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<tr>
<td>PRC</td>
<td>People’s Republic of China</td>
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<td>PRR</td>
<td>pollution reduction by rationalization</td>
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<td>SCC</td>
<td>social cost of carbon</td>
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<tr>
<td>SO₂</td>
<td>sulfur dioxide</td>
</tr>
<tr>
<td>tCO₂e</td>
<td>tons of CO₂ equivalent</td>
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<td>TCZ</td>
<td>two control zones</td>
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<td>UN</td>
<td>United Nations</td>
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<td>UNEP</td>
<td>United Nations Environment Programme</td>
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<td>US</td>
<td>United States</td>
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<td>VoSL</td>
<td>value of statistical life</td>
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<td>WDI</td>
<td>World Development Indicators</td>
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<td>WG1</td>
<td>Working Group I</td>
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<td>WHO</td>
<td>World Health Organization</td>
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<td>WTA</td>
<td>willingness to accept</td>
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<td>WTO</td>
<td>World Trade Organization</td>
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<td>WTP</td>
<td>willingness to pay</td>
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Escalating environmental degradation is attracting growing attention from both policy makers and the public. For Asian countries, decades of remarkable economic growth have borne mixed fruit in terms of environmental implications. Visible environmental challenges ranging from air quality deterioration to global climate change undermine people's standard of living and impose irreversible damage on the ecosystem. This book is a collection of 11 articles, some of which were presented at the Asian Development Bank Institute–World Economy Workshop on Globalization and Environment at the Asian Development Bank Institute in Tokyo, Japan on 26–27 September 2017. Globalization has been a defining trend since the 1970s that heralds a new era of interactions among nations, economies, and people through economic, political, information technology, and cultural integration. Globalization impacts the environment via economic development in a wide variety of ways and through various channels.

Globalization in terms of liberalization of trade and investment has contributed significantly to the world economy in expanding world production in both goods and services. The resulting increase in the gross domestic product (GDP) of many countries has prompted an increased demand for energy and resources. Japan, for example, moved from poverty after the Second World War in the late 1940s and 1950s to become a high-income, industrialized nation in the 1990s. Japanese demand for aluminum has gone up by 400%, for energy by 500%, and for steel by 2,500% over 5 decades. The surge in demand for energy and resources from recent emerging economies such as the People’s Republic of China (PRC) and India appears to be even more dramatic. As the population of these two nations combined is about 25 times more than that of Japan—and these two nations are among the fastest-growing regions in the world—they will likely be tremendous growth in the demand for energy and resources in the future to support these countries’ development.

The United States Energy Information Administration projects that global energy consumption will increase by 54% from 2001 to 2025. With rapid economic growth in Asia, demand for energy consumption in the region will double, accounting for about 40% of the projected increase in global energy consumption. This demand-driven depletion
of resources can disturb the ecological balance and inflict damage on the environment. In addition, the accompanying globalization can affect the environment through the process of production. A significant portion of the manufacturing sector (such as chemicals, metal products, and petroleum) creates pollution in nearby emerging economies in Asia. Recent trade frictions and protectionism can also hurt the environment through the increased production of capital-intensive importable goods, especially in developing countries.

A common observation in recent years is the globalization of production activities, foreign investment, international trade, and consumption. Globalization can be attributed to shortened travel time between various regions and the removal of barriers to commodity and factor movements directly resulting from reduced transportation costs, greater speed of telecommunications, and trade liberalization. By promoting an even greater degree of market integration, globalization induces capital flows from high-production-cost and environmentally sensitive regions into low-production-cost and environmentally lax countries. This has engendered the concept of local and global governance of multinational firms. Apparently, intricate relationships between trade liberalization, globalization, and the environment exist.

The 11 chapters, divided into three major parts, together aim at exploring the various aspects of environment and climate change in Asia. The first part overviews the environmental performance in Asia and assesses the economic impacts of climate change in the region. The second part offers an in-depth discussion of environmental regulations, environmental governance, environmental evaluation, and growth of carbon markets in Asia. The third part explores the various aspects of the relationships between globalization and the environment, including case studies in the PRC and India. The highlights of the chapters are presented below.

Chapter 1 gives an overview of the environmental challenges facing Asia and proposes policy recommendations for more effective environmental governance. For countries in Asia, decades of rapid economic growth have led to visible environmental degradation, ranging from poor air quality to global climate change that undermines quality of life. The challenges, which impose irreversible damage on the ecosystem, have gained increasing attention by policy makers and the public.

Chapter 2 summarizes the climate change impacts on Asia and overviews the methodologies to measure the economic cost of climate change. It also discusses the mitigation challenge and concludes with country-level policy options going forward.
Environmental research in recent decades has generated important lessons for policy makers to draw on to formulate more effective regulation of environmental externalities. Chapter 3 synthesizes the theoretical and empirical literature on environmental regulations and highlights important insights on formal regulation such as taxes, standards, and tradable permits, as well as on informal regulation such as information and voluntary approaches pointing to the direction of future research.

Chapter 4 offers empirical evidence showing that government expenditures of Asian nations on environmental protection as a share of GDP significantly impact environmental conditions. At the same time, the government should strike a balance between economic growth and environmental protection. All these steps directly or indirectly reduce carbon dioxide emissions and promote energy efficiency for a better environment.

Chapter 5 provides details on the carbon market situation across major Asian economies with specific attention to the PRC carbon market. It is proposed that governments discourage the use of fossil fuel energy by imposing higher taxes in addition to the effective implementation of the emissions trading system. At the same time, the government should supply the requisite renewable energy and provide tax incentives for individuals, organizations, and firms to adopt.

To determine the efficient allocation of limited resources among competing uses by policy makers, there is a need to evaluate society’s degree of preference for the environment vis-à-vis other goods and services. Chapter 6 overviews economic valuation methods including both demand and non-demand curve approaches, such as the dose–response method, contingent valuation method, and hedonic pricing for environmental goods, along with examples of real-world applications. The chapter further discusses the damage schedules approach and benefit transfer in cases where conventional valuation methods are less suitable.

Chapter 7 reviews recent works on trade and environment policies. Although the aggregate effects of trade on the environment are not large (except via the effects of trade on growth), there are important regional and sectoral variations where effects have been significant, and there are trade-induced firm-level responses that are not yet well understood. Evidence suggests that environmental policy reduces the competitiveness of polluting industries.

Chapter 8 explores the welfare consequences of international outsourcing in a three-stage model of North—South trade. It provides a framework for analyzing the effects of international outsourcing on the
environment under three scenarios of no, partial, or full accountability for outsourcing-induced environmental damages. To resolve the environmental problem fully, strengthening regulation or fostering international cooperation is desirable for implementation until the environmental costs of outsourcing are fully accounted for by the outsourcing firms in the North. These, however, may react by resorting to insourcing, diversified outsourcing, and other strategies.

Chapter 9 examines the impact of economic growth and trade openness on the environment in PRC cities, using the continuously updated fully modified method that allows for cross-sectional dependence and endogeneity. It is found that the environmental Kuznets curve hypothesis holds not only for the whole but also for different regions of the PRC.

Chapter 10 investigates the impact of international trade and foreign direct investment on climate change with special reference to India’s economy. Preliminary calculations using data from the World Bank show that GDP has a direct, proportional relationship with the extent of carbon dioxide emissions in India, and the relationship is even stronger after the introduction of the liberalization policy in the 1990s. However, trade seems to have an inversely proportional relationship, consistent with the view that Indian imports are mostly manufactured items that may involve polluting production processes and are currently being produced outside India.

The final chapter examines the impact of tariff liberalization and foreign direct investment on pollution in India from a plant-level perspective by analyzing the differential impact of input and output tariff liberalization in the country. However, this chapter finds that input tariff liberalization induces plants to invest more in pollution abatement equipment. Further, it finds that an increase in competition through output tariff liberalization does not lead to a decline in spending on pollution control equipment to cut costs, as some theories might suggest.
PART I
Overview of the Environment and Climate Change in Asia
Environmental issues are gaining increasing attention from both policy makers and the public across countries in Asia. Visible environmental challenges ranging from air quality deterioration to global climate change undermine people’s standard of living and impose irreversible damage on the ecosystem. For countries in Asia, decades of rapid economic growth have borne mixed fruit in terms of environmental implications. On the one hand, the accumulation of national wealth through unchecked industrialization and natural resource exploitation has compromised ecosystem vitality and brought about health hazards such as air and water pollution. On the other hand, Asia’s early-starter countries in economic growth, such as Japan and the Republic of Korea, are devoting more resources to environmental governance with greater regulatory efficiency, hence boosting environmental performance. This twofold effect highlights both the wide regional variances among different states in Asia and the complex nature of drivers behind environmental performance, as growth could either deteriorate or remediate the environment. Along this vein, this chapter intends to provide an overview of environmental performance in Asia, disentangle its drivers, and finally advance preliminary policy recommendations for more effective environmental governance in the region.

1.1 Environmental Performance in Asia

This section examines the overall state of environmental performance in Asia using the Environmental Performance Index (EPI). The EPI was developed jointly by Yale University and Columbia University in collaboration with the World Economic Forum to provide a relatively comprehensive measurement of national environmental performance.
Given the wide array of issue areas covered by this indicator, the EPI serves well the purpose of this chapter to develop a high-level overview of countries’ performances. Specifically, the EPI examines 10 issue areas categorized under the two sub-indicators of environmental health and ecosystem vitality, which respectively indicate the risks imposed on human health by environmental pollution and the impacts levied on the ecosystem through environmental degradation. In the 2018 EPI framework, for instance, the Environmental Health Index (HLT) measures air quality, water quality, and heavy metal exposure, while the Ecosystem Vitality Index (ECO) covers issue areas such as biodiversity and climate change. The weighted average of these two sub-indicators gives rise to the EPI as an overall assessment of a country’s environmental status (Wendling et al. 2018).

Figure 1.1 presents the 2018 regional average EPI and sub-indices. Asia shows a rather unsatisfactory performance—the second-lowest EPI score among all regions. While slightly higher than the level of sub-Saharan Africa, Asia’s EPI assessed at 50 is noticeably lower than the world average EPI of 56. Poor performance in environmental health is
a major reason behind Asia’s laggard status. The HLT level of Asia is assessed to be 49, which is 21% lower than the world average level of 62. In comparison, Europe and North America, where many developed nations are located, has achieved an HLT of 93, nearly double that of Asia. Poor performance in environmental health indicates the greater risks that the population in Asia is exposed to on average in terms of polluted air, water, and excessive heavy metal exposure. On the other hand, Asian countries received a relatively better assessment on ecosystem vitality, as the regional average ECO score of 51 is only slightly lower than the world average level of 53. Despite the much smaller gap with leading countries, Asia is still ranked the third-lowest region in terms of ecosystem vitality, which indicates considerable room for continuing regional sustainability endeavors.

Despite Asia’s poor environmental performance on average, country-level indices reveal strong regional variances. Indeed, the spread in rankings among Asian countries is larger than for any other region in the world (Wendling et al. 2018). Table 1.1 shows the 2018 environmental performance index and sub-indices for 24 Asian countries, along with the rankings and 10-year change in their performance. Japan and Singapore, as regional leaders, entered the top 50 with their rankings at 20th and 49th, respectively. In contrast, out of the five bottom-ranking countries in the world, three are from South Asia: Bangladesh at 179th, India at 177th, and Nepal at 176th. Besides absolute ranking, countries’ trends of change also illustrate a broad distribution. Nine out of the 24 Asian countries have experienced a decline in environmental performance over the last decade, including both the best performing states of Singapore and Japan and some of the bottom-ranking countries such as Nepal and Cambodia. In contrast, Uzbekistan, Viet Nam, and the Lao People’s Democratic Republic (Lao PDR) are among the economies that have significantly improved their environmental performance. The broad spectrum of rankings among Asian economies and their uneven progress over the last decade suggest the need to tailor the environmental policy solutions of different states with careful considerations of country specificities.

Breaking down the EPI into the HLT and ECO provides more information on countries’ status regarding various environmental issue categories, which drive their overall performance. Low performance in environmental health is a major contributor to the bottom ranking of laggard countries, most notably in South Asia. The HLT index for Bangladesh, India, and Nepal were all assessed to be around 10, which is over 80% lower than the world average at 62. The EPI report highlights air quality as a particularly problematic issue area in several Asian countries with a low HLT performance (Wendling et al. 2018).
### Table 1.1: Environmental Performance Index and Sub-Indices by Country in Asia

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<th>2018</th>
<th>2018 ranking</th>
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Sources: Yale University, Center for Environmental Law & Policy; Columbia University, Center for International Earth Science Information Network.
In India, exposure to air pollution caused about 1.24 million deaths in 2017, which accounted for 12.5% of the total deaths (Balakrishnan et al. 2019). In this sense, deterioration of air quality is triggering a severe public health crisis that demands urgent actions. Nevertheless, performance in environmental health has been steadily improving across Asia over the last 10 years, except for several countries that experienced mild decline. Nepal, India, and Bangladesh all demonstrate a leap forward of over 5% in environmental health performance compared to a decade ago. If the trend continues, the future holds a more positive outlook for healthy and sustainable living across Asia.

In contrast to the steadily improving environmental health conditions, more than half of the Asian countries in Table 1.1 have seen their ECO decline or remain stagnant. Rapid industrialization and economic growth in Asia over the past decades could be a major reason behind this trend. Increasing production activities, resource extraction, transportation, and consumption all invariably impose burdens on the ecosystem and reduce its vitality. To illustrate this phenomenon better, this section takes a closer look at the profile of carbon dioxide (CO₂) emissions of countries in Asia (Table 1.2). Atmospheric CO₂ is the largest contributor to human-induced climate change (Canadell et al. 2007) and is also the most heavily weighted component in calculating ECO. The contributing effect of economic growth to CO₂ emissions has been heavily studied and proved by scholars (Narayan and Narayan 2010; Canadell et al. 2007; Niu et al. 2011). Consistent with the trend of declining ecosystem vitality, CO₂ emissions per capita have been constantly rising for most Asian countries over the last 40 years (Table 1.2). Central Asia as an exception experienced a sharp decline starting from the 1990s due to economic contraction after the collapse of the Soviet Union (Karakaya and Ozcag 2005). Despite the overall trend toward heavier CO₂ emissions, the annualized growth rate of emissions has been declining for many Asian countries, especially between the 2000s and the 2010s. The slowdown and even negative growth show the increasing effectiveness of emissions governance. As many developed states in Europe have witnessed a steady decrease in CO₂ emissions per capita since the 1980s,¹ several Asian states could soon join their ranks if the trend of declining growth continues. To realize this future prospect, sophisticated policy design and efficient environmental governance are imperatives, which will be further explored in the rest of this chapter.

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¹ Source: Emissions Database for Global Atmospheric Research (EDGAR).
### Table 1.2: Carbon Dioxide Emissions per Capita in Asia, by Economy

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Source: Emissions Database for Global Atmospheric Research.
1.2 Drivers of Environmental Performance

The impact of human activities on the environment could advance in various directions. On the one hand, excessive and irresponsible industrial production results in pollution, natural resource depletion, and ecosystem deterioration. On the other hand, effective environmental governance and eco-friendly technologies alleviate the burden on the ecosystem, reduce environmental risks to human health, and promote sustainable growth. The driving factors that affect environmental performance can be divided into two broad categories. The first set encompasses socioeconomic factors including national economic achievement and the advancement in production technologies. The second category is concerned with regulatory effectiveness, relevant to issues such as governments’ fiscal commitment to environmental governance and the stringency of environmental regulations. This section examines the relationship between these factors and environmental performance, with statistics that profile Asian countries along with these drivers.

To start with, whether rising income level and economic growth improve or deteriorate the environmental performance of a country has been a topic of controversy. The two components of the EPI, indeed, demonstrate contrary responses to economic achievements. Environmental health is found to be positively associated with economic growth and prosperity, while ecosystem vitality comes under strain from industrialization and urbanization (Wendling et al. 2018; Gallego-Alvarez et al. 2014). Besides the diverging effects on various dimensions of environmental performance, different stages of economic development are also found to sway a country’s environmental status in contrasting directions. In fact, this phenomenon is well established as the environmental Kuznets curve and has been empirically proven by several scholars (Grossman and Krueger 1991; Selden and Song 1994; Dinda 2004). The theory essentially posits an inverted-U relationship between pollution and economic development: pollution grows rapidly in the early stage of industrialization when clean air and water are not priorities compared to jobs and growth. Then, as an economy becomes wealthier and the demand for environmental quality rises, pollution gradually falls to the pre-industrial level.

For Asia specifically, gross domestic product (GDP) growth is often found to be accompanied by the deterioration of environmental quality (Jalal and Rogers 2002), which results in Asian countries’ positioning in the left half of the environmental Kuznets curve (Chang et al. 2019). In other words, Asian countries are currently at an earlier development stage where resources, attention, and regulatory capacity devoted to
environmental governance are insufficient. Despite the lack of consensus on the exact way through which higher income affects environmental performance, eco-responsible growth and effective green governance remain imperatives for sustainable development. As countries in Asia continue to pursue growth, finding the right balance between growth and sustainability through effective policies and institutions will be an important theme in policy making in the coming decades.

One promising solution to tackle the environmental crisis without compromising economic growth is to advance green technology and production efficiency. As another driver of environmental performance, boosting productivity through eco-innovation is considered a win–win solution that reduces the environmental burden but increases GDP at the same time (Janicke 2012). As an example, Hou et al. (2018) found that the industrial green transformation of the People’s Republic of China (PRC) from 2010 to 2015 significantly contributed to a reduction in carbon emissions intensity, especially in the context of weak environmental regulation. Similarly, evidence from the United States manufacturing sector suggests that the adoption of clean production, as an eco-innovation in industrial processes, is associated with better compliance with environmental management standards and complements pollution reduction (King and Lenox 2001). In this sense, green technologies that improve production eco-efficiency amplify the positive effect of environmental governance to make green production a viable and attractive option for the industry.

Energy intensity can be used as a rough indicator of production eco-efficiency. While by itself a crucial dimension of environmental performance in terms of resource conservation, lower energy intensity is empirically found to correlate with lower CO₂ emissions (Anshasy and Katsaiti 2016; Sinha 2016). Therefore, improvement in production eco-efficiency potentially leads to multiple positive outcomes, including energy savings and emissions reduction. Figure 1.2 presents the energy intensity performance of 22 Asian countries from 1990 to 2014. Nearly all countries have realized a gradual decline in energy use per unit of production, despite some fluctuations, indicating a steady increase in energy efficiency across Asia.

Besides economic growth and the advancement of production technology, public sector governance is another crucial contributing factor to environmental performance. Given the nature of environmental protection as a public good with shared benefits and minimum exclusion, theories of public finance suggest that reliance on free market forces would result in inefficient and insufficient actions to combat environmental challenges (Zhou 2004; Pearce and Palmer 2001). Therefore, the state plays a critical role in driving and regulating
Figure 1.2: Energy Use (Kilograms of Oil Equivalent) per $1,000 Gross Domestic Product (Constant 2011 PPP)

(a) Central Asia

(b) East Asia

continued on next page
Figure 1.2  continued

PPP = purchasing power parity.
environmental protection endeavors. Several scholars have found that stronger public sector commitment, measured by government expenditures on environmental protection, is positively correlated with better environmental performance (Ercolano and Romano 2018; Halkos and Paizanos 2013). Chang, Dong, and Liu (2019) showed that a higher ratio of government expenditures on environmental protection to GDP significantly contributes to a reduction of CO₂ emissions and the promotion of energy efficiency. Notably, this effect is more significant among countries in Asia than in Europe, potentially given the stronger reliance on central government regulations in Asian countries and the lack of hybrid partnerships with non-state stakeholders on environmental actions. At a country level, Huang (2018) examined a sample of 30 provinces in the PRC between 2008 and 2013 and found that higher government spending on environmental protection effectively reduces sulfur dioxide emissions.

In addition to government fiscal commitment, the quality of environmental governance constitutes another dimension of its effectiveness. Regulatory stringency, policy mechanisms, the volume of environmental legislation, and several other indicators reflect environmental governance quality. To start with, empirical evidence suggests that the policy-stringency effect is a significant source of CO₂ emissions reduction, based on either data from the countries of the Organisation for Economic Co-operation and Development (Hille and Shahbaz 2019) or the country-specific case of the PRC (Ahmed and Ahmed 2018). While little doubt can be raised about the positive environmental effect of stringent policy, controversies are more often concerned with the economic costs of environmental regulation compliance, which could potentially sacrifice competitiveness. However, a strand of the literature has shown a positive overall effect on competitive performance, starting with Porter and Linde (1995) who pointed out that environmental regulation could stimulate innovation and raise resource productivity. Kozluk and Timiliotis (2016) also found that more stringent domestic environmental policies have no negative effect on overall trade but could enhance a country’s comparative advantage in “cleaner” industries. Besides regulatory stringency, carefully designed policy mechanisms could also boost the effectiveness of environmental governance. Evidence from Malaysia suggests that market-based policies such as carbon taxes are more effective in CO₂ abatement than sectoral emission standards. Production of renewable energy also increases strongly under a carbon tax policy, while no substantial negative effects on the Malaysian economy were observed as a result of emission regulations (Yahoo and Othman 2017).
Given the significant linkages between environmental governance and green performance, it is helpful to examine the regulatory commitment and quality of Asian countries. Figure 1.3 presents the central government expenditures on environmental protection as a percentage of total government outlays for nine countries. From 2010 to 2016, four out of nine countries experienced a rise in the share of environmental expenditures, with Nepal, Japan, and the PRC showing relatively high commitment. The strong fluctuation among countries and over the years, however, suggests the absence of a consistent trend toward greater government attention to environmental protection across Asia. Figure 1.4 shows the accumulative number of climate change laws and legislation by region in Asia, as a proxy for regulatory stringency. Asia is experiencing an increasing volume of environmental laws with an accelerating pace of growth. While East Asia had been the major contributor until 2006, Southeast Asia overtook it as the region with the largest number of climate change regulations launched each year. The growth and shift over the years signify the gradual establishment and maturation of environmental regulatory frameworks in Asia.
1.3 Policy Recommendations

For policy makers in Asia, environmental governance is embedded with challenges in the age of economic takeoff. The potential trade-off between growth and green stewardship as well as the economic burden of stringent regulations have been sources of resistance against devoted environmental efforts. At the same time, a substantial number of studies have been undertaken to show the positive economic impacts of effective environmental governance in terms of encouraging innovation, increasing productivity, and enhancing competitiveness, as examined in the last section. This research, on the one hand, highlights the benefits of consistent regulatory commitment to environmental protection. On the other hand, it underscores the fact that economically and socially effective environmental governance is contingent upon well-designed policies and efficient regulatory frameworks. Therefore, countries in Asia would benefit from a scientific approach to policy design that improves
the quality of environmental governance, in addition to the continuous commitment of resources. Based on the overview of Asia’s environmental performance and its drivers, several principles can be advanced for more effective environmental governance in Asia, as follows:

(i) Market-oriented regulations to incentivize stakeholders. Environmental regulation throughout the world is experiencing a transition from the conventional “command-and-control” regulation, such as emission standards, to the market-oriented approach, which includes a series of measures such as eco-taxes and subsidies, voluntary agreements, eco-labeling, and emission trading (Lemos and Agrawal 2006; Chang et al. 2019). Market-oriented policies have the benefits of greater flexibility and lower compliance costs. They also incentivize stakeholders to raise productivity and undertake innovations. Given that many developing states in Asia lack sufficient government capacity to enforce regulation, the market-oriented approach could establish a self-regulating mechanism, as the industry starts to reap the benefits of environmental protection efforts.

(ii) Technology-driven production upgrade. Most countries in Asia are at a development stage where economic growth often compromises environmental quality (Jalal and Rogers 2002). Eco-innovation in production technologies can be a key solution to help countries transcend the trap of growth-driven environmental degradation by achieving higher productivity and eco-efficiency at the same time. To realize technological advancement in this regard, policy makers need to signal and engage research at universities, research institutions, and corporations, and to encourage applications in both public and private sectors.

(iii) Hybrid partnership with non-state actors. The complexity of environmental issues is increasingly demanding society-wide participation from various non-state actors such as business institutions, markets, the public, and nongovernment organizations. Beyond the regulatory capacity of the state, self-regulation and mutual supervision are helpful to balance the interests of various stakeholders with the overall environmental protection agenda. Scholars also point out that non-state actors are no longer merely the subjects of environmental regulation but play an increasingly crucial role in policy-making processes (Bulkeley and Mol 2003; Armitage et al. 2012). In this sense, hybrid partnerships in Asia could become a new source of sustainable growth that harnesses society-wide actions.
References


2

The Economic Impacts of Climate Change: Implications for Asian Economies

Maximilian Auffhammer

2.1 Introduction

Climate change has been widely touted as the biggest environmental challenge humankind will encounter over the next centuries. The Intergovernmental Panel on Climate Change (IPCC) projects that, in the absence of significant policy intervention, unmitigated climate change may lead to unprecedented changes in the climate system—including, but not limited to, changes in temperature, precipitation, and sea levels (IPCC 2013). These changes in the climate system will significantly impact human society. If we are serious about engaging in mitigation actions, costs will be incurred in the near future to achieve emissions reductions. These will bring about hard-to-measure benefits in terms of avoiding climate change further in the future. As climate change is a global phenomenon, not a single economy or group of economies can solve this problem. It will take efforts by economies large and small to tackle this problem. Furthermore, the impacts are not evenly distributed. The IPCC (2014a) points out that poorer populations, especially in agrarian societies, are likely to suffer the most. In terms of sectors, agriculture and energy use are two that are highly exposed. Direct and indirect effects of climate change have also been shown to affect labor productivity, crime, violent conflicts, happiness, migration, mortality, and morbidity (Carleton and Hsiang 2016). However, our understanding of the impacts for most of these sectors is limited to specific locations and time periods. Also, solid methods to quantify impacts are still being developed.

This chapter proceeds as follows. Section 2 summarizes the impacts of climate change on Asia based on the IPCC’s fifth assessment report
Section 3 discusses the methodologies used by economists to quantify damages from climate change and overviews two recent papers that provide some empirical estimates of the impact of climate change on the gross domestic product (GDP) of individual countries, as well as country-level estimates of the implied social cost of carbon (SCC). Section 4 discusses historical and future emission scenarios more broadly and provides some insights into the contribution to emissions by different countries. Section 5 briefly discusses global efforts to arrive at a global climate policy regime. Section 6 discusses the country-level policy options going forward and concludes.

2.2 Physical Impacts of Climate Change

Joseph Fourier, a French mathematician, hypothesized the greenhouse effect in 1924. In 1896, Svante Arrhenius was the first scientist to make a quantitative prediction of global warming if one doubled the atmospheric concentration of carbon dioxide (CO\textsubscript{2}). Over the past 125 years, scientists have studied the impact of higher greenhouse gas (GHG) emissions on the global climate.

Every 7 years, the IPCC issues assessment reports that are meant to synthesize the current state of climate science. Working Group I (WG1) “aims at assessing the physical scientific basis of the climate system and climate change. Its main topics include changes in GHGs and aerosols in the atmosphere; observed changes in air, land, and ocean temperatures, rainfall, glaciers and ice sheets, oceans and sea level; historical and paleoclimatic perspective on climate change; biogeochemistry, carbon cycle, gases and aerosols; satellite data and other data; climate models; climate projections, causes and attribution of climate change” (IPCC 2013). The report includes summary predictions for the entire globe and provides regional projections of the impacts of climate change. For a more comprehensive and recent review, I suggest consulting Hsiang and Kopp (2018) or the WG1 report (IPCC 2013). Working Group II (WG2) “assesses the vulnerability of socioeconomic and natural systems to climate change, negative and positive consequences of climate change and options for adapting to it.” In short, therefore, WG1 deals with the physical climate system and modeling, while WG2 summarizes the literature on impacts on human and natural systems (IPCC 2014a).

The main conclusions drawn by the IPCC for their AR5 are paraphrased as follows (IPCC 2014a):

(i) Warming trends and increasing temperature extremes were observed across most of the Asian region over the 20th century. Observations show a rising number of warm days and a decreasing number of cold days. This trend is not
slowing down. Precipitation trends have changing degrees of variability across Asia.

(ii) Due to increasing water demand and suboptimal management practices, water scarcity is thought to be a major challenge going forward. Water availability is of massive importance due to the large population in the region. While future projections at the subregional level do not provide a clear forecast, rapid population growth and rising incomes can put additional pressures on water resources in the region. Management of water resources is paramount.

(iii) Climate change will affect food production, yet the impacts will be variable depending on the location. However, it looks like many regions will experience a climate-driven decline in production, the clearest predictions being for reduced rice production due to the shortening of growing periods. It is noted that CO₂ fertilization may offset some of these drops in yield. Overall, there may be some winners—Kazakhstan’s cereal production, for example—and some losers (Western Turkmenistan and Uzbekistan). It is further noted that the area dedicated to high-yielding wheat could decrease significantly in the Indo-Gangetic Plain. Rising seas will cause problems for low-lying areas.

(iv) There has been an observed shift in phenology, along with the growth and distribution of plant species and permafrost degradation. Climate change is expected to amplify these impacts going forward.

(v) There is evidence that marine and coastal systems are experiencing increased stress from climate and other factors. Rising sea levels are expected to lead to increased coastal erosion and high water levels. There may be damage to mangroves, salt marshes, and coral reefs.

(vi) Climate change is expected to compound impacts caused by urbanization and industrial and broader economic development. “Climate change is expected to adversely affect the sustainable development capabilities of most Asian developing countries by aggravating pressures on natural resources and the environment. Development of sustainable cities in Asia with fewer fossil fuel–driven vehicles and with more trees and greenery would have a number of co-benefits, including improved public health” (IPCC 2014a).

(vii) Human health, security, livelihoods, and poverty will be increasingly affected by extreme climate events. Mortality and morbidity, especially for vulnerable groups, are expected to rise due to heatwaves. The risk of diarrheal diseases,
dengue fever, and malaria is expected via the increased risk of floods and droughts.

(viii) The AR5 concludes that “studies of observed climate changes and their impacts are still inadequate for many areas, particularly in North, Central, and West Asia” (IPCC 2014a). The call is for better projections of all significant precipitation, which affects water supplies. The assessment report further highlights a greatly limited understanding of the impacts of climate change on many sectors but singles out the urban environment.

Table 2.1 outlines the IPCC’s summary of the state of scientific evidence in physical impact estimation. The crosses in the table suggest that, for the Asian region, evidence is insufficient and there are critical knowledge gaps. This table can serve as a map for a future research agenda, as understanding the historical and future impacts of climate change on these sectors is key for optimal policy making.

The IPCC synthesizes physical and socioeconomic impacts published in the literature. The literature attempting to estimate damage from climate change has exploded, even since the publication of the AR5. To fill the gap for some of the sectors pointed out by the IPCC as lacking evidence, economists and statisticians have developed several methods to estimate empirically the projected costs of climate change. The next section discusses the evolution of these impact estimation techniques and how they are used in practice.

<table>
<thead>
<tr>
<th>Topics/Issues</th>
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<th>East Asia</th>
<th>Southeast Asia</th>
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</thead>
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<td>Water supply</td>
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<td>Terrestrial and inland water systems</td>
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Table 2.1: State of Evidence Regarding Observed and Projected Impacts of Climate Change

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### Table 2.1 continued

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<tr>
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Key: / = Relatively abundant/sufficient information; knowledge gaps need to be addressed but conclusions can be drawn based on existing information.  
$\times$ = Limited information/no data; critical knowledge gaps, difficult to draw conclusions.  
NR = Not relevant.  

### 2.3 Economic Impacts of Climate Change

Climate scientists have spent billions of dollars studying the physical impacts of climate change. Much of this work involves data collection on, below, and above ground, which is costly. Furthermore, global climate models (GCMs) take up significant amounts of supercomputer time, which is expensive. Governments and funding agencies across the globe have not hesitated to fund this work, and the body of literature and our understanding of the climate system reflect the magnitude of this investment. From a social scientist’s perspective, one has to ask the question of how historical and future changes in the climate system will translate into impacts on human (e.g., urban) and human–natural (e.g., agricultural) systems. The literature on physical impacts is more extensive than the literature on the economic impacts of climate change. There are two major interconnected ways by which economists and modelers have tried to estimate the economic impacts of climate change. The first modeling pathway is via the so-called integrated assessment models (IAMs), which are frequently used to estimate the SCC, which
is the marginal damage of emitting 1 ton of CO₂ equivalent at a given point. The second approach uses econometric methods and variation in weather and climate to estimate damages from climate change. I discuss these modeling approaches and how they are interconnected below.

### 2.4 Integrated Assessment Models and the Social Cost of Carbon

The IAMs are, as the name would suggest, models that integrate projections of emissions pathways and a simple climate model with a damage function: this translates changes in, for example, surface temperature, precipitation, and sea level into economic damage. Figure 2.1 helps us conceptualize our thinking.

![Figure 2.1: Causal Chain in Integrated Assessment Models](source)

Due to the long lifetime of the major greenhouse gas (CO₂), IAMs usually go beyond this century and model damages over a few centuries. The most common models used go up to the year 2300. This poses a significant challenge, of course. The first step in an IAM is to project socioeconomic scenarios—most importantly, future income and population levels over the next 300 years. Imagine being King George III in 1738 and having to project income and population levels for 2019. Even had he been armed with a computer, this would have posed a stiff challenge. Most models adopt simplistic projections of these two variables, some suggesting continued increases in per capita incomes and a leveling of the population. One could, of course, adopt more advanced approaches, such as those proposed by Müller and Watson (2016). The socioeconomic pathways are then translated into emissions of CO₂ equivalent over the
time horizon. Some models also include other GHGs with shorter (e.g., methane) and longer (e.g., sulfur hexafluoride) lifetimes. The constructed emissions pathways are then fed into a simplistic climate model that translates global emissions pathways into changes in temperature, precipitation, and sea level, depending on the model. Some models are global—hence, the output from the climate model is a single time series for the entire planet; others have regional resolution for large aggregates (e.g., Europe, Asia). The regional models project the climate outcomes for each region. At this point, we are in the same position in the causal chain that the much more complex GCMs from the end of the previous section, although at a much more simplistic level in terms of the climate outputs they provide and the detail of modeling.

The IAMs’ point of departure is that they take the output from their climate module and feed it into a damage function. A damage function maps the levels of relevant climate variables (e.g., temperature) into outcomes of economic interest. These include, but are not limited to, productivity, agricultural production, human mortality, infrastructure impacts, energy consumption, and disease vector spread.

Furthermore, human society is extremely multidimensional and many products and services we consume directly or indirectly rely on factors that are not valued in markets. For example, the benefits of watershed protection provided by forests is extremely difficult to monetize. Another example is the existence of animal species whose existence humans value (e.g., tigers, bald eagles) yet which are not traded in markets. One way to think about this is that certain animals are traded for human consumption (e.g., chickens) and, hence, there are ways to monetize their market value. Since we do not consume tigers or eagles, it is extremely difficult to monetize damage to these species, so what is needed are damage functions for the most important “sectors” of the economy that map climate into welfare outcomes. I will describe in more detail below the cutting-edge methods that are currently being developed. However, it is fair to say that the current IAMs use damage functions that were developed in the 1990s and early 2000s, and are terribly out of date. To summarize, in theory, IAMs go end to end from a time series of emissions scenarios to a time series of values for outcomes of interest.

How is this related to climate change? Conceptually, this is pretty straightforward. What one could do is feed a time series of emissions scenarios that are consistent with a no or low climate change scenario, calculate a stream of damages, and compare this to the damages generated by feeding the model an emissions path consistent with increased emissions of GHGs. We could then simply calculate the difference between the two paths. If the costs from climate change (e.g., higher mortality) are bigger than the benefits (e.g., higher yields of crops in high latitude regions), we refer to this as damage. If changes in
emissions are small, assuming that there are no feedback effects on, for example, production is reasonable. However, if changes in emissions are large causing significant increases in, say, temperature, then there could be feedback effects through the direct effects of a change in climate on productivity. Some IAMs can represent this feedback loop and some do not have it built in. A few IAMs can be run as optimization problems and are well suited to these types of large impact simulations.

The most policy-relevant use of IAMs is arguably their use in estimating the SCC. The SCC is the present value of the damage caused by a ton of CO$_2$ emitted at a given time. To relate this to what is discussed above, what is done in practice is that modelers run an IAM with a baseline path of socioeconomics and the resulting emissions, and then add a “pulse” of CO$_2$ emissions at a given point. That pulse is supposed to mimic a 1-ton increase in emissions. One then calculates the stream of damages until 2300 by taking the difference between the two–time series of damages. This results in a stream of damages to the year 2300. As any economist will note, future consumption is valued differently from present consumption; hence, this time the path of damages must be discounted to make damages 300 years in the future comparable to damages today. When discounting, the most important choice a modeler must make is what discount rate to use. Policy applications have used 2.5%, 5%, and 7%. The higher the discount rate, the less weight is placed on future damages.

Calculation of the SCC in the past was a mostly academic exercise. Various teams worked on different models to calculate this global number. This is an important point: since GHGs are global pollutants, the origin of emissions does not matter. Furthermore, the right number from a social welfare point of view is the global SCC—much like in *The Lord of the Rings*, it is the “number to rule them all.” There have been some efforts, which I will discuss below, to calculate domestic numbers, even though from a global policy perspective the right number is the global one. From an academic perspective, it does not really matter who or which country calculates this number.

The Obama administration commissioned a set of federal agencies to put together a working group charged with calculating an official SCC to be used in federal rule making in the United States (US). This was the biggest effort put together by any government in history. This Interagency Working Group (IWG) involved representatives from the Council of Economic Advisers, the Council on Environmental Quality, the Department of Agriculture, the Department of Commerce, the Department of Energy, the Department of Transportation, the Environmental Protection Agency, the National Economic Council, the Office of Energy and Climate Change, the Office of Management and Budget, the Office of Science and Technology Policy, and the Department of the Treasury. In short, all agencies with any relation to
environment, climate change, energy, and cost–benefit analysis in the federal government were at the table. The IWG chose three IAMs to produce an SCC estimate: Dynamic Integrated Climate-Economy Model (DICE);\(^1\) Framework for Uncertainty, Negotiation and Distribution (FUND);\(^2\) and Policy Analysis of the Greenhouse Effect (PAGE).\(^3\) DICE is a global IAM, while FUND and PAGE both have regional resolution. The IWG aimed at feeding the three models an identical set of inputs (e.g., socioeconomics) using an identical set of discount rates, while at the same time characterizing uncertainty over the SCC. Figure 2.2 plots

**Figure 2.2: Interagency Working Group Estimates of the Social Cost of Carbon**


Notes: The figure combines the 50,000 2020 3\% discount rate estimates from each of the three models of the Government of the United States to illustrate their influence on the aggregate histogram that determines the official government SCC for 2020 at 3\%—the average ($42) and 95th percentile ($123).

Source: Rose, Diaz, and Blanford (2017).

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1. DICE can be found at https://sites.google.com/site/williamdnordhaus/dice-rice.
2. FUND can be found at http://www.fund-model.org.
3. PAGE is not open source. Some references can be found at https://www.climatecolab.org/wiki/PAGE.
the SCC estimates for the year 2020 across the three models, using a discount rate of 3%.

The figure displays the distribution of the 50,000 model runs for each IAM using quasi-identical assumptions about socioeconomic pathways and an identical discount rate of 3% for a ton of CO₂ emitted in the year 2020. The average value across all models and runs is $42/ton, which is the most frequently cited number. In an effort to be transparent, the United States Environmental Protection Agency published the full set of estimates for different assumptions about the discount rate and year of emissions. Table 2.2 below is a reproduction of the estimates.

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<td>3% Average</td>
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<tr>
<td>2015</td>
<td>$11</td>
<td>$36</td>
</tr>
<tr>
<td>2020</td>
<td>$12</td>
<td>$42</td>
</tr>
<tr>
<td>2025</td>
<td>$14</td>
<td>$46</td>
</tr>
<tr>
<td>2030</td>
<td>$16</td>
<td>$50</td>
</tr>
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<td>2035</td>
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<td>2040</td>
<td>$21</td>
<td>$60</td>
</tr>
<tr>
<td>2045</td>
<td>$23</td>
<td>$64</td>
</tr>
<tr>
<td>2050</td>
<td>$26</td>
<td>$69</td>
</tr>
</tbody>
</table>

Source: Author’s compilation.

A few things stand out from this table. First, as mentioned above, the higher the discount rate, the lower the SCC estimate. For a ton of CO₂ emitted in 2020, if one applies a discount rate of 2.5%, the SCC is $68, while for a 3% discount rate, this number falls to $42; if one goes to 5%, the number drops to $12. The last column only looks at the 95th percentile of the distribution of estimates and discounts it at 3%, which results in a much higher SCC, since only the highest estimates are considered by design. For 2020, this number is $123 per ton.

The second aspect to notice in this table is the fact that the SCC is increasing over time—in other words, the later a ton is emitted, the higher the SCC. For example, using the 3% discount rate, a ton emitted in 2020 causes $42 in discounted damages, while a ton emitted in 2050
causes $69 in discounted damages. There are two reasons for this. The first is that as time goes on, the stock of CO₂ in the atmosphere increases, resulting in incrementally higher damage due to nonlinearities. The second reason that the SCC is increasing over time is that damages in some models are proportional to income, which is a reasonable assumption given that, for example, the value of assets affected by climate change is thought to be increasing in line with income, which in turn is expected to grow over time.

Following the publication of the IWG SCC estimates, the Obama administration asked the National Academies of Sciences, Engineering, and Medicine (NAS) to author an independent report assessing the IWG effort and providing a path forward. The NAS (2017) report was published and has given rise to an effort at Resources for the Future, a Washington, DC-based think tank, to implement the recommendations. This effort is being conducted outside the federal government as the current administration has disbanded the IWG and halted all work on the SCC. The White House has instructed all agencies to use a domestic SCC using discount rates as high as 7%, which led to SCCs ranging from $1 to $7 per ton. A main recommendation of NAS was to improve drastically the quality and coverage of damage functions in the IAMs used for the calculation of the SCC. The next section discusses what has been done historically to estimate damage functions and what the current best practice is.

### 2.5 Damage Function Estimation

Damage function estimation helps conceptualize what a damage function is attempting to do. It sounds simple. A damage function maps climate into an outcome. This has often been described as how a long-run average of weather (climate) maps into an outcome of interest (e.g., electricity consumption, agricultural yields). Figure 2.3 helps fix these ideas.

When you leave your house in the morning, you have to decide what to wear. If you look outside your window and it is cold and rainy, you will wear warmer clothes and bring an umbrella, if you have one. The next day it may be sunny and warm, and you will leave your home in a short-sleeved shirt. What you encounter daily is weather. In most places, weather is highly variable across and even within seasons. Summer months are usually warmer and drier, while winter months are usually colder and wetter. Climate can roughly be characterized as the full statistical distribution of weather. If that distribution is stationary—meaning, the moments of the distribution are constant over time—one would still expect day-to-day variation in weather but, on average, expect a similar number of hotter and cooler days during
the summer, etc. Let us use a sector that is highly sensitive to weather: electricity consumption. Assume a location with a cool pleasant climate, like San Francisco. Fashionable San Franciscans old and young alike live in houses and apartments that usually do not have air conditioning equipment. What this means is that on the occasional very hot day, San Franciscans will complain loudly and head to the park to eat ice cream, hoping for cooler days. Since these hot days are a rare occurrence in a world without climate change, the cost of installing and operating air conditioning may be greater for most residents compared to the benefits they would derive from the few days they would use it. However, if San Francisco inherits the climate of Beijing, which is much warmer and more humid, especially in the summer, most San Franciscans would probably find it optimal to install air conditioning. Hence, electricity use would go up. We call this adaptation. The top left panel of Figure 2.4 in
light gray displays the weather from a pre-climate-change world. The temperature series is mapped via the damage function into electricity consumption. This pre-climate-change damage function is shown at the top right panel by the solid line. This translates into pre-climate-change electricity consumption shown by the dark solid line at the bottom right panel. Now, if climate changes and the weather are drawn from a distribution with a higher mean and variance, as indicated by the dark, solid time series at the top left panel, the question arises as to what the right damage function is. If individuals understand that climate has changed, they will adapt (in our case, install more air conditioning) and the damage function changes. The new damage function is steeper, especially at higher temperatures. This means that the new normal weather is mapped into electricity consumption through the “with adaptation” damage function at the top right panel. The correct post-climate-change consumption time series is the light gray dotted time series at the bottom right panel. The difference between the dotted line and solid line in the bottom panel is the damage from climate change.

Why is this so important? A damage function one would want to use in an IAM should account for adaptation, carry a causal interpretation, and have global coverage. This sounds straightforward but turns out to be quite difficult in practice. The literature on damage function estimation goes back about 5 decades and can be split into roughly five strands.

The first and most widely adopted approach for estimating climate change damage stems from a seminal paper by Mendelsohn, Nordhaus, and Shaw (1994). They study the agriculture sector for the US, but the method has been applied in many other sectors (e.g., energy consumption). The insight behind this method can best be demonstrated by Figure 2.4 below.

Assume that a farmer is currently planting crop 1 as (s)he is operating in an area with a cool climate. If the climate gets warmer, (s)he could continue to grow the same crop and make profits as given by point D. However, the farmer could make profits as indicated by point C by simply switching crops. If one assumes rational, profit-maximizing farmers, the impact of climate change on profits is given by the thick black line in the figure, the envelope of the individual crop/use profit functions, which are functions of, for example, temperature. Econometrically, this is relatively straightforward to implement. If one has data on profits and climate for farmers across different climate zones, one could assume that, conditional on other “confounders,” the cross-section would give one the envelope of the profit functions exactly. How does one implement this in practice? One runs a cross-sectional regression of outcomes of interest (e.g., agricultural profits) on complex functional forms of long-run...
averages of weather (= climate) and other observable confounders such as soil quality and proximity to roads, to name some examples. The advantage of this method is that it actually identifies the effect of climate on the outcome of interest. The disadvantage of this method is that if one omits confounders that explain variation in the outcome of interest and that are correlated with climate, the estimation suffers from omitted variables bias. The consequences of this can be grave.

Another method, which is less frequently used, is simply using time series regressions of outcomes of interest on functions of weather. One example is Franco and Sanstad (2008), who regress the electricity load for the state of California on population-weighted averages of weather. Other more recent examples of this work are Auffhammer, Baylis, and Hausman (2017) and Wenz, Levermann, and Auffhammer (2017), who estimate the impacts of weather on electricity for the entire US and European countries individually. The advantage of this approach is that it allows researchers to estimate dose response/damage functions for variables that are only available at very coarse levels of aggregation.

Note: Imagine a single farmer, who is currently growing crop 1 and earning profits corresponding to the y value at point A. If faced with a significantly hotter climate, the farmer becomes indifferent between growing crop 1 and crop 2 at point B. If the climate warms further still, the farmer would be much better off at point C (switching to crop 2) rather than at point D (continuing to grow crop 1).

Source: Auffhammer (2018).
However, these regressions only identify the effect of weather on the outcomes of interest and do not incorporate the impacts of adaptation, which is a significant, if not detrimental, drawback.

A third method, which has been more frequently applied in the recent damages literature, has employed the use of longitudinal data sets, often referred to as panel data sets. Early examples of these papers are Auffhammer, Ramanathan, and Vincent (2006) and Deschênes and Greenstone (2007). These papers build on the time series regression approach discussed above but have the advantage that one has observations on an outcome of interest for multiple units (e.g., states or counties) over several years. The existing literature has used variation across time and cross-sectional units to estimate the dose response/damage function. The advantage of this approach is that one can control for confounders through a fixed effects strategy that controls for unobservable differences that are time invariant across units and shocks common to all or a subset of units for a given year. Hence, the risk of omitted variables bias that the Ricardian approach suffers from is much lower here. The drawback is the same issue that time series regressions suffer from. The response functions estimate a relationship between an outcome of interest and weather—not climate, which lacks an adaptation response.

A recent paper by Burke and Emerick (2016) provides a fourth and very clever way to address the “confoundables” and the adaptation issue at the same time. The authors demonstrate a new method using data on US agricultural output across counties, benefiting greatly from the availability of highly disaggregated annual data over a long period. They calculate a long difference in agricultural output, where they determine the difference between a 5-year moving average at the end and the beginning of their sample period. This long difference is the outcome of interest, telling us how much agricultural yields or profits have changed over, say, 40 years. On the right-hand side, they use trends in temperature over the same period. The differencing is equivalent to including unit fixed effects, and using trends gives the estimated coefficients a long-run interpretation, which contains adaptation. Burke and Emerick (2016) show for their data that there is little evidence of an adaptation response. This approach has not been used widely outside agriculture.

The fifth and most recent approach has incorporated panel data estimation techniques but augmented the specifications by interacting the weather response coefficients with cross-sectional variables measuring both climate (long-run averages of weather) and income. A recent example of this work is Auffhammer (2018). The idea is a
simple one. The weather response of, for example, energy consumption is different in areas with a hot versus a cool climate. Interacting the weather variables with a cross-sectional climate variable allows for empirical estimation of this response heterogeneity. An interaction of the weather variables with cross-sectional income allows for heterogeneity in the weather response as a function of income. Richer economies are thought to be able to adapt more easily, and this interaction captures this difference in response. The beauty of this approach is that if one has projections of income and climate, one can simulate how the damage function changes as climate and incomes change going forward. Carleton et al. (2019) make some strong assumptions but, using a data set on mortality for the majority of humankind, provide extrapolations of climate change–induced mortality for all countries to the end of the 21st century. This approach is possible with shorter time series than the long differences approach. It also has the benefit of shifting the response function, which the long differences approach cannot do, in a forward-looking way. If one is interested in the detection and attribution of historical climate change, I would argue that the long differences approach is the best tool available, data permitting.

The papers discussed above mostly focus on single economies for individual sectors. In order to construct a damage function for all sectors across the world, one would need to write a few thousand empirical papers to get good coverage. I would encourage young scholars to start writing these papers. Another approach is to estimate a single damage function using GDP data, measuring the value of all goods and services produced for a country in a given year. Burke, Hsiang, and Miguel (2015) wrote a paper providing a correlation between the growth rate of GDP and a nonlinear measure of changes in temperature to estimate how damaging climate change will be to the major world economies. While there is discussion about the empirical model adopted in the paper, they show a highly nonlinear relationship between GDP growth rate and temperature which resembles an environmental Kuznets curve—an inverse U. Growth rates peak at about 13°C and are increasing at lower temperatures and decreasing at higher temperatures. The issue with approaches like this is that GDP does not measure everything that has value. It excludes the value of non-market resources, which is likely to be significant. Furthermore, this paper again uses short-run variation. For technical reasons discussed in McIntosh and Schlenker (2006), the approach here includes some adaptation response but not complete adaptation. In the next section, I will discuss some of these results and provide some context for the Asian economies.
2.6 Overview of Damage Estimates of Asian Economies

While the previous section discussed the different approaches to damage estimation, there is far from complete coverage for the different sectors of each economy. Several studies use the hedonic approach for agriculture while a few panel data studies are for agriculture, mortality, and energy consumption (Carleton and Hsiang 2016), and one study uses the panel data approach accounting for adaptation (Carleton et al. 2019). There is, of course, a larger literature in field journals and white papers that use various methods for single sectors. In this section, I provide an overview of the types of results targeted at providing estimates of the impact of climate change. The first set of estimates stems from the paper by Burke, Hsiang, and Miguel (2015) discussed in the previous section, using a simple regression framework accounting for partial adaptation. The projected climate change impacts shown in Table 2.3 for selected member countries of the Asian Development Bank (ADB) are presented for the worst-case emissions scenario and are calculated relative to per capita GDP in the year 2010. The estimates from this paper are the only country-level impacts of climate change currently available for most countries.

The average impact on per capita GDP across countries by the mid-century is –13.6%, and the average impact by the end of the century is –12.35%. These numbers do not sound unreasonable and are slightly higher than the number predicted by some IAMs. Yet what stands out here is the massive spread in impacts. At mid-century, the range of impacts just from climate change on per capita GDP is from –40% (Pakistan) to +88% (Mongolia). If we go to the end of the century, the spread of projections becomes even larger. Projected impacts range from –87% (Pakistan) to +881% (Mongolia). These projections have received a significant amount of media attention, and it has been widely reported that global per capita GDP is projected to decrease by roughly 23% by the end of the century. Before using these numbers in decision-making, it is important to remember that this paper has been criticized for making some strong functional form assumptions, where temperature affects the growth rate instead of the level of per capita GDP, which means that the shocks propagate through time. A recent working paper by Newell, Prest, and Sexton (2018) has pointed out that functional form assumptions have significant consequences for the point estimates. Furthermore, GDP only measures market impacts and ignores non-market impacts and is, hence, incomplete.

Another recent paper (Ricke et al. 2018) uses the GDP damage function of Burke, Hsiang, and Miguel (2015) and another paper by
The Economic Impacts of Climate Change: Implications for Asian Economies

Table 2.3: Climate Change Impacts on Selected Asian Development Bank Member Countries (%)

<table>
<thead>
<tr>
<th>Country</th>
<th>Change in GDP per Capita, 2040–2059</th>
<th>Change in GDP per Capita, 2080–2099</th>
</tr>
</thead>
<tbody>
<tr>
<td>Afghanistan</td>
<td>−4.78</td>
<td>−28.43</td>
</tr>
<tr>
<td>Armenia</td>
<td>19.28</td>
<td>73.50</td>
</tr>
<tr>
<td>Australia</td>
<td>−12.60</td>
<td>−44.77</td>
</tr>
<tr>
<td>Azerbaijan</td>
<td>−2.53</td>
<td>−18.17</td>
</tr>
<tr>
<td>Bangladesh</td>
<td>−36.49</td>
<td>−83.65</td>
</tr>
<tr>
<td>Brunei Darussalam</td>
<td>−34.16</td>
<td>−81.47</td>
</tr>
<tr>
<td>Bhutan</td>
<td>−1.17</td>
<td>−11.76</td>
</tr>
<tr>
<td>China, People’s Republic of</td>
<td>−7.51</td>
<td>−34.07</td>
</tr>
<tr>
<td>Fiji</td>
<td>−23.63</td>
<td>−65.40</td>
</tr>
<tr>
<td>Georgia</td>
<td>5.52</td>
<td>8.94</td>
</tr>
<tr>
<td>Indonesia</td>
<td>−31.44</td>
<td>−77.93</td>
</tr>
<tr>
<td>India</td>
<td>−38.78</td>
<td>−86.16</td>
</tr>
<tr>
<td>Japan</td>
<td>−5.97</td>
<td>−28.23</td>
</tr>
<tr>
<td>Kyrgyz Republic</td>
<td>29.53</td>
<td>132.64</td>
</tr>
<tr>
<td>Cambodia</td>
<td>−38.94</td>
<td>−81.57</td>
</tr>
<tr>
<td>Korea, Republic of</td>
<td>3.09</td>
<td>1.89</td>
</tr>
<tr>
<td>Kazakhstan</td>
<td>32.17</td>
<td>158.87</td>
</tr>
<tr>
<td>Lao People’s Democratic Republic</td>
<td>−32.21</td>
<td>−79.17</td>
</tr>
<tr>
<td>Sri Lanka</td>
<td>−32.14</td>
<td>−78.71</td>
</tr>
<tr>
<td>Mongolia</td>
<td>87.81</td>
<td>881.11</td>
</tr>
<tr>
<td>Malaysia</td>
<td>−33.53</td>
<td>−80.70</td>
</tr>
<tr>
<td>Nepal</td>
<td>−31.08</td>
<td>−78.09</td>
</tr>
<tr>
<td>New Zealand</td>
<td>−0.41</td>
<td>−6.41</td>
</tr>
<tr>
<td>Papua New Guinea</td>
<td>−24.30</td>
<td>−66.96</td>
</tr>
<tr>
<td>Philippines</td>
<td>−30.61</td>
<td>−76.38</td>
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<td>Pakistan</td>
<td>−39.54</td>
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<td>Solomon Islands</td>
<td>−31.35</td>
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<td>Thailand</td>
<td>−37.81</td>
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<td>1.36</td>
<td>−9.47</td>
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<td>Viet Nam</td>
<td>−33.60</td>
<td>−80.82</td>
</tr>
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<td>Vanuatu</td>
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<td>−69.40</td>
</tr>
<tr>
<td>Samoa</td>
<td>−27.87</td>
<td>−72.30</td>
</tr>
</tbody>
</table>

GDP = gross domestic product.
Dell, Jones, and Olken (2012) to calculate a country-specific SCC. As discussed above, in terms of global policy making, the number that matters from a global perspective is the global SCC. There is, of course, the issue that mitigation happens at the country level. An argument for a domestic SCC suggests that countries will only find it optimal from a domestic perspective to take action if the benefits of action outweigh the costs. Hence, one would want to calculate a country-specific SCC, which ignores damage caused elsewhere. The only framework that currently has country-level resolution is the Burke, Hsiang, and Miguel (2015) framework. Ricke et al. (2018) used this framework to calculate an SCC at the country level and characterize the uncertainty around it. To give us an idea of the magnitude, I have used their publicly available estimates and generated the median, 16.7th, and 83.3rd percentiles at the country level for ADB member countries with available estimates. I only included runs for SSP3 and 5, using the emissions scenario RCP 8.5 and a constant discount rate of 3%, yet using all available damage functions in the paper. Hence, these estimates differ slightly from the main estimates in the paper. The aggregate SCC across all countries for this study is estimated to be $1,060, which is significantly higher than the IWG estimate of $42 per ton. This is largely due to the damage function estimated by Burke, Hsiang, and Miguel (2015). If one takes that approach at face value, Table 2.4 reports the estimated SCC at the country level.

What emerges from this table is that the SCC for the smaller countries is, well, smaller. Population and economy size matter in this calculation. Hence, bigger economies are more likely to have larger estimates. The People’s Republic of China (PRC) and India have country-level SCCs above $100. Very few countries have country-level SCCs that are negative. The most noteworthy of these is Mongolia which, according to Burke, Hsiang, and Miguel (2015), is projected to experience significant increases in per capita GDP from climate change.

While, for the purposes of this chapter, the country-level estimates are interesting, I would like to caution the reader from taking these point estimates at face value since they depend on two papers (Burke et al. 2015; Dell et al. 2012), which adopt a very specific functional form to arrive at a damage function. The overall damages predicted by these models are much larger than those predicted by more recent incarnations of the classic IAMs. The DICE model by Nobel Laureate Bill Nordhaus in a recent publication (Nordhaus 2017) estimates an SCC of $31. This is, of course, significantly lower than what Ricke et al. (2018) report. An important route forward, in my view, is to build empirically validated sectoral damage functions that cover as much of the globe as possible and aggregate across space. The Climate Impact Lab is currently working on developing a credible method to do so, which should be applied to
Table 2.4: Country-Level Social Cost of Carbon

<table>
<thead>
<tr>
<th>Country</th>
<th>CSCC ($, 17th Percentile)</th>
<th>CSCC ($, Median)</th>
<th>CSCC ($, 83rd Percentile)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Afghanistan</td>
<td>-2.94</td>
<td>0.88</td>
<td>2.22</td>
</tr>
<tr>
<td>Armenia</td>
<td>-0.29</td>
<td>-0.08</td>
<td>0.04</td>
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<tr>
<td>Australia</td>
<td>1.20</td>
<td>5.89</td>
<td>7.53</td>
</tr>
<tr>
<td>Azerbaijan</td>
<td>-0.08</td>
<td>0.33</td>
<td>0.56</td>
</tr>
<tr>
<td>Bangladesh</td>
<td>7.57</td>
<td>10.33</td>
<td>11.68</td>
</tr>
<tr>
<td>Brunei Darussalam</td>
<td>-0.02</td>
<td>0.12</td>
<td>0.14</td>
</tr>
<tr>
<td>Bhutan</td>
<td>0.08</td>
<td>0.16</td>
<td>0.20</td>
</tr>
<tr>
<td>China, People’s Republic of</td>
<td>45.68</td>
<td>115.23</td>
<td>155.16</td>
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<tr>
<td>Fiji</td>
<td>0.03</td>
<td>0.04</td>
<td>0.05</td>
</tr>
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<td>Georgia</td>
<td>-0.10</td>
<td>0.08</td>
<td>0.19</td>
</tr>
<tr>
<td>Indonesia</td>
<td>15.24</td>
<td>25.64</td>
<td>29.92</td>
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<td>76.31</td>
<td>116.98</td>
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<td>0.61</td>
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<td>0.47</td>
<td>0.88</td>
<td>1.04</td>
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<tr>
<td>Korea, Republic of</td>
<td>-3.53</td>
<td>2.97</td>
<td>5.92</td>
</tr>
<tr>
<td>Lao People’s Democratic Republic</td>
<td>0.37</td>
<td>0.52</td>
<td>0.60</td>
</tr>
<tr>
<td>Sri Lanka</td>
<td>1.12</td>
<td>1.90</td>
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<td>-16.34</td>
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<td>2.06</td>
<td>5.91</td>
<td>6.96</td>
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<td>Nepal</td>
<td>1.32</td>
<td>1.68</td>
<td>1.92</td>
</tr>
<tr>
<td>New Zealand</td>
<td>-0.17</td>
<td>0.40</td>
<td>0.68</td>
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<td>Pakistan</td>
<td>8.42</td>
<td>11.38</td>
<td>13.24</td>
</tr>
<tr>
<td>Philippines</td>
<td>5.45</td>
<td>8.34</td>
<td>9.59</td>
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<tr>
<td>Papua New Guinea</td>
<td>0.56</td>
<td>0.75</td>
<td>0.86</td>
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<td>Solomon Islands</td>
<td>0.03</td>
<td>0.05</td>
<td>0.06</td>
</tr>
<tr>
<td>Thailand</td>
<td>3.95</td>
<td>8.41</td>
<td>9.96</td>
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<td>Tajikistan</td>
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<td>0.28</td>
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<td>0.71</td>
<td>0.88</td>
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<td>Uzbekistan</td>
<td>0.27</td>
<td>1.29</td>
<td>1.82</td>
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<tr>
<td>Viet Nam</td>
<td>4.45</td>
<td>6.34</td>
<td>7.42</td>
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<td>Vanuatu</td>
<td>0.02</td>
<td>0.04</td>
<td>0.04</td>
</tr>
<tr>
<td>Samoa</td>
<td>0.00</td>
<td>0.01</td>
<td>0.01</td>
</tr>
</tbody>
</table>

CSCC = country-level social cost of carbon.
more sectors than mortality, agriculture, and energy consumption. The Climate Impact Lab comprising experts from the University of Chicago; University of California, Berkeley; Rutgers University; and the Rhodium Group, is aggressively pursuing this approach. I would encourage readers to follow their advances and start expanding this literature.4

2.7 Mitigation

While the previous sections discussed projected physical and economic impacts of climate change, at the very heart of the problem is the fact that GHG emissions have grown at a steady pace since the dawn of the industrial revolution. Figure 2.5 shows the trajectory of emissions since 1751.

![Figure 2.5: Global Emissions of Carbon Dioxide](image)

The figure makes two points. First, the growth of emissions in the postwar period was massive. Annual emissions went from roughly 5 billion tons in 1950 to 35 billion tons in 2017, a sevenfold increase.

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4 The Climate Impact Lab can be found at https://www.impactlab.org.
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The second aspect to notice is that the global fuel mix has changed tremendously. While in the early years of the 20th century the vast majority of emissions came from coal, today these account for less than half. Liquid fuels, largely driven by increases in transportation demand, are now the second-biggest source of emissions, followed by natural gas.

Since CO₂ is a stock pollutant when using a human time horizon, one should ask who the biggest contributors to the stock of GHGs in the atmosphere are by region. Figure 2.6 below breaks down the cumulative emissions for several countries and aggregates.

This figure provides two interesting insights. In the early days of industrial carbon emissions, the lion’s share of emissions came from the home of the industrial revolution, the United Kingdom, with almost 100% of cumulative emissions. In the late 19th century, emissions in the US started growing rapidly, and the US share in cumulative emissions rose to a peak of 40% by the end of World War II. What one observes starting in the late 20th century is the emergence of the PRC and India as major sources of annual emissions. The growth in emissions was so rapid that even though aggregate annual emissions were growing

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**Figure 2.6: Cumulative Carbon Dioxide Emissions for a Subset of Countries and Aggregates**

- United States
- EU-28
- PRC
- United Kingdom
- India

EU = European Union, PRC = People’s Republic of China.

Source: Our World in Data based on Global Carbon Project (GCP).
quickly during this period, the PRC share in cumulative emissions broke 10% of total emissions, while India’s is hovering around 5%. Maybe the most significant insight behind this figure is that the slope of the emissions trajectories for the US and the European Union is declining and the share for the rapidly developing economies of India and the PRC is increasing, with the rate of change increasing.

The drivers of this growth are not surprising. In the climate literature, there is a simple decomposition of emissions called the Kaya identity. It hypothesizes that CO$_2$ emissions can be decomposed into a product of population, GDP per capita, energy intensity per unit of GDP, and carbon intensity (carbon per unit of energy consumed). If one takes this equation at face value, it suggests that more populous, richer, energy-intensive countries with a more carbon-intensive energy sector will have larger emissions. This is not surprising and has been confirmed in largely cross-sectional decomposition analyses. Taking a closer look at one part of this equation, the income emissions relationship is instructive. One common way to look at this is to plot the relationship between per capita emissions and per capita income, which compares measures per person across countries. A snapshot of how average income and emissions correlate is provided in Figure 2.7 for 2016.

**Figure 2.7: Per Capita Carbon Dioxide Emissions Plotted Against Per Capita Gross Domestic Product**

CO$_2$ = carbon dioxide, GDP = gross domestic product, Lao PDR = Lao People’s Democratic Republic, PRC = People’s Republic of China.

Source: Global Carbon Project, Maddison (2017).
It is important to note that this figure is plotted on a log-log scale. This suggests a highly nonlinear relationship between per capita emissions and income, higher incomes being consistent with significantly higher emissions. Sub-Saharan economies are shown to have a per capita GDP below $1,000 per person and emissions below 0.1 ton per year. The US, whose GDP is approaching $60,000 per capita, has per capita emissions exceeding 10 tons per person year. That is a difference of two orders of magnitude. The sign of this effect is intuitive, as wealthier countries consume (and produce) more goods and services, and therefore have higher emissions.

The cross-sectional figure below ignores what happens over time. The questions we need to ask are what will happen to the countries located at the bottom left of this graph as time goes on? Will they migrate to the top right quadrant, which means they would be wealthier—which is arguably a good thing—but also produce significantly higher emissions per person? The real question is what will happen to the energy intensity and carbon intensity of economies across the board? Will lower carbon intensity decrease overall emissions? Will improved energy efficiency be instrumental in driving down emissions? In a dream world, one would produce an ever-increasing amount of goods and services at rapidly declining levels of resource intensity, which is more than just decreasing carbon emissions.

There is a set of emissions scenarios that provide climate modelers with an idea of different “states of the world,” as emissions are a key input in the GCMs. Emissions for the next 80 years are the usual time horizon. Figure 2.9 below shows possible future emissions trajectories at the global level.

These worlds are, of course, extremely different. The top (yellow) fan indicates a world without climate policies, consistent with 4.1°C–4.8°C warming. The reported figures of warming here are for the global average temperature. Averages, as any statistician knows, are often misleading. In the context of climate change, we think that warming near the poles will be much higher than near the equator. This is, of course, problematic as the poles are covered by ice. Hot temperatures make ice melt, which is especially problematic at the South Pole, as that ice is sitting on land. If it melts, it will result in a significant rise in sea levels. The same is true for the Greenland ice sheet. The green trajectory in Figure 2.8 indicates a world with current policies, assuming that they are implemented and enforced, which will, of course, require significant commitment by regulators, politicians, and environmental authorities across roughly 200 countries. If we just stick with current policies, we will likely experience warming of 3.1°C–3.7°C. This is significantly above the targeted 2°C warming scenario that is thought to help us avoid the worst consequences of climate change. The purple scenario
incorporates pledges made by individual countries under the Paris Agreement, which I discuss in more detail below. If countries implement policies consistent with their pledges, which is an ambitious goal, this would reduce expected warming to 2.6°C–3.2°C, which is still above the target of 2°C. To get to 2°C, the red emission pathway needs to be met. This pathway is a significant challenge. It requires actual reductions in GHG emissions now. Given the continued growth of emissions, this is a tall order that I think is unrealistic. Given that CO₂ is a stock pollutant, one could shift reduction across time, but there is not much room to maneuver. Finally, the figure displays an ambitious goal of limiting warming to 1.5°C. This I think is unattainable from an economic point of view, since the reductions required are massive.

2.8 Global Policy

Countries across the world have actively engaged in designing a global agreement that will drive down GHG emissions to help avoid the worst scenarios of climate change. The first sign of this was the Rio Earth Summit in 1992, which ended in a general agreement by most countries to study what it would take to deal with the climate change problem. This resulted in the 1997 Kyoto Agreement, which had very little impact
on global emissions as the two largest emitters were not required to reduce their emissions. The 2015 Paris Agreement was the first global agreement that specified emissions targets for almost all countries—with the exception of the US, which spectacularly withdrew from the agreement under President Trump a few years after signing. At the time of writing of this chapter, all countries in Asia have signed the agreement and the major emitters have joined the agreement, which is the first positive step forward. This signals intent to do something about the problem among higher- and lower-income countries alike.

This begs the question of what was different about the Paris Agreement from previous attempts to bring together the countries of the world to engage in emissions reductions? I think this worked because individual countries came to the table with their own emissions reduction plans and targets, called intended nationally determined contributions (INDCs). These INDCs essentially stated what each country’s plans were post-2020 to reduce emissions. The word “intended” is removed once countries submit their ratification. For example, the PRC’s INDC stated that its emissions will peak by 2030 or earlier, with a 60%–65% reduction in emissions intensity per unit of GDP. India’s commitment is a 33%–35% reduction in the emissions intensity of its GDP by 2030 compared to the 2005 level. Specific links to the individual countries’ INDCs can be found on the World Resource Institute’s CAIT Climate Data Explorer.5

2.9 A Path Forward

Stating a willingness to do something about a problem is very different from actually doing something about it. This is very similar to the problem of an overweight person trying to lose weight: even with the best intentions, one often falls short. But there are rays of hope on the horizon. The choice facing each country is what tools to use to achieve their ambitious goals. The basic choice available to each country is a set of command and control strategies, or incentive-based tools, or a mix of both.

Command and control strategies usually come in three flavors: emissions standards, input standards, or technology standards. Emissions standards require individual firms, for example, to meet a specific emissions target. If firm A currently emits 1,000 tons of CO₂, an emissions standard could require it to reduce its emissions to, for example, 800 tons

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5 The CAIT Climate Data Explorer can be found at http://cait.wri.org. Another good resource is the Climate Watch Data website, which can be found at https://www.climatewatchdata.org.
or pay a fine. If the fine is larger than the cost of reducing emissions, the firm will reduce its emissions. The advantage of this approach is that if the fines are large enough and institutions can enforce a standard, one’s abatement goal can be reached. The disadvantage is that an emissions standard will almost never achieve emission reductions at least cost, since it does not consider the marginal cost of abatement, which is likely to differ across firms. To minimize the cost of abatement, the marginal cost of abatement should be equal across firms at the final level of emissions, which is called the equimarginal principle. Input standards differ from emissions standards in that they require firms to use a specific input to production. The simplest example I can think of is to require coal-fired power plants to use low, instead of high, sulfur coal. The advantage of this method is that it will likely lead to reductions in emissions if the new input is well chosen. However, input standards have been criticized along two dimensions: first, they offer very little incentive for emission reduction research and development, as firms will likely do what they are told and not look for lower cost options to reduce emissions using alternative technology. Second, the government chooses the “correct” input, which relies on regulators being able to pick the best input along technical and possibly cost dimensions. Furthermore, input standards will likely violate the equimarginal principle. Finally, technology standards prescribe a specific technology for either emissions reduction or production. Examples of this are catalytic converters, which were widely implemented in cars, or the nitrous oxide-scrubbing technology prescribed for many power plants across the planet. The advantage of this approach is that it will almost certainly lead to emissions reductions, yet ex ante quantifying how much emissions will be reduced by is difficult, making it hard to predict specific reductions. The two disadvantages are similar to those affecting input standards. Technology standards require governments to pick a technology, which they may not be well suited to do. Furthermore, they provide low to no incentive for technological improvements and likely violate the equimarginal principle.

Incentive-based methods come in essentially three types: taxes, tradeable permit systems, and subsidies (which are essentially a negative tax). These are economists’ preferred policy tools. Taxes, going back to Pigou, follow a simple goal—internalizing the negative externality generated by the production of a good. If a ton of CO₂ at the optimal emissions level causes $50 in damages, one should impose a tax of $50 per ton of carbon emitted. The insight behind this is a thing of beauty. If a firm is charged $50 per ton in addition to the private costs of input to production, the firm has an incentive to reduce its emissions, since costs have gone up. It will reduce its emissions until it is more expensive to reduce emissions compared to paying the tax. At that point, the firm
pays the full damage its marginal ton causes. This policy has many advantages. First, it reduces emissions to the socially optimal amount, assuming that the tax rate is set correctly. Furthermore, it provides significant incentives for firms to engage in emissions-related research and development to reduce their abatement costs. The tax also satisfies the equimarginal principle since all firms will reduce to the point where their marginal abatement costs are equal to the tax rate. Finally, a carbon tax generates significant revenue for governments, which can either be returned to producers and consumers or used to fund other government programs, such as social security or health care.

Tradeable permit systems work on a different dimension. The regulator chooses what the total desired amount of emissions is and issues a number of permits, equivalent to the total amount of desired emissions, which is hopefully below current emissions. Firms are allocated these certificates based on many possible criteria. Furthermore, there is a market for these permits. Firms will sell permits if they find reducing emissions to be cheaper than the market value of a permit. Firms will buy permits if they find it more expensive to reduce emissions than the price of a permit. In equilibrium, then, all firms produce at a point where marginal abatement costs are identical across firms, and if the number of permits is chosen correctly, the permit price will equal the external cost of emissions. This system has many advantages. First, total desired emissions reductions are achieved, as the number of permits dictates total emissions. Second, the equimarginal principle is again met. Third, this system provides significant incentives for firms to innovate in terms of lower cost abatement technologies. Finally, the policy tool generates significant revenue for governments, if initial permits are auctioned off.

In practice, some permits are usually given away to get some industries on board. While this may not be optimal from a government revenue point of view, it may just be from a political economy view. What are the disadvantages of this policy tool? The devil is in the detail here. There are issues with permit banking, allocation, and how long the permits last. Also, there are significant incentives for manipulating these markets. Moreover, from an ethical point of view, one may object to issuing a right to pollute. Finally, subsidies provide payments for adopting things. This could be low-emission technologies in our case. If we believe that a new technology needs a little “push” to be successful in the marketplace, one might want to subsidize this technology initially (e.g., electric vehicle subsidies). The advantage is that these subsidies lower the purchase price of the desired technology and result in higher adoption. The disadvantage is that governments must pick what to subsidize and by how much. This is hard to do. Finally, taking away subsidies is as politically as feasible as imposing new taxes.
In reality, governments across the world are implementing a mix of both types of tool at the same time. California, the world’s fifth-largest economy with one of the most aggressive emissions reduction goals, is a good example. The state has a cap and trade system, which covers many large sectors. It has a low-carbon fuel standard, which is targeted at increasing the share of low-carbon fuels in the transport sector. Also, it has a renewable portfolio standard designed to increase the share of renewables in electricity generation. It has some of the world’s most aggressive energy efficiency standards—and the list goes on. The cost of these emissions reduction goals ranges from negligible (energy efficiency) to wildly expensive (low-carbon fuel standard). Economists have pointed out repeatedly that one should at least attempt to achieve desired emissions reductions at the lowest cost. Unfortunately, the situation in California and in Europe is not even close to that, given the implementation of multiple emissions reduction tools at the same time.

That said, there is a glimmer of hope on the horizon in many Asian economies, the most significant of which may be the design of a tradable emissions system covering the electricity sector in the PRC. This would, for the first time, put a price on carbon at the national level for the world’s largest emitter. While this effort does not cover all sectors and many details remain to be worked out, this signifies an important first step on a hopefully rapid and impactful journey toward a world with limited climate change.
References


PART II

Environmental Regulations and Evaluation in Asia
3

Environmental Regulation: Lessons for Developing Economies in Asia

Jinhua Zhao

3.1 Introduction

Asia faces daunting environmental challenges. A recent report of the United Nations Environment Programme finds that about 4 billion people or over 92% of the population in Asia and the Pacific are exposed to air pollution exceeding or far exceeding the guidelines of the World Health Organization (WHO) (United Nations Environment Programme 2019). Some of the challenges arise from rapid industrialization, as evidenced by the severe air, water, and soil pollution in the People’s Republic of China (PRC) and India. Some arise from economic development leading to resource degradation, such as deforestation and loss of biodiversity in Southeast Asia. Despite the diversity in the nature, drivers, and possible solutions of environmental pollution across the nations in Asia, there are lessons about environmental regulation learned from decades of research and practice that can be useful for many Asian countries. This chapter aims to identify such lessons, discuss their theoretical background and practical applications, and explore their implications for Asia.

I will start by qualifying what this chapter does not do. First, Asia is large, with almost 50 countries and regions having all or some of their territories on the continent. The chapter does not even attempt to cover the details of environmental regulation in any single country. Second, Asia is heterogeneous in its stages of economic development but most Asian countries are developing economies. I will thus focus on lessons of environmental regulation for developing countries, while noting that developed economies (such as Japan) and oil-rich countries (such as Saudi Arabia) face unique environmental challenges. Third, although
there has been extensive research on environment and development, most of the review papers focus on the empirical side (e.g., Vincent 2010; Blackman 2010; Pattanayak et al. 2010; Somanathan 2010; Blackman et al. 2018). Although I will discuss both the empirical and the theoretical findings relevant for developing countries, most of the cited literature covers empirical findings.

Pollution is an environmental externality for which markets do not exist and thus polluters do not receive proper price signals that otherwise would steer their behavior toward socially optimal levels. This market failure calls for intervention, often by the government, to correct the externality. The bulk of economic research on environmental regulation deals with two issues on how to correct the externality: (i) the type of regulation, i.e., instrument choice among taxes, standards, and tradable permits; and (ii) the stringency of regulation, i.e., the appropriate levels of the instrument once it is chosen. Besides these top-down government regulations, there is considerable research on bargaining in economic theory that is relevant for pollution control. Over the last 2 decades, there has been growing research on informal regulation such as releasing information about firms’ environmental performance and voluntary approaches where firms engage in environment-friendly behavior without an explicit requirement by the government. My discussion will focus on instrument choices in formal regulation and how they can be complemented by informal regulation.

The core economic theory of environmental regulation was advanced in the context of developed economies. Some fundamental and often implicit assumptions underlying the theory are violated in developing countries, limiting the applicability of the theoretical findings. For example, although market-based instruments such as permit trading are often adopted in developed nations, many developing countries lack the capacity to quantify historical emissions of firms, making it difficult to determine the initial grandfathered allocations of permits. Further, even for thick markets where many polluters with large potential for permit trading exist, there might be a lack of market makers so that transaction costs cannot be reduced quickly. In discussing the theoretical findings, I will highlight the main assumptions and critical conditions needed for the findings to hold, show how some conditions are violated in developing country settings, and discuss which lessons are relevant for environmental regulation in developing countries.

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1 An academic journal, Environment and Development Economics, is devoted to this subject, and Review of Environmental Economics and Policy published a special issue in 2010 (Volume 4, Issue 2) devoted to environmental quality and economic development.
The rest of the chapter is organized as follows. In Section 2, I discuss the main components and findings of the economic theory of environmental regulation, and the (often ideal) conditions under which the findings hold. I move on to discuss the special characteristics of developing countries in Section 3, and the lessons learned about environmental regulation in these countries due to these characteristics in Section 4. I will conclude in Section 5.

### 3.2 A Primer on the Economics of Environmental Regulation

A core argument of neoclassical economic theory is that free markets can allocate scarce resources in a Pareto-efficient way under a set of ideal conditions, including perfect competition and the absence of externalities. In this ideal world, the role of the government is limited to ensuring property rights and free competition. However, free markets fail to be efficient in the presence of pollution externalities, and the field of environmental economics is largely devoted to finding ways to correct or internalize such externalities, with most of the approaches involving an active role by the government. Market failure is illustrated in Figure 3.1, which compares the market outcome with the socially

![Figure 3.1: Market Failure Due to Pollution Externalities](source: Author)
optimal allocation. Suppose a firm emits pollution in its production process, with the emission level represented on the horizontal axis. From the firm’s perspective, it chooses emission $E_{self}$ to equate the marginal benefit from emission MB (e.g., marginal profit from increased output) with its own marginal cost $MC_{self}$ (e.g., the marginal cost of using more coal). However, the pollution causes damage to society so that the social marginal cost $MC_{all}$ is higher than $MC_{self}$, and the optimal social emission $E_{all}$ is lower than $E_{self}$. Competition in the presence of pollution externalities leads to too much pollution.

The literature on environmental economics has studied five main approaches to correcting the externalities, with varying degrees of government involvement. These include pollution standards, Pigouvian taxes, tradable emission permits, Coase bargaining, and informal regulation such as information and voluntary approaches. Policies in practice are often combinations of these instruments but understanding the pros and cons of each is important in assessing the potential effects of these hybrid policies. We will discuss each of these regulatory instruments but we start by describing the criteria that have been proposed to compare them and to choose the optimal regulatory approach among them.

### 3.2.1 Criteria of Instrument Selection

Economic theory has primarily focused on two main criteria in choosing the level and format of environmental regulation. *Social optimality or efficiency* is a first-best criterion whereby the regulation is chosen to maximize social welfare, which in ideal conditions (e.g., no uncertainty, no information asymmetry, and no transaction costs) requires equating the marginal damage of pollution with the marginal cost of abatement. In Figure 3.1, a regulation would have to achieve the socially optimal emission level of $E_{all}$ with minimum cost to be socially efficient. Applying this criterion requires having sufficient information on the marginal damages and costs, which is difficult to obtain.\(^2\) Economists thus often resort to a less demanding second-best criterion of *cost effectiveness*, which requires minimizing the total costs of achieving a certain pollution level, while noting that the pollution level might not

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\(^2\) The bulk of empirical research on the costs of pollution, e.g., studying the health effects of pollution, focuses on obtaining values related to the total value of pollution damages rather than marginal damages (e.g., Currie et al. 2014; Ebenstein 2012). Nonmarket valuation methods, especially revealed preference methods such as travel cost models and hedonic price models, have been used to study the value of improved environmental quality, but the focus has not been on identifying marginal damages (Kling et al. 2012).
be socially optimal. In Figure 3.1, a regulation will be cost effective if it reduces emissions from \( E_{self} \) to a lower level \( at \ minimum \ costs \), but not necessarily to \( E_{all} \). Cost minimization is achieved if the marginal abatement costs are equalized across all polluting firms; otherwise, the total abatement cost can be reduced by making a high-cost firm abate less while a low-cost firm abates more.

Additional criteria have been proposed to evaluate the desirability of regulatory approaches, including incentives for polluting firms to adopt new abatement technologies, and administrative costs that include multiple components, such as regulatory simplicity, information requirement, transaction cost of implementation, etc. Given that environmental regulation is mostly at levels lower than socially optimal, it has been argued that environmental regulation should aim to encourage the innovation and adoption of new abatement technologies (Zhao 2003). This criterion has sometimes been called “dynamic efficiency” in contrast to the “static” efficiency discussed above. Market-based approaches such as taxes and tradable permits have been found to provide more incentives than standards, by putting less restrictions on what firms can do (including adopting new technologies). On the other hand, as we show below, standards have the advantage of having lower administrative costs, by requiring less information and reducing the complexity of transactions, than taxes and tradable permits. Finally, instrument choices can be affected by the institutional settings of a country. Ye and Zhao (2016) show that in countries with public firms contributing to pollution, the promotion incentives of public firm chief executive officers can be important in influencing the choice of optimal policy instruments.

Despite the extensive literature about these criteria, they have not always been influential in practical policy making, especially in developing countries. In fact, the less-studied criteria of administrative costs are often more important for developing nations (O’Connor 1999). In this chapter, I will emphasize the practical applicability of these criteria rather than their theoretical appeal. As such, I will not discuss one criterion, as proposed by Weitzman (1974), that has been influential in academic research on choices between tax and tradable permits under uncertainty and asymmetric information. The criterion requires information about the slopes of marginal benefit and marginal cost curves in Figure 3.1, which is extremely difficult to obtain.

### 3.2.2 Command-and-Control Approach: Standards

The most commonly used regulatory instrument, especially in developing countries, is emission standards; they are imposed on firms and differ
from ambient environmental quality standards. There are different kinds of standards. Absolute standards cap total emissions by polluting sources; performance standards limit emissions per unit of output or input; and technology standards specify the equipment, processes, or inputs firms must use (Helfand 1991). In Figure 3.1, an absolute standard could be imposed on firms so that each emission is capped at a level lower than \( E_{self} \). The European emission standards for vehicles, which serve as a benchmark for many countries in Asia such as in heavily polluted cities in the PRC and India, are performance standards that limit the amount of air pollutants emitted per kilometer driven for certain vehicle types. Standards are the main forms of \textit{command-and-control (CAC) approaches} where the government simply limits what firms can do as opposed to \textit{market approaches} where the government provides financial incentives to influence firm behavior, although violators of standards also face hefty fines.

Standards have the advantage of being relatively intuitive, straightforward, and transparent: if firms cause pollution, it makes intuitive sense to restrict their behavior. They often seem fair in the sense that polluters causing harm to the environment are restricted in their behavior, and such restrictions are uniformly imposed on all polluters. Standards are typically imposed on new firms, which can sometimes become significant entry barriers for new firms, thereby protecting the interests of existing firms. These features help standards become more politically feasible than other instruments. The Corporate Average Fuel Economy standards were adopted in the United States (US) partly because the first-best tool of taxing gas emissions is politically infeasible.

Compared with taxes and permit trading, standards target observable behavior and thus require less effort in terms of monitoring and enforcement. It might be difficult to monitor actual firm-level emissions but it is much easier to observe whether a firm has installed certain abatement equipment, or to track its use of dirty inputs such as coal. Thus, when taxing actual emissions is infeasible, the government can impose technology standards on equipment use or input standards. This is extremely important for developing countries, where the monitoring of emissions is typically inadequate and could be a major reason why many developing countries adopted standards as their primary regulatory approach.

Another main reason for the widespread adoption of standards is that they can bring observable environmental improvements quickly. Most environmental standards in the US were adopted in the 1970s when the government wished to obtain quick and big environmental improvements in response to serious pollution (Portney 2000).
In developing countries, Blackman, Li, and Liu (2018) find that CAC approaches (mainly standards) have worked better than market-based approaches in bringing environmental improvements. Environmental campaigns in the PRC mostly take the form of new standards or more enforcement of existing standards (Van Rooij 2006).

Despite their widespread use, standards suffer from a major drawback: they are neither efficient nor cost effective because uniform standards are imposed on heterogeneous firms. Firms are heterogeneous in many aspects, leading to their facing different abatement costs. They should thus undertake different levels of abatement, with low-cost firms abating more and high-cost firms abating less. Standards, by failing to attend to such heterogeneity, can be extremely costly. During the 1990s, the US removed several standards imposed on its major power plants and replaced them with a sulfur dioxide permit trading program. During the early phases of the program, permit trading was sporadic but the abatement costs went down by as much as 50% due to the flexibilities the power plants enjoyed, thanks to the removal of the rigid standards (Carlson et al. 2000). Among the instruments available, standards also provide the least incentive for innovation and adoption of low-abatement technologies. Because of the lack of static and dynamic efficiency, economists have argued for a long time in favor of moving away from standards in environmental regulation.

### 3.2.3 Pollution Taxes

Figure 3.1 indicates that the reason why firms do not internalize pollution damages is that they do not have to pay for the emissions. In other words, there is no price signal for the emissions that guides firm behavior, as in what happens with other market-traded goods. A “natural” solution is to provide such a price signal through what is known as a “Pigouvian tax.” In Figure 3.2, a tax $t$ on emissions that equals the difference between the social marginal cost $MC_{all}$ and the private marginal cost $MC_{self}$ at the socially optimal emission level $E_{al}$ can restore the first best. Now the firm’s marginal cost of emissions equals $MC_{self} + t$. When equating it with the MB curve, it will choose its emission at the socially optimal level. Even when the tax does not equal the wedge between the two marginal cost levels, Pigouvian tax is cost effective because every firm will choose emission so that its marginal abatement cost equals the tax level, implying that the marginal costs are equalized across all firms. Further, it can be shown that the tax policy also provides higher incentives than standards for firms to adopt new abatement technologies. In sum, taxes dominate standards because taxes are always cost effective, and can be both statically and dynamically efficient.
Emission taxes have another advantage if the tax revenue is used to reduce other distortionary taxes, leading to double dividends (Goulder 1995; Bovenberg 1999). Most taxes, such as income tax and sales tax, are distortionary and lead to misallocation of resources. Pollution tax is not distortionary since it provides the “correct” market price. If the tax revenue is used to (partly) satisfy the government’s budget needs, there is less need for other distortionary taxes for revenue purposes. The double dividend hypothesis argues that doing so is desirable because social welfare can be improved by having lower and fewer distortionary taxes.

One main difficulty facing pollution taxes is that emission levels should be measured relatively accurately for each individual polluter; this is not a trivial task in developing countries. Another difficulty arises from the public’s resistance to taxes and political economy considerations. The resistance varies across countries and depends on both the political landscape and the governance capacities. Tax revenues can also lead to corruption incentives. Nevertheless, pollution tax is often the policy instrument most widely promoted by economists.

### 3.2.4 Tradable Permits

Fundamentally the lack of price signals for emissions is because there is no market for emissions. As Montgomery (1972) suggested, such a
market can be created through the government issuing permits for emissions and the firms engaging in permit trading. In a tradable permit scheme, a firm's emission cannot exceed the number of permits it holds. The government first caps the total emissions by determining the total number of permits distributed to the firms, and then the firms can trade the permits in a market. The cap-and-trade approach has gradually gained popularity since the US launched the sulfur dioxide trading program in 1990 with the European carbon market, the European Union (EU) Emissions Trading System, being currently the largest permit trading market in the world.

Permit trading has several desirable features. First, it is always cost effective. Through permit trading, all firms face the same permit price, which they will equate with their own marginal abatement costs. The result is that the marginal costs are equalized across all firms. Second, permit trading can be efficient if the total number of permits equals the socially optimal pollution level, i.e., $E_{all}$ in Figure 3.1. Third, permit trading provides strong incentives for firms to adopt new abatement technologies—as strong as those provided by taxes under most conditions. Thus, it can achieve dynamic efficiency. Fourth, the cost effectiveness and efficiency features are independent of how the permits are allocated, e.g., through grandfathered or auctioned permits, or through permits evenly or unevenly distributed to the firms. This feature helps make permit trading more politically feasible as initial permit allocation can be used to gain political support.

 Tradable permits, as an example of market-based approaches, dominate emission standards in that they are both quantity policies, i.e., policies that restrict total emissions. However, tradable permits provide enough flexibility to achieve cost effectiveness while emission standards are much more rigid. As such, economists have promoted permit trading in many settings beyond pollution control. For example, tradable catch quotas have been promoted by economists and adopted by many fisheries as an effective way to limit fishing in a cost-effective way.

For tradable permits to work, two fundamental conditions are needed. One is the clear definition and protection of property rights in that firms must have permits to emit, and the permits are private properties. The second is that the market needs to be “thick” enough, i.e., the number of participants must be high enough so that the transaction costs of permit trading are sufficiently low. A thick market is needed for market makers to step in to make transactions. In the US, many local and regional water trading markets failed largely due to the limited number of participants (Garrick 2015).
3.2.5 Coase Bargaining

The government plays a central role in the three regulatory instruments discussed above. When polluters and victims are few, Coase (1960) argues that the government does not have to be directly involved in restricting emissions. Instead, all it needs to do is to define the property right structure clearly, namely, whether the polluters have the right to pollute or the victims have the right to a clean environment. The polluters and victims can bargain toward an efficient solution. For example, if the victims have the right to a clean environment, the firms can “bribe” or “compensate” the victims by offering them a price higher than the marginal damages from the pollution. In Figure 3.1, at any emission level lower than $E_{all}$, the marginal cost (or damage) suffered by the victims from emission is lower than the firm’s marginal benefit from emitting the pollutant. In this case, a price can always be found between the marginal cost and the marginal benefit, so that the victim is willing to be compensated at the price (since it exceeds the marginal damage of pollution to them), and the firm gains from being able to emit more by paying the price for its emission (since the price is lower than its marginal benefit from emission).

Although Coase’s argument is intuitive, its conclusion is sensitive to several implicit assumptions and simplifications, such as complete information about the marginal costs and benefits, low transaction costs in bargaining toward an efficient solution, and ambiguity about the bargaining setting. The large literature on bargaining has shown that the conclusion breaks down when information is asymmetric, in which case the outcome is sensitive to the bargaining institutions (Muthoo 1999). More importantly, the transaction costs can become prohibitively high if polluters and victims are many, which is usually the case in environmental regulation.

Nevertheless, when formal environmental regulation is insufficient and when pollution is concentrated in a small area, bargaining between the victims and polluters can reduce pollution. As we discuss later, this has happened in Asia, and can be a supplement to formal regulation.

3.2.6 Informal Regulation through Information and Voluntary Measures

In recent years, there has been increasing interest in studying nonregulatory approaches toward reducing pollution, with the literature focusing on two areas: (i) dissemination of firms’ environmental performance to the public, and (ii) firms voluntarily reducing their emissions. Information and voluntary programs started in developed
nations and gained popularity in recent years to supplement formal regulation, in many cases to encourage overcompliance with existing regulations (Koehler 2007; Morgenstern and Pizer 2007). Part of the reason behind the rise of these informal approaches is increased public awareness and concern about the environment that has translated to green market forces. It might thus pay to reduce emissions, as greener products might get higher prices and green practices can lead to stock market rewards. Conversely, heavy polluters will get punished in the marketplace. Figure 3.1 shows that these approaches effectively raise the private marginal cost of emissions, $MC_{self}$, so that firms find it optimal to reduce their emissions.

Public dissemination of firms’ environmental performance, such as their emissions, might work because public image matters in the marketplace and because the resulting public pressure might cause regulators to target heavy polluters for enforcement. Konar and Cohen (1997) find that mandatory disclosure of firms’ emissions through the Toxic Release Inventory in the US caused the stock values of some firms to decline after the release of information; and these firms subsequently reduced their emissions in response. Stock markets penalize heavy polluters when the information is made public either because investors expect market responses to the firms’ products and/or because of the threat of targeted enforcement. Bennear and Olmstead (2008), using data from water utilities in the US, show that requiring the utilities to disclose information on water quality violation to their customers is effective in reducing total violations, and this is especially true for larger utilities. Public utilities are regulated entities, and their response to information dissemination might be due to expectations of regulatory responses. In Asia, Arimura, Darnall, and Katayama (2011) show that environmental certificates such as ISO 14001 are effective in improving green supply chain management.

One benefit of information dissemination that has been less studied is that public information about firms’ emissions allows the victims to take more appropriate defensive measures (Evans et al. 2009). For this purpose, the most important piece of information is the ambient pollution levels rather than emissions from individual firms.

Firms have several incentives to undertake voluntary measures to reduce their emissions. Lyon and Maxwell (2008) list demand-side forces due to the public’s increasing awareness and valuation of green products and practices, supply-side forces due to product differentiation and potential cost savings, and public policy forces due to current and potential regulation. Ambec and Lanoie (2008) show that firms can gain from improving their environmental performance through several channels, including better access to more environmentally demanding markets,
being able to differentiate their products from others, gaining competitive advantages in pollution control technologies, and saving on materials and energies. Albertini (2013) reviews 52 studies over a 35-year period and finds a positive relationship between firms’ environmental performance and their financial performance—this relationship is influenced by the specific performance measures used. Earnhart (2018) reviews the empirical literature on the effects of environmental performance on firms’ financial performance and finds a positive relationship in general, but it is sensitive to how financial performance is measured. However, voluntary programs do not always create win-win situations for the environment and for the firms’ bottom-line. Fisher-Vanden and Thorburn (2011) find that firms suffered reduced stock returns after they joined Climate Leaders, a program of voluntary greenhouse emission reductions of the US Environmental Protection Agency.

3.3 Characteristics of Environmental Regulation in Asia’s Developing Countries

Although Asian countries are diverse in their environmental challenges and responses, many of these countries share some patterns. For important pollutants such as particulate matter, ozone, and nitrogen dioxide (NO₂), most of these countries have ambient quality standards that are less restrictive than but structurally like the WHO guidelines. Figure 3.3 shows a comparison of ambient standards of particulate matter (PM)_{2.5} and PM_{10} (Panel A), and of ozone and NO₂ (Panel B) for the WHO versus a sample of Asian countries. Most of the standards are less stringent than the WHO guidelines, and a certain degree of similarity exists in terms of the stringency of the standards across the countries. For example, several countries have adopted a 24-hour standard of 100 or 150 micrograms per cubic meter (µg/m³) of PM_{10}, in contrast to the WHO guideline of 50. Another common feature is that, despite the relatively lax standards, they are often violated by almost all these countries. Figure 3.4 shows the population weighted annual PM_{2.5} exposure from 1990 to 2017 for our sample countries. Except for Indonesia, Malaysia, and the Philippines, all other countries violate the standards, with India, the PRC, and Bangladesh far exceeding their standards.

A natural question, then, is how to govern the environment in these countries better. Most of the economic theory on environmental

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3. Not all countries have the same types of standards as the WHO guidelines. For example, Cambodia has standards on total suspended particulates, but not on PM_{2.5} or PM_{10}.
regulation grew out of experiences in developed countries. This is reflected in the fact that the bulk of the theory is devoted to studying instruments that are either socially efficient or cost effective, with the implicit assumption of adequate capacity for monitoring and enforcement, benevolent social planners, and efficient markets, with environmental externalities being the only market failure. Although these assumptions seem ideal even for developed countries, they are often violated to a much larger degree in developing countries. Below we discuss the main departures from these assumptions in developing countries; they will form the basis for understanding and guiding Asia-specific environmental policies.

Figure 3.3: Ambient Air Quality Standards of Selected Asian Countries vs. World Health Organization Guidelines

Panel A: Ambient standards on particulate matter

Panel B: Ambient standards on ozone and NO₂

PRC = People’s Republic of China, NO₂ = nitrogen oxide, PM = particulate matter, WHO = World Health Organization.
Sources: Joss et al. (2017), CAI–Asia Center (2010a, 2010b), and UNEP (2019).
Limited capacity for monitoring and enforcement. Perhaps the biggest challenge facing developing countries is insufficient capacity for monitoring regional ambient concentrations of major pollutants, especially of emissions of individual firms. Accurately monitoring ambient concentrations is the first step in environmental regulation because environmental quality targets of national policies are often expressed in ambient concentrations of major pollutants. For some air pollutants such as particulate matter, satellite data are increasingly available to help monitor ambient concentrations. However, for most other pollutants, adequate monitoring requires locally installed equipment and professionals to operate it and collect and disseminate data. A major achievement of the US Environmental Protection Agency since its creation in 1970 is a well-functioning nationwide air quality monitoring system, which forms the backbone for the National
Ambient Air Quality Standards. However, adequate monitoring of ambient pollution is far from being sufficient for effective regulation, which requires monitoring of firm emissions. Without firm-level data, point source pollution is turned into nonpoint source pollution, and it is well known that nonpoint source pollution is difficult to control because it is difficult to hold individual polluters responsible for their emissions (Shortle and Horan 2001). For nonpoint sources, either firms are not regulated, or regulation takes the form of rigid standards on observable activities such as equipment and input use. For example, small-scale polluters in the PRC are often forced to shut down instead of being required to pay emission fees as large firms do (Ma and Ortolano 2000).

Developing countries also tend to lack adequate capacity for enforcement, which is closely related to monitoring. Enforcement is much more than having the personnel and funding to support environmental agencies; it involves governance structures, legal systems for both civil and criminal cases, and incentives provided for enforcers. In the PRC, for example, local environmental protection agencies report both vertically to the Ministry of Ecology and Environment (of the central government) and horizontally to local governments, resulting in compromises in enforcement efforts when local governments prioritize economic growth. Many developing countries lack adequate legal systems for citizens to bring civil or criminal lawsuits against polluters. Furthermore, corruption often undermines the incentives of government officials to enforce environmental laws and regulations.

**Environmental protection and economic growth.** Economic development and growth are central concerns of developing countries. Since the 1980s, many of Asia’s developing countries have experienced periods of rapid economic growth, often at the cost of the environment. Although economists often argue against the “pollute first and clean up later” approach, no firmly established theory on the optimal pollution path is applicable to all developing countries. The environmental Kuznets curve (EKC), as an empirical summary of the past experiences of many countries, predicts that environmental quality falls first and rises later as a nation grows its economy (Dasgupta et al. 2002). However, the EKC does not offer guidance on effective regulatory approaches and on how growth and environmental protection should be related to each other.

An often-neglected aspect in the growth–environment debate is that environmental regulation is more likely to hurt economic growth if it is not efficient or cost effective. A case in point is the backlash against government efforts to reduce coal used for heating in northern parts of the PRC during the 2017–2018 winter season (Bradsher 2017). To meet air quality goals, some local governments took extreme CAC measures to shut down many factories and coal-based heating devices for schools.
and families. Although effective in improving air quality in the short term, such regulatory approaches can contribute to the argument that environmental regulation is economically costly, possibly leading to less public and private effort to reduce pollution in the long run. It is precisely because countries are concerned about economic growth that they should be careful to design and implement environmental regulation that is at least cost effective. Economic growth should be an argument for market-based regulatory approaches, rather than an argument against pollution control.

**Market and regulatory distortions.** No market is perfectly competitive and environmental regulation always coexists with other types of regulations. There is a sizable literature on environmental regulation when the output market or the permit market is characterized by market power (Kennedy 1994; Requate 2006). Most of the findings are concerned with adjusting the levels of the policies, such as the tax level or the distribution of permits to reduce the distortions caused by imperfect competition. Although there is no consensus, imperfect competition does not seem to favor one type of policy instrument over another. These observations also apply to environmental regulation facing other distortionary government policies. If the government subsidizes industrial growth that causes pollution, any regulation that tries to reduce the pollution will interact with the subsidy policy. When multiple distortions coexist, it might be desirable to use a mixture of policies to correct the environmental externalities, with some policies directly targeting pollution while others deal with the interactions with other policies (Bennear and Stavins 2007; Lehmann and Gawel 2013). For example, Wang, Zhao, and Bhattacharya (2015) consider an economy with both pollution externalities and a lack of adequate health insurance—this combination applies to many Asian developing countries. Economic growth causes pollution, which in turn raises health risks. Without adequate health insurance, individuals make precautionary savings to prepare for an increased likelihood of being sick at an older age. The higher savings rate leads to more economic growth and more pollution, leading to a vicious cycle of growth–pollution–savings–growth. The authors find that the optimal intervention needs include a combination of environmental, health, and social redistribution policies.

**Top-down versus bottom-up approaches.** Most Asian countries have government-centered environmental institutions. Governments play much larger roles than nongovernment organizations and the civil society in environmental protection, and environmental activism and consumers’ environmental attitudes play relatively limited roles. These observations are especially true in East Asia (Shin 2015). As a result,
formal regulation tends to be more important in Asia than in other parts of the world. For informal regulation to be more effective, Asian countries need to first improve their formal regulation, including its enforcement. The top-down approach also implies that regulatory tools such as pollution taxes might have an easier time being adopted than others in countries such as the US. There should be reasons other than political feasibility, such as economic efficiency and tax burden of firms, to justify the use of cap-and-trade instead of pollution taxes. The debate on double dividends, especially the argument for recycling pollution tax revenues to reduce other distortionary taxes, becomes especially relevant and important for Asian countries.

### 3.4 Lessons for Environmental Regulation in Asia’s Developing Economies

In this section, I discuss the findings in the economics literature on environmental regulation that are relevant for developing Asian countries. I have drawn mostly from the theoretical literature in the previous sections but, in this section, I will emphasize the empirical literature. Several papers review environmental policies for specific countries and regions. Ma and Ortolano (2000) comprehensively review the PRC’s environmental policies during the early stages of their development; many of the structures discussed remain valid today. He et al. (2012) review the history of environmental regulations in the PRC and show that although the majority of environmental regulations are CAC policies, there is increasing adoption of market-based approaches such as green credits, emission charges, emissions trading, ecological compensation, and voluntary and information approaches. Auffhammer and Gong (2015) review the PRC's regulation of carbon emissions, particularly its experiments with the regional carbon markets. Divan and Rosencranz (2001) review environmental laws and regulations in India. Tan (2004) briefly reviews the environmental laws in Southeast Asia. O'Connor (1999) reviews the adoption of various forms of regulatory instruments in many countries, including Asia.

#### 3.4.1 Regulatory Process and Capacity Building

Often the first step in environmental regulation is to set the environmental quality targets for a country and for various regions. Such ambient standards specify the target levels of environmental quality, often expressed as maximum intensities or concentrations of certain
pollutants. For example, almost every country has national or regional ambient air quality and water quality standards. Economic theory argues that, as shown in Figure 3.1, such standards should be set to balance the benefits of a clean environment against the costs of reducing pollution. However, fully implementing this criterion of maximizing social welfare requires information on the social benefit and cost functions, which is difficult to obtain. Instead, protection of human health has often been used as the single most important criterion in setting such goals. The standards set by the WHO offer the most commonly used benchmark, and typically developing countries set their own ambient standards at levels similar to or lower than WHO levels. For example, the 2005 WHO air quality guideline value for ozone is 100 $\mu g/m^3$ (8-hour mean). However, the air quality standard in the PRC’s urban areas is 160 $\mu g/m^3$ (8-hour mean), while that for the Republic of Korea (henceforth, Korea) is 128 and for Thailand, 140. Similarly, the WHO guideline value for PM$_{2.5}$ is 25 $\mu g/m^3$ (24-hour mean) but the standards are 75 for the PRC and 50 for Korea and Thailand. These laxer ambient standards are often violated, showcasing the severity of Asia’s pollution problems.

The next step is to make and enforce laws and regulations to meet the ambient environmental quality standards. Ultimately, environmental quality depends on the degree to which polluters are incentivized to reduce their emissions, so the law and regulations should target polluter behavior. A key requirement for successful enforcement is observability: the target behavior should be observable, measurable, and verifiable, either directly or indirectly. This is where capacity building is most needed to be able to measure important behavior and to enforce regulations based on the measured behavior. Increasingly, citizens are involved in monitoring and reporting heavy pollution and environmental accidents, mostly on local scales (Martens 2006). Citizen participation can also be important in pushing actions by both government agencies and polluters.

Although specific regulations might be different, the regulatory processes are similar in both developed and developing countries. For example, the PRC’s environmental regulations, including the types and levels of environmental quality standards, are like those in the US and the EU. In this regard, developing countries can learn from developed nations regarding regulatory capacity building. For transboundary pollution, harmonized monitoring methods also serve to help form and enforce international environmental agreements. For example, international climate negotiations have always included the stipulation of helping developing countries build up their capacities in monitoring and accounting for greenhouse gas emissions.
3.4.2 Lessons About Formal Regulation

Many important lessons have been learned from the empirical literature on environmental regulation. Before we discuss the specific lessons learned about the specific regulatory instruments, we discuss some general guidelines.

First, cost effectiveness is a more reasonable goal than social efficiency in guiding regulatory choices, but ultimately efficiency should be the grand goal. To achieve efficiency, more information is needed to estimate the economic damages from pollution, including health damages and productivity losses. The economics profession has developed sophisticated nonmarket valuation methods to estimate both the use value and the existence value of a clean environment, and applications of these methods are increasing in developing countries, especially in the PRC and India. The bulk of the empirical nonmarket valuation literature values pollution and the environment in developed countries, especially in the US and the EU. Benefit transfer methods have been developed to rely on estimates in one place (e.g., developed countries) to value the environment in other places (e.g., developing countries), but the methodology requires at least certain valuation studies in the latter (Richardson et al. 2015).

Second, it is important to assess the performance of environmental regulations regularly to make improvements and provide insights for future regulatory choices. The assessment should include not only the environmental effects but also the social and economic effects of environmental policies. Such assessments will help promote cost-effective policies over CAC approaches, especially when the economic costs are included. They also help address the question of whether environmental protection must come at the cost of economic growth.

Third, regulators themselves should be incentivized to enforce environmental laws and regulations effectively. Several studies have found that in the PRC, providing incentives to government officials through evaluation and promotion in the bureaucratic systems is effective in incentivizing these officials to enforce environmental regulations (Zheng and Kahn 2013; Zheng et al. 2014; Kahn et al. 2015; Liang and Langbein 2015; Chen et al. 2018; Lin et al. 2019). Specifically, the Government of the PRC included water quality targets in the annual evaluation of provincial governors. These studies found that the scheme was effective in reducing water pollution, and the reduction was higher when the governors were younger and thus had more room to be promoted.

Pollution taxes. Taxes in general work better than standards in terms of cost effectiveness, but they might not be able to bring environmental
quality improvements as quickly as standards unless they are set at sufficiently high levels and are adequately enforced. The purpose of pollution taxes is to provide price signals about emissions. Therefore, it is important that the tax burden of a firm is tied to its emission level. This has not always been the case in developing countries. For example, the PRC's emission fee system, which is only now being replaced by an emission tax system, was only partially linked to a firm's actual emissions, as the total amount paid by a firm was negotiated between the firm and the local environmental agency and was capped by a ceiling (Ma and Ortolano 2000). Further, the tax revenue should be used to reduce other distortionary taxes the firms pay. This aspect is important in making pollution taxes more effective and more politically feasible, but has received insufficient attention in actual policy making.

** Tradable emission permits.** Despite the increasing adoption of tradable permits, they can require more institutional capacity than taxes to work properly. For example, the initial distribution of permits is often tied to firms’ historical emissions, for which data might not be available. Firms might not be experienced in trading permits, and the market must be sufficiently thick for market makers to come in and facilitate trading. Price fluctuations can add additional uncertainties to firms and can lead to arbitrage opportunities. A lack of information about firms' abatement costs might lead to too many or too few permits being issued, leading to prices that are too high or too low. Price collars can be included in the system but the levels of the price ceiling and floor can be arbitrary. The main lesson is that permit trading is not necessarily the best regulatory approach. It can be dominated by a well-crafted tax system, both in terms of efficiency and implementation. The upside is that developed nations are now much more experienced in operating permit markets, and developing countries can build on their experiences. In the PRC's ongoing effort to establish a nationwide carbon market, it has learned from international experiences and from its own experimentation with several regional markets.

**Command-and-control policies.** CAC policies such as standards can bring concrete improvements in environmental quality, but often at high costs. The PRC has adopted many authoritarian measures, such as shutting down polluters to meet environmental quality targets. Zhu et al. (2015) find that one particular measure—freezing environmental impact assessment on construction and investment projects (effectively stopping such projects from being approved) in regions that fail to meet regional environmental targets—worked in improving environmental quality, but it lacked legal foundations and can be costly. The main lesson about CAC policies is that they should be as flexible as possible. If a quantity policy is needed, i.e., to restrict the total amount of emissions
such as in the case of highly damaging pollutants like carcinogens, a tradable permit system works better than standards uniformly imposed on all polluters. If standards must be used, they should be designed to target larger entities or areas, while leaving room for the smaller entities to choose their responses. For example, instead of restricting emissions from each polluting facility, a “bubble” can be imposed on a larger firm that operates multiple units.

**Imperfect enforcement.** Despite the large theory literature on environmental regulation with imperfect enforcement (Malik 1990; Livernois and McKenna 1999; Montero 2002; Stranlund et al. 2009; Stranlund and Moffitt 2014; Oestreich 2017), the empirical literature on imperfect enforcement in developing countries is rather thin. For example, there is no empirically derived guidance on how to balance the probability of inspection and the magnitude of fines imposed on violators. Firms in developing countries, especially small-scale firms, may have limited financial resources and thus face tighter bankruptcy constraints. For these firms, Earnhart and Segerson (2012) find that increased enforcement can lead to increased pollution because the penalties faced by firms are limited by the bankruptcy constraint. This has occurred in the PRC, where recent environmental campaigns have forced some firms to close or go out of business. The literature has also studied reasons for which firms might comply with regulations under weak enforcement. Earnhart, Khanna, and Lyon (2014) find that foreign ownership and information disclosure programs can help improve environmental performance. Dasgupta, Hettige, and Wheeler (2000) find that in Mexico, firms’ compliance with environmental regulation is significantly affected by environmental management of the firms, implying that compliance is likely to increase if firm managers receive environmental training. Thus, countries with limited enforcement capacities should try to design education and information programs to increase the willingness of firms to comply with regulations.

### 3.4.3 Lessons about Informal Regulation

Due partly to the imperfection in formal regulation, there is increasing interest in informal regulation of developing countries. Informal regulation is important in filling the gaps before a system of formal regulation is established and enforced, and as complementary to formal regulation in achieving or overachieving the regulatory goals. But does informal regulation work? Is it a substitute for or complement to formal regulation? The literature has found different answers to these questions. For example, Zhang, Mol, and He (2016) argue that the PRC has significantly increased its information disclosure and environmental
transparency in response to increasing pollution, but caution that the environmental effects of these measures are still not clear. Below I discuss several findings, mostly confirming that informal approaches do work. However, they need formal regulation to back them up and should not be used as a substitute for formal regulation.

**Informal regulation can improve the environment in developing countries.** There is a rich literature finding that informal regulation can work to reduce pollution. For example, Khanna and Liao (2014) review the literature on voluntary and information approaches in both developed and developing nations, including the case of ISO 14001 certification, and find that these informal approaches can be effective in many developing countries. Blackman, Afsah, and Ratunanda (2004) and García, Sterner, and Afsah (2007) show that the Program for Pollution Control Evaluation and Rating, a public exposure program in Indonesia similar to the US Toxics Release Inventory, was effective in reducing firm emissions in the short and medium terms, especially among firms with poor compliance records before the program. Similarly, Wang et al. (2004) find that public rating of the environmental performance of firms in the PRC was effective in raising tier environmental performance. Hettige et al. (1996) use data from 1992 to 1994 in South and Southeast Asia to show that, despite the lack of strong formal regulation, many firms have adopted clean production practices, driven by new production technologies, community actions, and sometimes public ownership. Kathuria (2007), using water pollution data in the state of Gujarat in India, finds that the press has served as informal regulation and has been effective in some cases in reducing emissions. Powers et al. (2011) demonstrate that India’s Green Rating Project, a program that discloses information about firms’ environmental performance, reduced the pollution loads of firms, especially those that are heavy polluters and those located in wealthier communities. The firms’ behavior might have been driven by market responses. Dasgupta, Laplante, and Mamingi (2001), using data from Argentina, Chile, Mexico, and the Philippines, find that capital markets react to the announcement of major environmental events related to polluting firms.

There is some evidence that these alternative approaches might work even better than formal regulation if the formal regulation is not strictly enforced. Dasgupta et al. (2001), using data from the PRC, find that back then, inspections of polluting firms were more effective than pollution charges in reducing firms’ emissions. Zhang et al. (2008) show that community pressures and the market’s reward for improved environmental performance have played increasing roles in reducing pollution in the PRC, relative to formal regulation. In fact, they find
evidence that some firms have overcomplied with environmental standards, but only if these other incentives are strong.

**Informal regulation should not be a substitute for formal regulation.** A caution about voluntary approaches in developing countries is that they should not replace formal regulation or their effective enforcement. Voluntary programs can work in developing countries but they should be complemented by strong regulatory measures that are sufficiently enforced (Blackman 2008). Without formal regulation as backup or as a threat, it is difficult for the voluntary programs to succeed by themselves. For example, Blackman et al. (2010) show that, although the Clean Industry Program in Mexico attracted polluting firms to participate, the program did not have long-lasting effects in reducing the participants’ emissions after they graduated from the program (and obtained clean certificates). Talukdar and Meisner (2001) find that well-functioning domestic capital markets are associated with reduced environmental degradation, and Tamazian and Rao (2010) find that financial markets must be accompanied by complementary institutions (such as institutions in environmental regulation) to improve firms’ environmental performance.

Partly because there are not always strongly enforced formal regulations to back them up, informal regulation does not always work. Blackman (2010) summarizes 30 studies on alternative (nongovernment) pollution control policies in developing countries, including community pressure, public disclosure of emissions such as performance evaluation and rating programs, and voluntary approaches. He finds that overall the literature does not provide strong evidence that these alternative policies worked in significantly improving environmental quality. Failures occur when firms are not properly incentivized by the market or by threats of future regulation. In other words, firms must be incentivized to reduce emissions, and these alternative policies will need the threat of formal policies or a well-functioning market that fully internalizes public pressure and perceptions. The latter is particularly difficult when causality is hard to establish and when stock markets do not fully incorporate market information.

**Informal regulation should complement formal regulation.** A natural conclusion from the above discussions is that informal approaches can and should complement formal regulation. With formal regulation in place and enforced, properly designed informal approaches can encourage firms to comply with formal regulation. For example, using data from the paper and pulp industry in British Columbia, Canada, Foulon et al. (2002) show that tightening standards and publishing a list of firms violating the standards worked complementarily to improve firms’ environmental performance.
Similarly, McGuire (2014) shows that ISO 14001 certification increased firms’ compliance with environmental regulations in the PRC. In these cases, firms reduced pollution not to improve environmental quality per se. Instead, information dissemination provided incentives for them to comply with existing environmental regulations.

**Firms may or may not benefit from voluntary emission reduction.** Despite the widespread evidence that firms sometimes undertake voluntary activities to reduce their emissions, it is not clear that they profit from doing so. For example, Lyon et al. (2013), using data from 2008 to 2011 in the PRC, find that firms that won the Green Company Award did not gain in terms of shareholder values. Some of them, especially those in low-pollution industries and private firms, saw reductions in shareholder values. This observation further supports the argument that voluntary actions arise not only from increased public concern about the environment but also from the threat of enforcement and future regulation. Firms will benefit from voluntary actions if the market anticipates payoff from such actions, which comes from both increased consumer awareness and tougher regulation. The environmental attitudes of firms’ management can also play a role beyond profit maximization. Nakamura, Takahashi, and Vertinsky (2001) find that for Japanese firms, the incentives to incorporate environmental goals and to obtain ISO 14001 certification arise from both profit maximization and the environmental attitudes and values of managers.

**Coase bargaining and citizen action.** As pollution increases, there has been an increasing number of cases in many developing countries where the victims have taken actions to confront the polluters directly. In the PRC, this type of case has arisen more frequently, resulting in what has been called “regulatory pluralism” (van Rooij et al. 2016). Pargal and Wheeler (1996) discuss a case of water pollution in Indonesia and show that, similar to the predictions of bargaining theory, the outcome depends on the relative bargaining power of the community (victims) and the firms (polluters). Citizen action is a special case of Coase bargaining, which has not been used much in practical environmental regulation. However, for developing countries that lack formal regulation, Coase bargaining can help reduce pollution, although most likely at high transaction costs.

### 3.5 Concluding Remarks

Many Asian countries are experiencing severe environmental pollution and need effective environmental regulation. Decades of theoretical and empirical research in environmental economics have generated several important and useful lessons for environmental policy making. Some
of the lessons are derived from the experiences of developed countries, but some lessons are particularly relevant for developing countries. In this chapter, I discussed the theoretical underpinnings and empirical regularities of both formal and informal regulation and summarized several useful lessons for practical policy making.

Several lessons are particularly important. First, the foundation for both formal and informal regulation is sufficient capacity for monitoring and enforcement, with adequate information on both ambient pollution at regional and local levels and emissions of firms. Effective environmental regulation requires teams of professionals in addition to politicians, and all of them need to be properly incentivized. Second, a realistic criterion in assessing regulations is cost effectiveness, but nonmarket valuation studies on important pollutants can go a long way toward making regulation socially optimal. Third, for many developing countries, tax might outperform tradable permits but the key challenge is to use the tax revenue properly. There is plenty of room for generating double dividends by recycling the tax revenue to reduce other distortionary taxes. Finally, informal regulation can improve the environmental quality in developing countries but it should be a complement to rather than a substitute for formal regulation.

I end the chapter by highlighting several areas where I believe future work is needed. While the empirical literature includes many papers on developing countries, especially on the PRC and India, the theoretical literature is extremely sparse on environmental regulation in developing countries. Few of the theory papers adequately account for the institutional and economic peculiarities of developing countries, and the environmental economics profession has a long way to go in coming up with theories of “envirodevonomics.” Part of the reason for this is that empirical research on developing countries is restricted and mostly driven by data availability rather than by policy and societal needs. As such, their conclusions, while important, may not provide much-needed empirical regularities that would form the basis for useful theories. Sometimes data are simply not available due to insufficient capacity, but often environmental and health data are guarded and not made public. International environmental negotiations have made headway in this regard, but ultimately it is up to governments of developing countries to make data policies amenable to relevant empirical research.
References


4 Environmental Governance and Environmental Performance

Chun-Ping Chang, Minyi Dong, and Jiliang Liu

4.1 Introduction

Human demand on raw materials has continuously increased since the beginning of the Industrial Revolution, leading to an inevitable depletion of natural resources. As Karl Marx (1887) stated in his book *Capital*, the entire development of both civilization and industry has always been destructive to the forests and, by contrast, the effect of cultivation and production on the forests can be considered negligible. The environmental problem sourced from this predicament has gained momentum and finally resulted in a series of environmental pollution incidents. For instance, the most notorious air pollution incident, the Great Smog, happened in London in the winter of 1952. Large amounts of soot, dust, ash, and exhaust accumulated at that time and hung like a vast pall over London; the pollutant load of noxious gases, such as sulfur dioxide ($\text{SO}_2$) and suspended particulate matter, surged to 300% of their normal value; and about 12,000 people died of respiratory disease. The environmental problem has obviously become a conspicuous and widespread problem in the world today, and its ever-intensifying negative effect not only severely hinders global economic growth but also damages human health and destroys the ecosystem.

The concept of environmental protection did not step into the limelight until 1962, when Rachel Carson (2002), a famous American marine biologist and author, published *Silent Spring*. This book illustrated the environmental contamination caused by a highly toxic pesticide named dichlorodiphenyltrichloroethane or DDT. As a result of the warnings in the book, the United States (US) government launched an investigation targeting highly toxic pesticides. Subsequently, the government established the Environmental Protection Agency to take charge of enforcing federal laws involving the environment; the states
sequentially formulated relevant policies and laws on forbidding the production and application of highly toxic pesticides. Thus, this book is of epoch-making significance against the social background of conquering nature at that time. In addition, the environmental measures taken by the US government triggered chain reactions among western European nations, promoting the development of large-scale civil environmental movements. There appeared many nongovernment environmental research institutions and community organizations, represented by the Club of Rome, established in 1968, and Greenpeace, established in 1971. These nongovernment organizations (NGOs) used their increasing power to pressure the national governments to take corresponding actions and resolve environmental issues. People in increasing numbers came to realize that it is in everybody’s self-interest to deal with environmental problems. The public started criticizing enterprises that stubbornly chase high profits at the expense of environmental pollution and the apathetic governments that take a position of willful blindness toward environmental deterioration. In 1969, the United States government struggled to promulgate the National Environmental Policy Act and formulated a corresponding environmental governance system. That was when the national governments’ function of environmental governance was put on the agenda.

On 5 June 1972, the United Nations (UN) held the first session of the UN Conference on the Human Environment in Stockholm, Sweden. The well-known Declaration of the United Nations Conference on the Human Environment was proposed at the meeting, which indicated that environmental protection had formally gained increasing awareness among countries all over the world. Environmental pollution has taken its toll on people and forced them to suffer heavy losses due to their destructive exploitation of natural resources, and humans have begun to rethink their position in the meantime. In fact, as most developing countries and emerging industrializing countries have experienced during their economic takeoff phases, environmental deterioration and irreversible resource depletion have already become among the most critical challenges stifling long-term sustainable economic development and endangering fundamental social stability.

Looking at the evolution of international environmental protection, the development of environmental protection awareness can be divided into the following three stages. The first stage ranges from the early 1960s to the 1970s, that is, from the time that human society started to realize the existence of the environmental pollution problem to the time that national governments implemented relevant environmental policies. During this period, the state and the government regarded the environmental problem simply as a sort of technical problem
and tried to resolve it through pollution control. The second stage of environmental governance covers the early 1980s to the early 1990s, during which the authorities began to recognize the interrelationship between the economy and the environment and started to apply economic stimulation as a major measure of environmental governance. In the third stage of environmental governance and protection, most national governments saw the environmental problem as a sort of development problem and tried to enact efficient policies to coordinate the relationship between environmental protection and economic development. The state also intended to decentralize its decision-making authority in environmental policies to various stakeholders, institutions, and organizations. We believe that political institutions are involved in every stage of environmental governance and play an “unneglectable” role.

Focusing principally on the relationship between environmental governance and environmental performance in Asia, Section 2 looks at the history of environmental governance and presents a literature review on the relationship between government expenditure and environmental performance. Section 3 briefly reviews the environmental conditions in Asia and analyzes the evolution of environmental governance in representative Asian countries. Section 4 discusses the data and empirical results and Section 5 concludes.

4.2 Basic Description of Environmental Governance

This section first introduces the basic definition of environmental governance, then briefly looks through the whole evolution of environmental governance since the 1960s. It finally reviews the relevant literature on the relationship between environmental performance and environmental governance, which can be proxied by government expenditure on environmental protection.

4.2.1 Definition of Environmental Governance

Objectively speaking, environmental pollution is closely tied to humans’ production activities and living behaviors, and human activity inevitably results in environmental degradation. Long before the Industrial Revolution, the impact of human activity on the ecological environment was limited and partial, and most pollution issues were still within the range that could be resolved and accommodated by the environment itself. Therefore, the self-adjustment function of
the environment acted as the main measure for solving the problem of environmental pollution, while anthropogenic governance and protection only served as assistance. Equipped with advanced machines and developed technologies, human society has enjoyed unprecedented galloping progress since the beginning of the Industrial Revolution. Nevertheless, massive environmental pollution problems have occurred at the same time, and the adverse impact of human activity on the ecological environment has been deepening and spreading over the whole world in the last few decades. The increasingly acute conflict between the environment and human society has gone beyond the capacity of the environment, and further endangered the existence and development of mankind.

Following economic theory, the law of value is an essential element in governing human behavior, and people are likely to pursue the maximization of benefits, that is, people prefer to generate greater profit at lower cost. Moreover, people will not subjectively or proactively consider the negative externalities of their behaviors, indicating that enterprises tend to dump waste materials directly into the environment because of the low treatment costs. That was the main reason for calling on the state government to take responsibility for environmental governance in the early 1960s. In recent years, environmental governance has become a rapidly growing field in both academia and business due to its significant implications for conservation practice. Paavola (2007) thinks that a broad and deep understanding of environmental governance is necessary for handling conflicts over environmental resources.

Against the background of global environmental change, environmental conservation is facing complex, nonlinear, and cross-scale challenges (Rockström et al. 2009); and it is of great importance to have a clear recognition of environmental governance (Chapin et al. 2010). Armitage, de Loë, and Plummer (2012) distinguish environmental management from environmental governance, where the former consists of a set of operational decisions aimed at achieving certain conservation outcomes while the latter covers a wider range of both responsibilities and actors. Oakerson (1992) proposes that environmental governance should involve broad processes through which societies make decisions related to the environment or may exert certain impacts on the environment. Lemos and Agrawal (2006) define environmental governance as a set of regulatory processes, mechanisms, and organizations through which political actors resolve environmental problems and impact environmental outcomes. They further emphasize that governance is not a synonym for government. Environmental governance, especially in today’s world, is a much broader concept
covering the environment-related actions of the state and other actors, such as businesses, communities, and NGOs.

### 4.2.2 The Evolution of Environmental Governance

The ecological environment possesses characteristics of non-rivalrous consumption and “non-excludability,” which are the typical features of public goods. Environmental pollution is an inevitable by-product of human economic and social activities. Environmental governance is thus an essential function and responsibility of both national and local governments. Looking back at and reviewing the history of environmental governance reveal that it has experienced three main eras: (i) the centralized “command-and-control” (CAC) regulation, (ii) the intervention of a market-oriented economic approach, and (iii) the hybrid partnerships among the state and other actors.

The first generation of environmental governance was dominated by the national government with its mandatory policy measures, and this phase roughly ran from the 1960s until the end of the 1970s. Western countries, including particularly the US, attempted to resolve environmental problems by formulating relevant environmental laws and regulations, which forced environmental polluters to internalize their environmental costs. The national governments designed, formulated, and implemented relevant laws and regulations according to specific environmental standards, and enterprises were forced to adopt clean technologies to control pollutant emissions in accordance with corresponding emission standards. For example, the US government promulgated the Clean Air Act in 1963, which marked its first attempt to adopt a CAC approach to resolving the pollution problem. The Clean Air Act proposed strict air quality standards for six kinds of air pollutants in accordance with the standards set by the Environmental Protection Agency, and said legislation required states and territories to formulate specific solutions to meet the standards. The CAC approach made extraordinary contributions to environmental protection during the 1960s and 1970s. This mandatory policy instrument was frequently used by developed countries, such as the US, Canada, Japan, and some western European countries, in the early stage of environmental governance. The conventional CAC regulation can improve environmental protection performance

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1 “Non-rivalrous” indicates that its consumption by individuals does not reduce the amount that is available to be consumed by other individuals, and “non-excludability” refers to the fact that no individual can be effectively excluded from consuming it (Gravelle and Rees 1992).
in a relatively short period and has a remarkable promoting effect on the environment, yet its deficiency is obvious as well. The CAC approach is generally regarded as a short-term emergency measure and has been criticized for its lack of flexibility and the huge cost hiding behind the implementation process. The success of such regulation is at the expense of large social and environmental costs, such as social costs related to resolving the problem of compliance and enforcement, and environmental costs in regulating and provisioning ecosystem services (Armitage et al. 2012; MEA 2005). Moreover, the effective implementation of CAC regulation requires a higher level of government regulation, which may indirectly increase the government's administrative expenses.

After the initial stage of the successful construction of the state-dominated environmental governance pattern in the 1960s and 1970s, more and more stakeholders began to criticize the failure of government to prevent environmental risks and resolve environmental degradation (Jänicke 1986; Bulkeley and Mol 2003). Just like market failure in the business field, there also exists government failure in the governance process, and people are gradually realizing that government alone may fail to resolve the environmental problem. Moreover, corporations' abilities to respond to environmental governance are relatively weak at the beginning of CAC regulation. Faced with severe environmental problems and enormous social pressure, most corporations can only be regulated passively and they struggle to meet the emission standards at high costs. Nevertheless, corporations have gained more and more speaking rights in environmental governance over time; they have made alliances against the government, requesting the government to consider the huge cost of pollutant treatment and emission control. The tremendous and increasing environmental costs also exert a certain negative impact on the competitiveness and economic development of a country, which has increasingly aroused public concern and brought about considerable pressure on government regulation. In the mid-1980s, the environmental governance pattern changed in accordance with the market economic system, and market mechanism measures were brought into the process of environmental protection, which is the most striking feature of second-generation environmental governance. This kind of market-oriented approach is called “environmental–economic means” and includes a series of measures, such as environmental fees, taxes, subsidies, preferential credit, and a differential tax rate, among which emission trading is one of the most effective environmental–economic policy instruments. Emission trading is a form of “cap and trade” system. Generally speaking, an overall limit (or ceiling) on pollutant emissions is set by the state
(government or other authorities) since regulated polluters must hold a permit that is equal to or higher than the amount of pollutants actually emitted during a certain period. The participants can thus sell or buy the emission permit according to their real demand. For example, a participant could choose to adopt clean technology to reduce its own emissions and sell the excess permits to other polluters. This approach successfully redistributes pollutant emissions by establishing the legal right to trade emission permits and contributes to achieving the goal of total emission control. Compared with the conventional CAC approach, the environmental–economic means basically possesses the following advantages: (i) it is more cost-effective due to its flexibility in terms of implementation, which allows polluters and stakeholders to make the choice that maximizes their own interests and benefits; (ii) it provides certain incentives for the innovation of clean technology and environmental management, mainly because its potential profit mechanism leads people to believe that environmental protection can be profitable; and (iii) since environmental–economic refers to source control mechanisms of environmental governance, it favors the prevention of environmental pollution and degradation.

In the 1990s, environmental governance gradually stepped into its third generation. As stated by Armitage, de Loë, and Plummer (2012), government is not the most significant source of decision-making authority on environmental issues, at least in today’s complex environmental conditions. Stewart (2001) also argues that the optimal structure of environmental governance develops over time, and environmental protection strategies must change with the change of circumstances (Esty 2001). More and more new actors are participating in the decision-making on environmental protection, such as private actors and non-state organizations. Bache and Flinders (2004) propose that multilevel governance is also an important aspect of environmental governance, and non-state actors ought to play a crucial role in the decision-making process at various governance levels, which further emphasizes the significance of stronger decentralization in policy formulation and implementation (Jordan 1999; Papadopoulos 2007; Kluvánková-Oravská et al. 2009). Lemos and Agrawal (2006) conclude that cross-scale governance, market instruments, and individual incentives are the most important themes of environmental governance. Currently, the involvement of stakeholders, businesses, institutions, markets, the public, and NGOs in policy making is the main idea and theme of environmental governance (Bulkeley and Mol 2003); and hybrid partnerships among various actors in decision-making have become the most obvious feature of the third generation of environmental governance.
4.2.3 Literature Review on the Relationship between Environmental Governance and Environmental Performance

As explained before, environmental protection is generally regarded as a public good, which is widely the responsibility of the government. Another strand of research links environmental performance with environmental governance, proxied by government expenditure on environmental protection.

López and Galinato (2007) classify government expenditure into two categories: (i) expenditure on public goods, and (ii) expenditure on private goods. The former includes expenditure on pure public goods as well as expenditure to mitigate the impact of market failure, while the latter refers to expenditure that cannot be justified on these grounds. For example, expenditure on public goods includes environmental protection, health and social transfers, research and development, and subsidies to households through education. Expenditure on private goods includes subsidies on energy consumption, fossil fuel production, and government grants to corporations. Pearce and Palmer (2001) propose that government expenditure on environmental protection generates certain improvements for social welfare. The European Commission (2012) states in its Report on Public Finances in EMU that increasing environmental protection expenditure contributes to dealing with market failures related to negative environmental externalities. Antweiler, Copeland, and Taylor (2001) argue that the reallocation of government expenditure on public goods and private goods can influence environmental pollution like the impact of trade on the environment. In addition, López, Galinato, and Islam (2011) state that government spending on public goods may have a certain impact on the environment through three different channels, namely, the scale effect, the composition effect, and the technical effect. Based on the research of López, Galinato, and Islam (2011), López and Palacios (2014) propose two more channels to illustrate how the scale and composition of government expenditure may influence the environment. They point out that increasing government expenditure on public transportation has a certain substitution effect on private transportation, while the former has less energy demand and fewer pollutant emissions than the latter. Another channel indicates that a higher level of investment in research and development contributes to the promotion of energy-efficient and energy-saving appliances.

Based on a panel data set of 21 European countries covering 1995–2006, López and Palacios (2010) find that both government expenditure and energy tax have significant negative influences on air pollution, regardless of the composition of government expenditure.
on public goods. Before long, motivated by increasing government expenditure aimed at stimulating economies during the recent economic crisis, López, Galinato, and Islam (2011) model and examine the impact of both the level and composition of government spending on the environment. They conclude that a higher proportion of government spending on public goods to total government spending significantly helps alleviate water and air pollution. An increase in total government spending with an unchanged composition does not reduce emissions. Employing government expenditure as a proxy for government size, Carlsson and Lundström (2001) state that the size of government has a regulation effect on environmental problems since environmental protection is a public good and needs certain political interventions. They construct a sample of 77 countries covering 1977–1996 and find that greater economic freedom can significantly contribute to the promotion of environmental quality if the government is small, while it will exacerbate air pollution when the size of the government is large. Based on a panel data set of 42 countries from 1971 to 1996, Bernauer and Koubi (2013) apply the proportion of government spending to gross domestic product (GDP) as a measure of government size. Empirically, they find that a higher proportion of government spending to GDP significantly increases $SO_2$ emissions, which is mainly attributed to the negative consequences of large governments, such as bureaucratic inefficiency and the influence of special interest groups. Some studies in this field suggest that the influence of government expenditure on environmental performance may be moderated by other factors, such as national income level and democracy level. Halkos and Paizanos (2013) use the generalized method of moments method on a sample of 77 countries between 1980 and 2000. They find that government expenditure alone directly and negatively influences per capita $SO_2$ emissions, while it only insignificantly impacts carbon dioxide ($CO_2$) emissions in the sample countries. Their results also indicate that the relationship between government expenditure and air pollution is influenced by incorporating the level of national income, that is, government expenditure contributes to decreasing $SO_2$ emissions if the national income is low, and vice versa; the impact of government expenditure on $CO_2$ emissions is significantly negative regardless of the level of national income. Galinato and Islam (2017) find a countervailing effect of a shift in government expenditure toward public goods, indicating that an increased income level leads to severer pollution yet increased environmental regulation helps alleviate environmental problems. They also state that a larger scale of government expenditure on public goods significantly lowers the level of nitrogen dioxide and ozone emissions for countries with a higher democracy level.
In a similar vein, Ercolano and Romano (2018) study a similar case in Europe and find that government expenditure on environmental protection is positively associated with better environmental performance. Huang (2018) uses a sample of 30 provinces in the People’s Republic of China (PRC) between 2008 to 2013 and examines the impact of environmental protection expenditure on SO₂ emissions. The estimations indicate that government expenditure on environmental protection is significantly conducive to the reduction of SO₂ emissions. Moreover, Gholipour and Farzanegan (2018) examine the impact of environmental protection expenditure on air pollution at different levels of governance quality based on panel data of 14 Middle Eastern and North African countries during 1996–2015. They find that government expenditure on environmental protection alone cannot significantly promote environmental quality, and its effect is found to rely on the quality of governance.

4.3 Environmental Performance and Environmental Governance in Asia

This section presents a brief overview of the basic environmental conditions of countries all around Asia from three different perspectives: (i) the greenhouse effect, measured by the level of national CO₂ emissions; (ii) energy utilization, represented by the level of energy intensity; and (iii) comprehensive environmental performance, proxied by the Environmental Performance Index (EPI).

4.3.1 Greenhouse Effect in Asia

Human beings have been keen to exploit natural resources aggressively since the Industrial Revolution. This brought the direct consequence of exhaust gases and waste materials being unscrupulously released into the environment. The increase in CO₂ in the atmosphere resulting from the burning of fossil fuels, such as coal and oil, is creating a so-called “greenhouse effect,” consequently raising the world’s average temperature. Although the greenhouse effect has been part of the earth’s workings since its earliest days, a runaway greenhouse effect may in turn make the earth a hostile environment for living things due to its soaring temperature. Recent statistics reveal that the average global temperature has increased by 0.6°C since meteorological observation records began. Global warming and climate change have already resulted in alarming shifts all over the world, bringing about natural disasters such as melting glaciers and rising sea levels.
CO₂ is generally the main type of greenhouse gas (GHG), and the primary source of CO₂ emissions is the use of fossil fuel, which can be emitted from human-induced influences on forests or other land use. CO₂ emissions are directly connected with the production and lives of human beings, and are also an effective indicator measuring the degree of climate change. Global CO₂ emissions grew by 3.7% in 2014 and reached a historic high of 36.14 gigatons. Seen from the entire evolution of CO₂ emissions, Figure 4.1 shows global CO₂ emissions grew moderately throughout the 1990s and enjoyed faster growth from 2002. The growth rate of Asian CO₂ emissions presents a similar path to that of global CO₂ emissions. Global CO₂ emissions increased from 22.15 gigatons to 36.14 gigatons during 1990–2014, with Asian countries contributing the most while the European Union (EU) and North American countries gradually reduced their weight. Amazingly, the share of Asian CO₂

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2 The data source for CO₂ emissions is the World Development Indicators (WDI) database provided by the World Bank, and the available data range for CO₂ emissions covers the period 1960–2014.

3 Data for the EU contain CO₂ emissions of all 28 member states, which are calculated manually; data for North America are sourced directly from the WDI database, excluding low- and middle-income countries.
emissions has more than doubled in the past 2 decades, from 25.3% in 1990 to 51.6% in 2014. In contrast, the EU successfully reduced its share of CO₂ emissions by half during the same period, from 18.5% in 1990 to 9.0% in 2014.

In recent years, it has become more and more apparent that the differences between developed continents (represented by Europe and North America) and developing continents (such as Asia) not only lie in the aspects of economic and social development but also exist in the condition of environmental pollution. Taking CO₂ emissions as an example, the CO₂ emissions of Asian countries were 18.65 gigatons in 2014, which was almost six times that of the EU and about 3.2 times that of North American countries (Figure 4.2).

This phenomenon apparently fits the well-known environmental Kuznets curve (EKC) theory, which proposes the existence of an inverted U-shaped relationship between economic development and environmental quality; that is, the level of CO₂ emissions increases with rapid economic development in developing countries (most of which are located in Asia) and the level of CO₂ emissions decreases with steady economic development in developed countries (most of which are located in Europe and North America).

If further analysis of regional contributions to global CO₂ emissions is conducted (Figure 4.3), it becomes obvious that developed countries, such as the EU members and the US, have achieved demonstrable

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**Figure 4.2: Carbon Dioxide Emissions of Different Regions in 2014**

<table>
<thead>
<tr>
<th>Region</th>
<th>Carbon dioxide emissions in 2014</th>
</tr>
</thead>
<tbody>
<tr>
<td>European Union</td>
<td>3.24</td>
</tr>
<tr>
<td>North America</td>
<td>5.79</td>
</tr>
<tr>
<td>Asia</td>
<td>18.65</td>
</tr>
</tbody>
</table>

Note: The unit of carbon dioxide emissions is the gigaton.  
Source: Authors’ calculation based on data provided by the World Development Indicators database.
success in reducing GHG emissions. The US was the biggest emitter of CO$_2$ in 1960, accounting for 30.76% of the total, and although it was still the top CO$_2$ emitter in the world in 2014, its share fell to only 14.54%. Similarly, the EU successfully lowered its share of CO$_2$ emissions as well, from 25.11% in 1960 to 8.97% in 2014. Asian countries, by contrast, present an ever-increasing trend in CO$_2$ emissions, with the PRC being the typical representative of unchecked GHG emissions. During 1960–2014, the PRC suffered a quadruple increase of CO$_2$ emissions and jumped to become the biggest CO$_2$ emitter in the world, accounting for 28.48% of the total. Another representative is India, whose condition is similar to that of the PRC, both in economic development and in the evolution of GHG emissions, which suffered an incredible fivefold expansion in CO$_2$ emissions from 1.28% in 1960 to 6.19% in 2014, while Japan, one of the most developed countries in Asia, barely maintained a relatively moderate growth rate of CO$_2$ emissions (from 2.48% in 1960 to 3.36% in 2014) compared to other Asian countries. The evolution of CO$_2$ emissions in Europe, North America, and Japan again provides sound evidence supporting the existence of an EKC relationship between economic development and CO$_2$ emissions.

### 4.3.2 Energy Utilization in Asia

Generally speaking, a very close relationship exists between energy and the environment. On the one hand, the original natural environment is influenced—and a huge amount of exhaust and pollutants is generated—
during energy acquisition and utilization; and the natural environment, on which people rely for their very existence, is polluted and eventually destroyed if these wastes are left mishandled. On the other hand, the development of both energy and the economy does give a tremendous boost in terms of improving environmental conservation; an ever-increasing energy consumption is accompanied by a strengthening economic force, which in turn provides sufficient financial and technological support to accelerate environmental governance and pollution control.

Although energy is greatly significant to the generation of industrial development and social wealth, it produces considerable pressures on the environment as well, such as GHG and air pollutant emissions generated from the combustion of fossil fuel, oil spills during the production process, and nuclear waste that is seriously detrimental to human health. Energy intensity is a widely adopted variable assessing the level of energy efficiency,\(^4\) that is, lower energy intensity shows that less energy is used to produce one unit of economic output and thus represents higher energy efficiency, and vice versa.

A rough comparison of energy intensity in 1990 and 2015 shows that energy intensities in all regions successfully improved (Figure 4.4). Global energy intensity decreased from 7.58 to 5.13, with a 32.3% reduction in total. Asia most successfully improved its energy intensity during this period, decreasing from 11.14 to 5.85, a 47.5% reduction in total.

On the one hand, this implies an ascending awareness of energy conservation. The energy situation of developed countries has effectively improved since the establishment of the International Energy Agency, an intergovernmental organization set up by the Organisation for Economic Co-operation and Development in 1974. Serving as a global energy authority, the International Energy Agency suggests that governments formulate policies that would improve energy reliability, affordability, and sustainability, encouraging member states to improve their energy efficiency and offset the negative impact of energy production. On the other hand, sustained economic growth not only improves the quality of human life, it also brings sufficient capital input for the technological improvement of energy efficiency and environmental governance, which is evidenced by the relationship between economic development and energy intensity in Asia (Figure 4.5).

\(^4\) Energy intensity is calculated as the amount of energy consumed per unit of GDP output, of which the unit is the megajoule/$2010 purchasing power parity GDP. Data are sourced from the WDI database, and the data range of energy intensity analyzed in this section covers 1990 to 2015.
Figure 4.4: A Comparison of Energy Intensity, 1990 and 2015

Source: Authors’ calculations based on data provided by the World Development Indicators database.

Figure 4.5: The Evolution of Economic Development and Energy Intensity in Asia

GDP per capita = gross domestic product.
Note: GDP per capita is in the form of constant 2010 United States dollars.
Source: Authors’ calculations based on data provided by the World Development Indicators database.
For further detailed analysis on relevant issues, we divide our sample into five subgroups: 5 East Asia, Southeast Asia, South Asia, West Asia, and Central Asia. 6 Among Asian countries, Central Asia had the highest average energy intensity of 14.43 during the 1990s, which was almost three times that of Southeast Asia and 1.5 times that of East Asia. Although energy efficiency in Central Asia has effectively improved in recent decades, the energy intensity of Central Asia still represents a dark side of Asian energy conservation. If we follow the thoughts concluded from the relationship between economic development and energy intensity in Asia (Figure 4.5), we should come up with the idea that highly developed economies usually go with a relatively lower energy intensity. While the regional data in Table 4.1 show that East Asia, where the most developed Asian economies can be found, has the second-highest average energy intensity, South Asia, where all countries

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<tr>
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<tbody>
<tr>
<td>East Asia</td>
<td>10.63</td>
<td>7.68</td>
<td>6.37</td>
</tr>
<tr>
<td>Southeast Asia</td>
<td>5.22</td>
<td>4.93</td>
<td>4.34</td>
</tr>
<tr>
<td>South Asia</td>
<td>5.11</td>
<td>4.45</td>
<td>3.79</td>
</tr>
<tr>
<td>West Asia</td>
<td>7.40</td>
<td>5.57</td>
<td>5.12</td>
</tr>
<tr>
<td>Central Asia</td>
<td>14.43</td>
<td>8.87</td>
<td>8.26</td>
</tr>
</tbody>
</table>

Source: Authors’ calculations based on data provided by the World Development Indicators database.

5 Our sample countries comprise 21 Asian economies, and considering data availability, our sample contains several representative countries in each regional category: three countries in East Asia (the PRC, Japan, and Mongolia); six countries in Southeast Asia (Indonesia, Malaysia, the Philippines, Singapore, Thailand, and Viet Nam); four countries in South Asia (Bangladesh, India, Pakistan, and Sri Lanka); seven countries in West Asia (Armenia, Azerbaijan, Iran, Israel, Lebanon, Turkey, and Yemen); and one country in Central Asia (Kazakhstan).

6 The country classification of East Asia is referenced from Miller (2008) and Holcombe (2011). The classification of Southeast Asia is based on the list of member states of the Association of Southeast Asian Nations (see details at https://asean.org/asean/asean-member-states/). The classification of South Asia refers to the list of member states of the South Asian Association for Regional Cooperation (see details at http://saarc-sec.org/about-saarc). For Central Asia, the country classification is referenced from https://www.britannica.com/place/Central-Asia. In addition, since the definition of West Asia is admittedly relatively vague, we follow the classification of both the Organisation for Economic Co-operation and Development and the United Nations Industrial Development Organization (see details in Angus [2003] and the International Yearbook of Industrial Statistics 2011 published by United Nations Industrial Development Organization in 2011).
are developing ones, possesses the lowest average energy intensity. This phenomenon indicates that the energy intensity of Asia is not fully determined by the level of economic development, which contradicts the evolution presented in Figure 4.5, and further indicates that other factors influencing the level of Asian energy intensity must exist.

### 4.3.3 Comprehensive Environmental Performance in Asia

This section adopts the EPI to measure national environmental performance comprehensively.\(^7\) The EPI is widely adopted as an effective indicator of environmental trends, and it serves as a national measurement of how close countries are to environmental policy goals. This indicator suits the objective of this chapter perfectly, which is to find the interlink between the environment and environmental governance. As an example, the 2016 version of the EPI comprises two sub-indicators, namely, the Environmental Health Index and the Ecosystem Vitality Index (EVI), with the score of each accounting for 50% of the total EPI. Under this basic framework, the Environmental Health Index reflects the risk of possible environmental pollution to humans, which is comprehensively measured by the level of health impacts, air quality, water, and sanitation. The Ecosystem Vitality Index assesses the vitality of the whole ecosystem of the country, which is evaluated from several perspectives related to environmental governance and species conservation, including water resources, agriculture, forest cover, fish stock, biodiversity and habitat, and climate and energy.

The 2018 EPI reveals that countries with the highest environmental quality are generally located in Europe, the top three being Switzerland, France, and Denmark. Most countries with the worst environmental performance are located in Africa and Asia, with the Democratic Republic of Congo, Bangladesh, and Burundi comprising the bottom three. Undoubtedly, Europe and North America achieve greater environmental performance accompanied by the advanced and stable development of both their economy and their society. South America and Asia, which mainly comprise developing countries with increasingly rapid economic growth but immature governance mechanisms, maintain the second-tier level of environmental performance across the world. The third tier of environmental performance is composed of countries

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\(^7\) Yale University and Columbia University jointly calculate the EPI score in collaboration with the World Economic Forum. The data period covers the years 2000 to 2018. The group has continuously revised the compilation of the EPI, and different versions of the EPI contain different shares of policy categories as well as sub-indicators.
in Africa, most of which still suffer from unsettled and volatile political situations as well as relatively backward economic development. Roughly judging from the regional distribution of EPI levels across the world, environmental performance is considered to have a close connection with both economic development and political situation, which is in line with the general view of most scholars. Furthermore, in the case of Asia, it is quite unexpected that the environmental performance of West Asia is much better than that of East and Southeast Asia, while the worst environmental performance is found in South Asia and Central Asia.

4.3.4 Environmental Governance in Representative Asian Countries

This section presents a simple but clear retrospect on the path of the environmental governance of Asian countries, represented by Japan and the PRC. These two countries have experienced different paths of economic growth, environmental pollution, and environmental governance.

Japan, as one of the few developed countries in Asia, also experienced severe environmental pollution in the early 1960s; yet, as shown in the ranking of the latest-released EPI, the environmental quality of Japan is at the forefront of Asian countries. The improvement in the environmental quality of Japan is obvious, which makes its environmental governance worth learning. The environmental pollution problem of Japan arose approximately from the mid-1950s to the mid-1970s, when Japan was in the postwar period and its economy was enjoying blistering catch-up growth. The Government of Japan energetically developed heavy industries, such as steel, electricity, and petrochemical, which consumed a huge amount of natural resources and further resulted in several typical environmental pollution incidents. Suffering the directly adverse impact of environmental pollution, residents spontaneously organized campaigns against environmental pollution, which finally evolved into a national protest campaign. Benefiting from the unremitting effort of the public, the Japanese government eventually enacted an official environmental protection act in 1968, the Atmospheric Pollution Prevention Law. Later, the government gradually formulated a series of environmental policies aimed at fighting various kinds of environmental pollution, including laws and acts on air, water, and ocean pollution; energy conservation; and resource recycling. Nowadays, the environmental quality of Japan has comprehensively improved compared to a half century ago, and has mainly benefited from the ever-improving environmental legislation system and the government and authorities being focused on environmental governance.
The Japanese government has released four types of environment-related laws to help enhance the rigors of environmental governance. The first is the basic law that is related to environmental conservation and the control of environmental pollution. These basic laws draft a set of broad principles and general provisions for industries, organizations, enterprises, and citizens. The second type is the general laws specialized in specific aspects of environmental conservation. These laws have meticulous provisions and stipulations and put forward corresponding measures on environmental governance and pollution prevention. The third type of laws is the comprehensive laws on environmental protection, aimed at supervising and regulating relevant behaviors of environmental governance. Furthermore, although the last of these does not directly belong to environmental laws, its content is closely associated with environmental protection, such as the Law on Rationalization of Energy Utilization enacted in 1979. As well as a relatively complete legal system relating to environmental governance, the Japanese government has established various specialized political institutions relating to environmental governance. The Japanese central government set up the Environment Agency in 1972 and the Lower House of Parliament voted to upgrade the Environment Agency to a ministry in 2001. The Ministry of the Environment of Japan is responsible for formulating environmental policies and supervising environmental governance, and comprises seven branches: (i) the Department of Environmental Policy; (ii) Department of Global Environment; (iii) Department of Waste Management and Recycling; (iv) Department of Air and Transportation; (v) Department of Water, Soil, and Ground Environment; (vi) Department of Health and Chemicals; and (vii) Department of Nature and Parks.

Another representative country is the PRC, which is the biggest developing country and now the second-largest economy in the world. From an overall perspective, the environmental situation has still been grim for the PRC in recent years, with environmental problems such as sustained hazy weather, soil contamination, overgrazing, desertification, garbage disposal, and serious destruction of biodiversity having plagued its future development. The idea of the PRC's environmental governance was first proposed in 1973; subsequently, the Environment Protection Law of the PRC was decreed in 1989 and marked the formation of the legislative framework for the country's environmental governance. The National

---

8 Examples are the Basic Law on Environmental Pollution Control and the Basic Law on Pollution Countermeasures released in 1967, and the Environmental Basic Law enacted in 1993.

9 This type of law is represented by the Law on Air Pollution Control and the Noise Control Ordinance implemented in 1993.
People’s Congress and its Standing Committee have already formulated and implemented nine laws involving environmental protection and 15 laws on the protection of natural resources since the foundation of the PRC. The national government has enacted or modified environmental protection laws on several aspects, including the prevention, treatment, and control of water pollution, air pollution, environmental noise pollution, solid waste pollution, marine environmental pollution, and radioactive contamination. Examples of these are the Law on the Prevention and Control of Pollution from Environmental Noise released in 1996, the Law on the Prevention and Control of Environmental Pollution Caused by Solid Waste promulgated in 2005, the Law on Prevention and Control of Water Pollution implemented in 2008, the Law on the Prevention and Control of Atmospheric Pollution revised in 2015, the Environmental Impact Assessment Law revised in 2016, and the Marine Environment Protection Law revised and promulgated in 2017. The government has implemented laws on aspects closely correlated with environmental protection, such as laws on the regulation and supervision of cleaner production, agriculture, animal husbandry, and renewable energy resources, including specifically the Law on Desert Prevention and Transformation formulated in 2001, the Renewable Energy Law of the PRC adopted in 2005, the Energy Conservation Law implemented in 2008, the Regulations for the Administration of the Recovery and Disposal of Waste Electric and Electronic Products enacted in 2011, the Regulations on Urban Drainage and Sewage Treatment implemented in 2014, and the Law of the PRC on Conserving Energy revised and implemented in 2016. In addition, the PRC has actively participated in international cooperation and has signed more than 50 international treaties related to environmental protection, represented by the United Nations Convention to Combat Desertification and the United Nations Convention on Biological Diversity.

The construction of the PRC’s political institutions relating to environmental governance started in the 1970s and experienced a winding course from scratch. In 1974, the State Council officially organized the Leading Team of Environmental Protection, aimed at organizing the environmental conservation efforts of local areas and helping finalize national plans for environmental protection. In 1982, the Ministry of Urban and Rural Development and Environmental Protection was established, with one internal department called the Environmental Protection Bureau being reshuffled into the National Environmental Protection Agency in 1984 and taking over the responsibility for regulating and supervising national environmental governance and protection. In 1998, the National Environmental Protection Agency was upgraded to a higher administrative level and was renamed the State Environmental Protection Administration. In 2008, it was again upgraded to a higher administrative level, namely, the Ministry of Environmental Protection, being an integral department directly under the governance of the State
Council. The evolution of the PRC’s political institutions relating to environmental protection reflects an increasing national recognition of environmental governance; however, the PRC’s capability in terms of environmental governance remains limited.

Comparing the environmental governance of the PRC with that of Japan, we certainly find several similarities. For example, both countries have established specialized political institutions responsible for environmental protection, regulation, and governance. Moreover, both countries have formulated a series of correlative environmental laws to set environmental standards; clarify the rewards and punishments related to corresponding environmental outcomes; and constrain the behavior of industries, enterprises, and citizens. It is worth noting that the environmental governance in Japan and the PRC started at very similar times. Yet it seems that Japan has successfully achieved a remarkable amount in that regard, while the PRC’s environmental deterioration remains extraordinarily disturbing. The reason for this may lie in the inequalities in the allocation of power within the PRC’s political system. In other words, the Ministry of the Environment of Japan is a fully independent political institution with absolute power in terms of integrated environmental governance. In contrast, the Ministry of Environmental Protection of the PRC is directly affiliated with the authority of the State Council. Thus, said ministry of the PRC needs to maintain more independence and authority to rule on the formulation of environmental laws and environmental policies.

4.4 Relationship Between Environmental Governance and Environmental Performance in the Context of Asia

4.4.1 Data Description

Despite the limited availability and quality of the data, we construct an unbalanced panel data set of 18 Asian countries from 2005 to 2014 and 26 European countries covering the period 2008–2013.10 In the empirical

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10 The 18 Asian countries are Afghanistan, Azerbaijan, Bahrain, Bangladesh, the PRC, Georgia, Iran, Israel, Jordan, the Kyrgyz Republic, Lebanon, Nepal, Oman, Pakistan, the Philippines, Qatar, Thailand, and Turkey. The 26 European countries are Austria, Belgium, Bulgaria, Croatia, Cyprus, the Czech Republic, Denmark, Estonia, Finland, France, Germany, Hungary, Italy, Latvia, Lithuania, Luxembourg, Malta, Norway, Poland, Portugal, Romania, the Slovak Republic, Slovenia, Spain, Sweden, and the United Kingdom. The data of Asian and European countries are sourced from the International Monetary Fund and Eurostat, respectively.
estimation, we mainly focus on the relationship between government expenditure on environmental protection and three representative factors of environmental quality, which have all been introduced before. Among them, government expenditure on environmental protection (Expenditure) is presented as the percentage of GDP, while three environmental variables are CO₂ emissions, energy intensity, and Environmental Performance Index, respectively. We further introduce a series of control variables that may exert a certain influence on environmental governance, including GDP per capita, population size, level of urbanization, industry structure, foreign direct investment (FDI), and trade openness. Here, we will briefly define these control variables.

(i) GDP per capita (GDP): We introduce GDP per capita as one of the major control variables following the EKC theory, which demonstrates the close correlation between economic development and environmental quality. The unit of GDP is constant 2010 US dollars.

(ii) Population (Population): This is proxied by the national population size. The significant relationship between demography and the environment has aroused interest among scholars for more than 200 years, when Malthus (1798) first proposed that unrestrained population growth would be restricted by limited natural resources. Most scholars have agreed with the conclusion that increasing population size is significantly correlated with environmental pollution such as CO₂ emissions (Dietz and Rosa 1997; Cramer 1998).

(iii) Urbanization rate (Urbanization): This is measured as the proportion of urban population to total population. There are inconsistent conclusions on the effect of the urbanization rate on environmental performance. For example, Panayotou (1997) stated that an increasing degree of urbanization is accompanied by rising consumption of fossil fuels and thus poorer environmental quality. However, Poumanyvong and Kaneko (2010) found that an ascending level of urbanization lowers energy use in the low-income group, but exacerbates the energy use in upper-middle-income groups.

(iv) Industry structure (Industry): This is measured as the ratio of industry sector value added to GDP. It is widely argued that the level of industrialization is a crucial determinant of environmental performance, and Cherniwchan (2012) empirically demonstrated that the increasing ratio of industrial output to total output is significantly correlated with the rising level of emissions per capita.

(v) Foreign direct investment (FDI): This is proxied by the net inflows of FDI as a share of GDP. The effect of FDI on
Environmental pollution is a controversial issue. Some scholars support the “pollution haven hypothesis” proposed by Walter and Ugelow (1979), arguing that greater foreign investments would be detrimental to the environment of investee countries (Baumol and Oates 1988; Cole 2004). Other scholars agree with the “pollution halo hypothesis,” which states that FDI toward developing countries would promote the enhancement of energy efficiency and thus benefit the environmental conservation of investee countries (Letchumanan and Kodama 2000; Eskeland and Harrison 2003).

(vi) Trade openness (Openness): This is calculated as the sum of export and import of goods and services measured as a share of GDP. The evidence from relevant studies on the relation between trade openness and environmental quality is mixed. Scholars like Antweiler, Copeland, and Taylor (2001); Baek, Cho, and Koo (2009); and Boulatoff and Jenkins (2010) found that trade openness is linked with decreasing environmental pollution and waste emissions, while several researchers support the argument that trade openness would worsen environmental quality (Kellenberg 2009; Managi and Kumar 2009). Other scholars like Le, Chang, and Park (2016) state that the impact of trade openness on the environment differs according to the income level of countries.

<table>
<thead>
<tr>
<th>Variables</th>
<th>Obs</th>
<th>Mean</th>
<th>Std. Dev.</th>
<th>Min.</th>
<th>Max.</th>
<th>Source</th>
</tr>
</thead>
<tbody>
<tr>
<td>CO₂ emissions</td>
<td>180</td>
<td>7.483</td>
<td>10.902</td>
<td>0.053</td>
<td>62.824</td>
<td>WDI</td>
</tr>
<tr>
<td>Energy intensity</td>
<td>180</td>
<td>5.601</td>
<td>2.273</td>
<td>1.268</td>
<td>11.293</td>
<td>WDI</td>
</tr>
<tr>
<td>EPI</td>
<td>163</td>
<td>53.782</td>
<td>12.244</td>
<td>21.57</td>
<td>82.176</td>
<td>EPI released by Yale University</td>
</tr>
<tr>
<td>Expenditure</td>
<td>160</td>
<td>0.343</td>
<td>0.400</td>
<td>0</td>
<td>2.444</td>
<td>International Monetary Fund</td>
</tr>
<tr>
<td>GDP</td>
<td>180</td>
<td>10,561.63</td>
<td>16,117.71</td>
<td>389.416</td>
<td>72,671</td>
<td>WDI</td>
</tr>
<tr>
<td>Population</td>
<td>180</td>
<td>1.14e+08</td>
<td>3.01e+08</td>
<td>864,863</td>
<td>1.36e+09</td>
<td>WDI</td>
</tr>
<tr>
<td>Urbanization</td>
<td>180</td>
<td>58.360</td>
<td>24.649</td>
<td>15.183</td>
<td>99.159</td>
<td>WDI</td>
</tr>
<tr>
<td>Industry</td>
<td>178</td>
<td>14.976</td>
<td>6.935</td>
<td>4</td>
<td>32.452</td>
<td>WDI</td>
</tr>
<tr>
<td>FDI</td>
<td>180</td>
<td>4.491</td>
<td>4.897</td>
<td>-0.423</td>
<td>33.795</td>
<td>WDI</td>
</tr>
<tr>
<td>Openness</td>
<td>180</td>
<td>0.736</td>
<td>0.445</td>
<td>0</td>
<td>1.905</td>
<td>WDI</td>
</tr>
</tbody>
</table>

CO₂ = carbon dioxide, EPI = Environmental Performance Index, FDI = foreign direct investment, GDP = gross domestic product, WDI = World Development Indicators.

Source: Author’s calculations.
Tables 4.2a and 4.2b present the descriptive statistics and the data sources of the variables, respectively, for the Asian and European samples used in the empirical estimation.

### 4.4.2 Empirical Investigation

First, we conduct panel fixed-effect regression to investigate the relationship between \( \text{CO}_2 \) emissions and environmental governance, proxied by the ratio of government expenditure on environmental protection to GDP, and the results are displayed in Table 4.3. Quite unexpectedly, the estimation results present entirely different implications for Asian and European countries. For Asian countries, the variable *Expenditure* has a significantly negative influence on \( \text{CO}_2 \) emissions at the 10% level, indicating that a greater scale of government expenditure on environmental protection contributes significantly to the reduction of \( \text{CO}_2 \) emissions in Asian countries. Moreover, *GDP* has a significantly positive and relatively large impact on \( \text{CO}_2 \) emissions, implying that \( \text{CO}_2 \) emissions increase along with the development of the Asian economy. This finding further provides evidence that most Asian countries are located on the left half of the EKC, where environmental pollution increases along with rising GDP per capita. For European
Table 4.3: Estimation Results of Carbon Dioxide Emissions and Environmental Governance

<table>
<thead>
<tr>
<th>Variables</th>
<th>Asia</th>
<th>Europe</th>
</tr>
</thead>
<tbody>
<tr>
<td>Expenditure</td>
<td>-0.288*</td>
<td>0.008</td>
</tr>
<tr>
<td></td>
<td>(0.164)</td>
<td>(0.007)</td>
</tr>
<tr>
<td>GDP</td>
<td>7.086*</td>
<td>-2.816</td>
</tr>
<tr>
<td></td>
<td>(3.693)</td>
<td>(2.870)</td>
</tr>
<tr>
<td>GDP²</td>
<td>-0.386*</td>
<td>0.186</td>
</tr>
<tr>
<td></td>
<td>(0.233)</td>
<td>(0.146)</td>
</tr>
<tr>
<td>Population</td>
<td>-0.369</td>
<td>-0.133</td>
</tr>
<tr>
<td></td>
<td>(0.632)</td>
<td>(0.271)</td>
</tr>
<tr>
<td>Urbanization</td>
<td>-1.813</td>
<td>-2.943***</td>
</tr>
<tr>
<td></td>
<td>(1.898)</td>
<td>(0.871)</td>
</tr>
<tr>
<td>Industry</td>
<td>-0.441</td>
<td>0.394***</td>
</tr>
<tr>
<td></td>
<td>(0.516)</td>
<td>(0.099)</td>
</tr>
<tr>
<td>FDI</td>
<td>-0.025</td>
<td>0.001</td>
</tr>
<tr>
<td></td>
<td>(0.087)</td>
<td>(0.001)</td>
</tr>
<tr>
<td>Openness</td>
<td>-0.313</td>
<td>-0.366***</td>
</tr>
<tr>
<td></td>
<td>(0.529)</td>
<td>(0.093)</td>
</tr>
<tr>
<td>Constant</td>
<td>-15.970</td>
<td>24.841</td>
</tr>
<tr>
<td></td>
<td>(16.760)</td>
<td>(16.141)</td>
</tr>
</tbody>
</table>

| Observation | 140 | 156 |
| R²          | 0.099 | 0.509 |
| F           | 1.56 | 15.81 |

FDI = foreign direct investment, GDP = gross domestic product.
Notes: The values in parentheses denote the standard errors. *, **, and *** denote significance at the 10%, 5%, and 1% levels, respectively.
Source: Author’s calculations.

countries, the variables Expenditure and GDP only have insignificant impacts on CO₂ emissions, while Urbanization, Industry, and Openness exert significant influence on CO₂ emissions. Specifically, Urbanization and Openness are proved to have significantly negative influences on CO₂ emissions, indicating that both the promotion of urbanization levels and greater trade openness benefit the reduction of CO₂ emissions in Europe. Industry has a significantly positive effect on CO₂ emissions, representing a greater share of industrial value added, which implies that accelerated
industrial activities aggravate the pressure on CO₂ emissions and further worsen the environmental quality in European countries.

The results show that only CO₂ emissions in Asian countries are significantly affected by the government expenditure on environmental protection. A relatively reasonable explanation for this phenomenon is that most Asian countries rely on the national government to a great extent regarding the issue of environmental protection. The major governance instruments of Asian countries are still mandatory regulations and legislations set by the state. Most European countries, however, have already adopted and developed a hybrid partnership pattern in relation to environmental governance where the state or the government is no longer the most crucial source of decision-making in the field of environmental protection (Armitage et al. 2012). Government expenditure on environmental protection may present partial environmental governance actions only taken by the state. Thus, for European countries, the influence of environmental expenditure on CO₂ emissions can be insignificant or negligible. However, for Asian countries whose decision-making authority largely belongs to the state government, environmental expenditure tends to have a significantly negative impact on CO₂ emissions.

Turning the focus onto how government expenditure on environmental protection influences the level of energy intensity, Table 4.4 presents the basic estimation results generated from the panel fixed-effect model for both Asian and European samples. Much like the results in Table 4.3, the results in Table 4.4 show that expenditure has a significantly negative influence on energy intensity, and GDP has a significantly positive impact on energy intensity for the Asian sample. This indicates that government expenditure on environmental protection can significantly enhance energy efficiency while economic development may be detrimental to energy use. This again provides evidence for the argument that most Asian countries are on the left half of the EKC. In the case of energy intensity, the variable FDI is significantly negative for the Asian sample, implying that large-scale FDI helps promote energy efficiency and lower energy intensity. As far as the European sample is concerned, the variable GDP has a considerable negative impact on energy intensity, thereby indicating that the economic development in European economies helps reduce energy intensity and, thus, improves energy efficiency. Such a negative relationship also reflects the argument that European countries, most of which are developed countries, are on the right half of the EKC where environmental quality improves along with the development of the economy. Like the results for CO₂ emissions, urbanization, openness, and industry present significant impacts on energy intensity as well,
Table 4.4: Estimation Results of Energy Intensity and Environmental Governance

<table>
<thead>
<tr>
<th>Variables</th>
<th>Asia</th>
<th>Europe</th>
</tr>
</thead>
<tbody>
<tr>
<td>Expenditure</td>
<td>-0.095***</td>
<td>0.005</td>
</tr>
<tr>
<td></td>
<td>(0.031)</td>
<td>(0.005)</td>
</tr>
<tr>
<td>GDP</td>
<td>4.034***</td>
<td>-4.235**</td>
</tr>
<tr>
<td></td>
<td>(0.707)</td>
<td>(2.077)</td>
</tr>
<tr>
<td>GDP²</td>
<td>-0.246***</td>
<td>0.201*</td>
</tr>
<tr>
<td></td>
<td>(0.045)</td>
<td>(0.106)</td>
</tr>
<tr>
<td>Population</td>
<td>-0.199</td>
<td>0.254</td>
</tr>
<tr>
<td></td>
<td>(0.121)</td>
<td>(0.196)</td>
</tr>
<tr>
<td>Urbanization</td>
<td>-0.461</td>
<td>-1.237*</td>
</tr>
<tr>
<td></td>
<td>(0.363)</td>
<td>(0.631)</td>
</tr>
<tr>
<td>Industry</td>
<td>0.010</td>
<td>0.254***</td>
</tr>
<tr>
<td></td>
<td>(0.099)</td>
<td>(0.720)</td>
</tr>
<tr>
<td>FDI</td>
<td>-0.036**</td>
<td>0.001</td>
</tr>
<tr>
<td></td>
<td>(0.017)</td>
<td>(0.001)</td>
</tr>
<tr>
<td>Openness</td>
<td>-0.132</td>
<td>-0.134***</td>
</tr>
<tr>
<td></td>
<td>(0.101)</td>
<td>(0.067)</td>
</tr>
<tr>
<td>Constant</td>
<td>-9.312***</td>
<td>24.290**</td>
</tr>
<tr>
<td></td>
<td>(3.209)</td>
<td>(11.684)</td>
</tr>
<tr>
<td>Observation</td>
<td>140</td>
<td>156</td>
</tr>
<tr>
<td>R²</td>
<td>0.285</td>
<td>0.412</td>
</tr>
<tr>
<td>F</td>
<td>8.92</td>
<td>10.68</td>
</tr>
</tbody>
</table>

FDI = foreign direct investment, GDP = gross domestic product.

Notes: The values in parentheses denote the standard errors. *, **, and *** denote significance at the 10%, 5%, and 1% levels, respectively.
Source: Author’s calculation.

with the first two negatively affecting energy intensity and the last one affecting it positively. These results indicate that a higher level of urbanization, greater trade openness, and a lower share of industrial value added contribute to the perfection of energy intensity in Europe.

Finally, we attempt to figure out the relationship between environmental governance and comprehensive environmental performance, proxied by the EPI. Table 4.5 displays the corresponding results for both Asian and European countries. Surprisingly, government
Table 4.5: Estimation Results of the Environmental Protection Index and Environmental Governance

<table>
<thead>
<tr>
<th>Variables</th>
<th>Asia</th>
<th>Europe</th>
</tr>
</thead>
<tbody>
<tr>
<td>Expenditure</td>
<td>-0.002</td>
<td>-0.011</td>
</tr>
<tr>
<td></td>
<td>(0.068)</td>
<td>(0.012)</td>
</tr>
<tr>
<td>GDP</td>
<td>-3.558*</td>
<td>0.193</td>
</tr>
<tr>
<td></td>
<td>(1.973)</td>
<td>(5.178)</td>
</tr>
<tr>
<td>(GDP^2)</td>
<td>0.209*</td>
<td>-0.018</td>
</tr>
<tr>
<td></td>
<td>(0.118)</td>
<td>(0.263)</td>
</tr>
<tr>
<td>Population</td>
<td>0.150</td>
<td>0.538</td>
</tr>
<tr>
<td></td>
<td>(0.296)</td>
<td>(0.488)</td>
</tr>
<tr>
<td>Urbanization</td>
<td>-0.350</td>
<td>7.713***</td>
</tr>
<tr>
<td></td>
<td>(0.779)</td>
<td>(1.572)</td>
</tr>
<tr>
<td>Industry</td>
<td>0.093</td>
<td>-0.669***</td>
</tr>
<tr>
<td></td>
<td>(0.242)</td>
<td>(0.179)</td>
</tr>
<tr>
<td>FDI</td>
<td>0.036</td>
<td>-0.001</td>
</tr>
<tr>
<td></td>
<td>(0.038)</td>
<td>(0.001)</td>
</tr>
<tr>
<td>Openness</td>
<td>0.134</td>
<td>1.703***</td>
</tr>
<tr>
<td></td>
<td>(0.247)</td>
<td>(0.168)</td>
</tr>
<tr>
<td>Constant</td>
<td>17.374**</td>
<td>-35.522</td>
</tr>
<tr>
<td></td>
<td>(8.075)</td>
<td>(29.119)</td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th></th>
<th>Observation</th>
<th>140</th>
<th>156</th>
</tr>
</thead>
<tbody>
<tr>
<td>(R^2)</td>
<td>0.070</td>
<td>0.635</td>
<td></td>
</tr>
<tr>
<td>(F)</td>
<td>0.94</td>
<td>26.5</td>
<td></td>
</tr>
</tbody>
</table>

FDI = foreign direct investment, GDP = gross domestic product.
Notes: The values in parentheses denote the standard errors. *, **, and *** denote significance at the 10%, 5%, and 1% levels, respectively.
Source: Author’s calculations.

Expenditure on environmental protection has no impact on EPI levels for both Asian and European countries. The potential reason for this unexpected phenomenon could be that the EPI is a composite indicator containing both the performance of environmental health and the level of ecosystem vitality, which may result in an insignificant or negligible relationship between the EPI and government expenditure on environmental protection. Other variables present results similar to the foregoing analysis. To be specific, GDP is significantly negatively
correlated with the EPI, indicating that continuous economic development is harmful to comprehensive environmental performance in Asia and further demonstrates Asia's left-of-center location in the EKC. For European countries, urbanization and openness are significantly positively correlated with the EPI, implying that the development of urbanization of trade openness contributes to the promotion of comprehensive environmental performance. Nevertheless, the negative impact of FDI on environmental quality is again proved by the significantly negative relationship between FDI and the EPI.

4.5 Conclusion

With the interaction between humans and the environment throughout the entire development of society, the devastating influence of humans on the environment has continued to escalate in the last few centuries. Environmental pollution has undoubtedly become an extremely severe problem in today’s world, and the authorities and all stakeholders are believed to be responsible for taking strong and effective measures to deal with this problem. Through the history of environmental governance, although the operating mechanisms, policy instruments, and even the actors involved have changed over the last few decades, people have never stopped pursuing a better ecological environment.

Asia, the largest continent in the world in terms of both land mass and population, has long been facing the exhaustive challenge of environmental pollution. Most Asian countries are developing countries and emerging economies, and rapid economic development has been accompanied by growing environmental problems for decades. From the perspective of environmental governance, there are both well-governed countries, such as Singapore and Japan, and poorly governed countries, such as India, in Asia. By analyzing the basic environmental conditions in Asia from different perspectives, such as CO₂ emissions, energy intensity, and comprehensive environmental performance, we find that West Asia generally has the best environmental quality among all parts of Asia. The environmental performance in South Asia and Central Asia, on the other hand, is much worse than in East and Southeast Asia. Based on the current reality that environmental conditions differ a lot from country to country, we further investigate whether the quality of environmental governance contributes to creating such a difference.

We empirically demonstrate the significant impacts of governments’ environmental expenditure on environmental quality for Asian countries, while the impact is insignificant for European countries. We highlight several conclusions for Asian countries as follows: (i) a greater scale of government expenditure on environmental protection
contributes to a reduction of CO$_2$ emissions and the promotion of energy efficiency; (ii) excessive economic growth is detrimental to the environment, and increasing GDP per capita leads to increasing CO$_2$ emissions, decreasing energy efficiency, and decreasing comprehensive environmental performance; and (iii) although FDI has no impact on CO$_2$ emissions and the EPI, it exerts a significantly negative impact on energy intensity and thus influences energy efficiency. For European countries, all estimations present similar conclusions; that is, a higher level of urbanization, greater trade openness, and a lower proportion of industrial value added benefit the environment through lowering CO$_2$ emissions, improving energy efficiency, and promoting environmental quality.

 Compared with developed economies such as those in Europe, most Asian economies are still located on the left half of the EKC, indicating that rapid economic development of these developing and emerging countries would seriously exacerbate the environmental pollution problem. Emerging Asian economies should balance the pros and cons of both economic development and environmental conservation. Empirical estimation shows that FDI benefits the energy intensity of Asian countries and does not exert a detrimental effect on CO$_2$ emissions and environmental quality. This suggests that Asian economies should implement relevant policies to attract more foreign investment. Although the level of urbanization, industrial value added, and trade openness do not significantly impact the environment in Asia, we observe from the case of Europe that these indicators may still exert a certain influence along with the development of the economy. Hence, we recommend that policy makers of Asian countries emphasize these factors, especially for the share of industrial value added, whose enhancement may be potentially and significantly destructive to the environment.
References


5.1 Introduction

Over the past 20 years, academia and industry practitioners have persistently paid a lot of attention to studying how to address global climate change and decrease carbon emissions efficiently. Historically, emission trading programs are widely believed to have played a prominent role in environmental policy, especially to reduce carbon emissions. As a result, many regional and national carbon trading markets emerged, including the Regional Emissions Trading Scheme in Australia, the People’s Republic of China (PRC), New Zealand, the Republic of Korea (henceforth Korea), and the United States, and most notably the emissions trading system set up by the European Union (EU ETS). As far as we are concerned, to date the EU ETS is the major emission cap-and-trade trading program across the world, accounting for about $175 billion a year. The history of the EU ETS can be traced back to 2005; today it allocates tradable emissions permits to more than 12 large power stations and industrial plants in more than 20 countries, accounting for about half of the EU’s total greenhouse gas (GHG) emissions in aggregate.

Previous research focused on the impacts of energy, financial markets, and macroeconomic factors on the EU carbon emission market (see, for instance, Alberola et al. 2008; Chevallier 2009; Koch 2014). However, to the best of our knowledge, fewer studies have focused on the relationship between the carbon emission markets of the PRC and the EU. As the PRC is the largest GHG emitter, its government has given a lot
of attention to setting up carbon trading markets, such as the Shanghai and the Shenzhen carbon emission markets. Therefore, the growth of carbon emission markets in the PRC has raised the question: what is the relationship between regional carbon emission markets? Moreover, the lessons from the linkages between regional carbon emission markets could help enhance the carbon emission market cooperation both in the PRC and around the world and provide additional insight for other developing countries into the domestic emissions trading system (ETS). Our main empirical findings suggested the existence of co-integration between the futures prices in the PRC’s carbon emission markets. In particular, the Shanghai carbon emission market positively affects the Shenzhen carbon emission market, but a similar vice versa relationship could not be found. Our findings shed light on the policy formulation of the carbon emission markets for the PRC government to achieve sustainable economic growth and meet the carbon reduction targets of the country.

Since the well-known economic revolution that started in the late 1970s, the economy of the PRC has undergone decades of significant developments and became the second-largest economy in the world in 2016, according to the World Bank’s data on gross domestic product (GDP) at the end of 2017. Despite its undebatable economic success, a side effect of the PRC’s economic progress is the unprecedented large volume of carbon emissions. As a result, the National Development and Reform Commission issued “interim measures for voluntary GHG emissions trading management” in June 2012 to explore the possibility of a voluntary emissions trading program in the PRC. This is the country’s initial attempt to reduce carbon emissions and has been generally regarded as a signal to the public. It lays out the theoretical foundation for further practical implementation of mandatory carbon trading mechanisms as well as the establishment of carbon emission rights exchanges in Shanghai, Shenzhen, and other places in the PRC. As a matter of fact, the first carbon emission exchange in the PRC was set up in Shenzhen in June 2013, and Shanghai and other places had followed suit by the end of November 2013. These have since become the two most important carbon emission exchanges in the PRC, parallel to the two most important stock exchanges in the country in Shanghai and Shenzhen. Given the current prominent status of the PRC’s economy, a significant reduction in the level of carbon emissions of the PRC stock market will be expected from these programs.

In theory, the establishment of carbon emission markets will significantly decrease the level of carbon emissions of the country, which has milestone implications for its carbon emission reductions. In practice, however, there was a trivial volume before 2007. To date, to
the best of our knowledge, no paper has discussed the interrelationship among the futures prices in multiple carbon emission exchanges in the PRC. However, a large strand of literature has already discussed the interrelationship between the stock prices in the Shanghai and Shenzhen stock exchanges. This chapter aims to focus on these two most important carbon emission exchanges. To be specific, it will examine whether the futures price in the Shanghai carbon emission exchange affects the futures price in the Shenzhen carbon emission exchange, and vice versa, considering the exogenous shock mainly from the EU ETS. This chapter will contribute to the existing literature in the following three ways: first, it is the first to investigate the dynamic linkage of the futures prices between these two markets. The study results will shed light on the interrelationship of the futures prices in these two markets and provide risk management guidelines for the two markets’ investors. Second, this study applies daily high-frequency data from 3 January 2017 to 9 October 2018. An investigation into the futures price on a lower-frequency level cannot capture the dynamic influence, while intraday high-frequency data will suffer from much misconstruction noise in this study. Third, we introduce the autoregressive distributed lag model to this topic, which integrates short-run adjustment effects with long-run equilibrium without losing information (see Jalil and Feridun 2011).

5.2 Background

Since the emergence of the EU ETS in 2005, the carbon emission market has emerged to become a prominent system for mitigating climate change across the globe. Countries of the Association of Southeast Asian Nations have focused their attention on this modern and global idea of mitigating climate change, having noted the rate at which emissions continue to increase in the region. Northeast Asian countries such as the PRC, Japan, and Korea contribute more than one-fifth of the world's GDP and more than a quarter of global emissions (Ewing 2016a). Equally, the increase in emissions in Southeast Asia is almost sprinting forward at the same rate as its economic growth, with a nearly 5% annual increase from 1990 to 2010 (Raitzer et al. 2015). This surge has been traced to the relative escalation of emissions in sectors such as manufacturing and transportation. These two sectors are associated with the region's structural change rather than rural or agrarian areas.

There is, therefore, an urgent need to curtail the increase in emissions, owing to the fact that emissions cause much harm to humanity and the economy as a whole. Given that, global carbon finance needs to be upheld by an order of “weight-carrying” in the second commitment period of the Kyoto Protocol. To enable the stakeholders in
the carbon market to see the opportunities far away, outside the project-based Clean Development Mechanism (CDM), alternative schemes, such as allowance-based market mechanisms, may be considered in the emerging economies in a similar manner to the one that functions in Europe, i.e., the EU ETS (Grubb 2012; Perdan and Azapagic 2011). Evidently, there has been significant progress in this market as the trend has gained popularity among the major Asian economies, including the PRC, India, Japan, Kazakhstan, and Korea. The PRC, which is the world’s major growing market, is expected to have a greater potential for the scale of carbon trading (Jotzo et al. 2013; Wang 2013, Fankhauser 2011, Guan and Hubacek 2010).

Based on the above, we see clearly the significance of carbon emission mechanisms, which are considered popular instruments in the domestic and international circuit for enhancing efforts toward mitigating climate change. These instruments often put a price tag on emissions that lead to climate-damaging GHGs, which eventually leads to the promotion of climate change mitigation efforts. More specifically, two approaches led to the creation of carbon markets, i.e., the cap-and-trade mechanism and the carbon tax mechanism (Hartmann 2017).

**Cap-and-trade mechanism.** The cap-and-trade mechanism is used by governments or intergovernmental bodies to set a limit on GHG emissions in a given period. This mechanism is designed to offer licenses to both companies and industries to minimize the growth of pollution (Stavins 2008). The scheme is designed in such a way that if companies that do not meet their cap buy licenses from others, they will have a surplus. The technicality here is that it provides an opportunity to reduce GHGs with a cost-effective platform. However, it is also widely criticized because it rewards most polluters with windfall profits while undermining the efforts to reduce pollution and achieve a more sustainable economy.

Despite this criticism, its popularity remains high, going by the wide range of potentials, which can engender considerable revenues. These collected revenues have significant implications for distribution in the spirit of fairness and economic growth. The possible uses of carbon revenues could include: (i) counterbalancing the unequal effects of high energy prices on low-income households in the form of discounts on electricity bills; (ii) providing transitional assistance (e.g., arranging or providing financial support for job training) for communities and individuals whose livelihood depends on fossil fuels; (iii) providing financial support for communities that face an unequal burden from carbon emissions from fossil fuels; and (iv) investing in clean/renewable energies, energy-efficient technologies, and clean vehicles, which all
play an important role in moving the economy from a carbon-intensive to a clean-energy economy.

In addition, cap and trade in the form of ETSs has progressed faster and attracted the attention of policy makers and practitioners since the emergence of the EU ETS in 2005. For instance, in 2014, 18 ETS programs were initiated across 18 national and subnational jurisdictions. This initiation was planned for one more country and was also being considered for implementation in 11 more national and subnational jurisdictions (World Bank 2014). Subsequently, the implementation and planning of ETSs spread across all regions except in Africa. The ETSs also covered nearly 9% of global GHG emissions by 2015. Before 2015, the World Bank (2014) documented that the majority of implemented ETS programs have made landslide achievements in recent years in terms of scope, linkages, and development of approaches. To this end, the cap-and-trade mechanism could be said to be gaining ground among European, Asian, and North American countries. It is hoped that this will also spread to other regions such as sub-Saharan Africa and Africa.

**Carbon tax mechanism.** Carbon tax is another mechanism used to curtail excessive emissions around the world. Carbon tax is a form of tax whereby each unit of GHG emissions gives participating firms (and households, depending on the scope) an incentive to reduce pollution (Taschini et al. 2013). Hence, the rate of pollution is reduced, depending on the chosen level of tax. For these reasons, carbon tax could be assessed by considering the cost or damage associated with each unit of pollution and the cost associated with controlling that pollution. Therefore, getting the tax level correct is the key, even though some firms and households will most likely opt to pay the taxes, while continuing to pollute the environment, beyond and above what is optimal for society. If the level of pollution is high, then the cost will rise higher than necessary to reduce emissions, and it will have a considerable impact on profits, jobs, and end-use consumers.

It is also important to highlight here that carbon tax can cause an increase in business overheads. In this case, companies will be pushed to find efficient ways to manufacture their products and/or deliver their services, as this would be beneficial to their bottom-line users. This idea supports more environment-friendly and creative ideas among industries. Furthermore, carbon emissions have also been linked to worsening public health as they have a major impact on the quality of the air we breathe. However, implementing a carbon tax could improve air quality by reducing carbon emissions, thereby decreasing the rate of respiratory issues and the number of asthma attacks (Kramer and Fraser-Hidalgo 2018).
Based on the above, many countries have demonstrated knowledge of carbon trading via various approaches hinged on the flexibility of carbon taxes to fit into national circumstances. For instance, in 2014, over 14 countries implemented carbon tax approaches to fight or reduce the level of emissions in their countries. Some of these countries took various approaches in light of national circumstances (to enable the use of carbon taxes together with other carbon pricing appliances), while others turned toward approaches that considered industry competitiveness (World Bank 2014).

On the other hand, politicians around the world favored the use of the carbon trading option over the second option, carbon tax. This is because in the carbon trading scheme, each participant firm is given a certain emission allowance quota in the primary ETS market, and then trades with other members in the secondary ETS market for extra allowance to support its ongoing production or for benefits if the permitted quota is not used (Tang et al. 2016; Zhang et al. 2015). Usually in the initial stages of trading, the emission allowance quota is either given to businesses for free or sold at auction. Therefore, over time the number of existing permits decreases, putting more pressure on participating firms to invest more money in production technologies to reduce the growth of carbon emissions. Hence, this approach will help firms to come up with more innovations, which will significantly assist them in reducing the price of new technologies.

5.3 The Growth of Carbon Markets in Asia

Since 2015, the issue of climate change has become a global agenda as many efforts have been made to meet the Paris Agreement. Specifically, the participating countries are dedicated and prepared to address the climate change issue through various means. To date, 175 countries around the world have ratified the agreement, including, among other Asian countries, countries in Northeast and Southeast Asia (Ewing 2016a). This section seeks to explore the current status in the Asian region, particularly in the major economies of Northeast Asia. To address the increasing concerns around the issue of climate change, the major economies in Northeast Asia have developed carbon markets. The growth of carbon markets has been very impressive in the last few years. More specifically, the number of carbon markets has doubled since 2012 and general interest in carbon trading is also on an upswing. Further, emission pricing is worth $50 billion (Ewing 2016b). In the following, we discuss three major Northeast Asian economies: the PRC, Japan, and Korea.
**The People’s Republic of China.** The main objective of the Kyoto Protocol is to address the issue of climate change by introducing a proper market mechanism. It has been noted in the recent past that the international carbon market is growing rapidly and is well diversified across various forms of markets, such as the CDM, voluntary emission reduction, and emission allowance trading. The PRC is not an exception, as it has challenged itself to build a more formidable carbon market, having realized the volatile nature of an economy in which carbon emissions are at an extreme level due to industrialization. As a result, in 2011 the PRC pledged its support toward promoting low carbon emission development through carbon trading where the set target of energy consumption per unit of GDP would drop by 3.4% in 2014 and 2.32% in 2015 (*The Economist* 2015). By the end of September 2017, the market had covered 3,000 entities from more than 20 industry sectors, with the total trading volume reaching 200 million tons of CO₂ equivalent (tCO₂e) while the total trading value was about CNY45.1 billion and the price ranged from CNY1–123 /tCO₂e (Environmental Defense Fund [EDF] 2017). This therefore indicates that the carbon markets in the PRC are gaining momentum and moving toward achieving a low-emission economy.

As an addendum to the above, huge successes have been recorded in the reduction of carbon emissions (mainly in the power sector) with an estimated 0.09 billion tCO₂e difference when compared to the level of 2016. Fossil fuel power generation will reach 4.88 kilowatt-hours (kWh) with an average efficiency of 305 grams of coal equivalent per/kWh by 2020. By contrast, in 2016, non-fossil fuel power generation increased by 0.06 trillion kWh, and this is expected to decrease to 0.51 trillion tCO₂e by 2020 (EDF 2017). This, therefore, implies that the PRC adopted all carbon mechanisms to achieve its slated plan.

In the ongoing effort to achieve a stronger carbon market, the PRC is establishing strong collaboration with other international counterparts with respect to carbon pricing. For instance, in October 2017, both the PRC and the EU declared continued collaboration on emissions trading. On this platform, the EU agreed to continue to support the PRC in developing its national ETS for 3 years (EU 2017). In addition, on 4 December 2017, the PRC and Canadian governments released a joint declaration on climate change, which basically explains their commitment to strengthening their collaboration, particularly regarding clean energy and carbon markets (Government of the PRC 2017).

Several researchers have also considered the price dynamics of CO₂ allowances to explain the position of emissions. Some authors argue that the PRC’s carbon market is competent. Tang et al. (2017), for example, offered a multi-agent-based ETS simulation approach to test for a
suitable carbon allowance auction design for the PRC’s ETS. The authors concluded that the ETS has a substantial positive impact on reducing emissions and improving the energy structure. However, the authors advised that it adversely affects economic growth. Likewise, Mu et al. (2018), using a computational general equilibrium framework, document that only the eight sectors planned to be part of the initial execution of the ETS in the PRC are likely to have a much larger mitigation cost. However, the mitigation costs can be minimized by up to 3.3% of real GDP by 2030, but only if other prominent energy-intensive sectors, which account for about an additional 24.8% of total emissions, are included in the system.

**Japan.** The development of Japan’s carbon market can be viewed in two different dimensions. The first is the development of the CDM project, which took effect in 2002–2004 with 12 projects, two of which are being validated. The projects focus on the decomposition of carbon emissions, and are estimated to have 4.8 million tCO₂e per year (Mizuno 2004). The second is the development of the Japanese Voluntary Emissions Trading Scheme (JVETS), which was introduced in 2005 by the Ministry of Environment, Japan, and is aimed at reducing the GHG emission activities in the Japanese companies listed under the Keidanren Voluntary Action Plan (Ministry of Environment, Japan 2012). As of 2012, the JVETS had 389 members, which achieved a great reduction of 59,419 tCO₂ in their total emissions. It was noted that the mean trading price was roughly ¥216 ($2.60)/tCO₂. In addition, the 389 companies that participated in the JVETS recorded a cumulative emissions decrease of 2,217 million tCO₂e, much higher than the planned target of 1,245 million tCO₂e. The JVETS increased further to cover 0.3% of the emissions in 1990 (3.4 million tCO₂/year as observed in the fourth term) (Institute for Global Environmental Strategies, EDF, and International Emissions Trading Association 2016).

Studies have analyzed the ETS linking in Japan. For example, Wakabayashi and Kimura (2018) point out that the Tokyo ETS alone cannot reduce carbon emissions but has other factors, such as energy-saving tools. However, Lu et al. (2013) used a multi-regional model and found that the carbon leakage ratio in the case of full auctioning in the ETS of Japan is not very different from the grandfathering and output-based scenarios and does not reduce carbon emissions.

**Republic of Korea.** In compliance with the Kyoto Protocol, in May 2012 the Korean government passed the Act on the Allocation and Trading of Greenhouse Gas Emission Permits. However, the implementation could not begin until 2014 with the establishment of the ETS’s simple proposal and formulation of the First National Emission Permit Allocation Plan. At the beginning, the Ministry of the Environment decided to allocate about 1,598 million Korean allowance units to 525 companies for 2015 (IETA 2017). The allocation of credits
was successfully completed in August 2016, which was part of the first phase in the first year of compliance.

The second phase of the Korean ETS was introduced in January 2018 and will run through until 2020 with an estimated emission cap of 538.5 MtCO₂ to be considered as against 0.4 MtCO₂ less than the previous year as documented in the National Allocation Plan. At the beginning of 2019, auctioning was considered in the subsectors that failed to meet the standards for trade intensity and further production-related cost due to the ETS. They are estimated at 3% of total volume of allowances auctioned to these subsectors. However, the other subsectors that meet the standards continue to obtain 100% free allocation (Ministry of Environment, Republic of Korea 2018). Basically, in the second phase, free allocation is stretched to eight subsectors rather than the usual three subsectors, namely, cement, refinery, and aviation. Moreover, guidelines have also been developed to allow the use of certified emission reductions generated outside Korea for compliance.

5.4 The Potential Challenges for the Growth of Carbon Markets in Asia

Asia is one of the fastest growing regions of the world. However, the region is also responsible for a major share of global emissions, particularly since the Paris Agreement in 2015. Therefore, in this section, we aim to highlight the challenges that the region is facing to address the issue of climate change without damaging its economic development and prosperity.

Inadequate implementation of Emissions Trading System policy. Adequate policy implementation is indispensable, particularly when beginning to establish a carbon market. The reason for this is that the market relies heavily on it to boost outputs through its ETS policy by binding the emission set to all firms that need to limit their GHG emissions, or by controlling the number of firms based on their commitment to reducing emissions. However, such an emission cap in the Asian carbon market is evidently yet to be introduced. Further, it is also important to highlight that the market demand is not strong enough to help the new industry meet its emission reduction targets.

This may be because the Asian carbon market is still very young and most countries in the region are in the first phase of its implementation. In the absence of full implementation of the ETS policy, it is well documented that most existing ETSs do not cover all sectors and only a few major carbon emitters are included in the ETS policy (Mu et al. 2018). The ETS policy has been driven by chemicals, petrochemicals,
iron and steel, construction materials, paper, air transport industries, nonferrous metals, and electricity. However, the wide sectoral coverage is to estimate the true picture of mitigation costs as compared with the limited sectoral design, which can underestimate the mitigation costs.

**Limited private financing.** Financial institutions have been argued to play an important role in the functioning of the carbon market (United Nations Environment Program Finance Initiative 2011). This does not exclude Asia, where the banking sector plays the major financing role. However, severe capital adequacy requirements and maturity mismatches have been attached to lending. According to Watson et al. (2017), 18 Asian countries received a total of $3.8 billion in climate funds from 17 multilateral climate funds and initiatives for 422 projects and programs in 2016. Among the funds are the PRC’s public climate fund which currently dominates the total investment in climate finance (Wang et al. 2012). An estimated $294 billion was offered by state-owned banks in 2011 and the government climate spending was $41 billion, while foreign and domestic bank loans stood at $10 billion (The Climate Group 2013). Thus, this implies that the major source of climate finance is public funds. It is also important to highlight that the current system cannot divert private investments into the carbon markets. However, this is in contrast to the global practice where private investment is the major source of finance for climate mitigation (Stadelmann et al. 2013; Grubb et al. 2011).

**Lack of a regulatory system.** Going by international standards in carbon-trading management systems, there should be an integrated institutional setup in which the system takes control of regulating carbon trading and its functioning. For instance, in the EU, the carbon-trading management system adopted a two-level organization, which includes central management and environmental protection departments from the governments of members (Knight 2010). Further, it clearly specifies the division of work, and explicitly draws a distinction between rights and obligations. Moreover, coordination and cooperation are ensured to achieve high efficiency in the operation and management of the carbon-trading system.

However, this is not applicable in the Asian carbon market where no specific regulatory institution is responsible for the carbon trade setup. Because of that, it is difficult to set up a reliable and robust monitoring system and reporting and authentication mechanisms, which are the backbone of the success of the carbon market in Europe. So, legal administration continues to be a problem challenging all walks of life in society. Hence, we highlight the fact that currently no regional regulatory system helps strengthen and smooth the functioning of the ETS and its administrative operation. Although the ETS program has been implemented by the respective countries in the region, there is no harmonized system, and it is useful to review and adopt a more unified
regulatory system for the region, which will play a key role in this aspect.

**Corruption, access to information, and local capacity.** Most Asian countries either operate a central economy or are transiting to a market-driven economy. Thus, efforts to attract foreign investors are always faced with problems of corruption and bureaucracy, including poor law enforcement and poor infrastructures, all of which may impact the ease of doing business generally and the execution of CDM projects in these countries. In addition, information on CDM potentials is paramount in the carbon market because it enhances the viability of the market. However, this information is often not accessible. For example, Nguyen, Sousa, and Uddin (2010) argued that national potentials of renewable energy resources have not been investigated and are often not available to users. As such, investors and developers are deterred from investing in the Viet Nam carbon market. As these barriers continue to linger, the awareness of CDM potentials among enterprises, the private sector, public entities, and nongovernment organizations is limited.

### 5.5 Evidence from Previous Studies

We can reasonably divide the existing international literature on carbon emission markets into two segments. One strand of the literature has analyzed the interactions between carbon and energy markets (see, for example, Mansanet-Bataller et al. 2007; Alberola et al. 2008, Fezzi and Bunn 2009, Hintermann 2010). These authors find that energy prices—i.e., prices of oil, gas, coal, and electricity—are important driving factors for carbon futures prices of the EU carbon emission market in phase I. Bredin and Muckley (2011) investigate the association between carbon and energy prices in phase II using a co-integration method. Using the dynamic conditional correlation generalized autoregressive conditional heteroskedasticity model, Koenig (2011) and Chevallier (2012) provide evidence on the dynamics of correlations between carbon and energy markets.

Another strand of the literature examines the theoretical and empirical link between the carbon and financial markets. For example, Daskalakis, Psychoyios, and Markellos (2009) find significant negative links exist between the EU carbon emission market and equity market. Gronwald, Ketterer, and Trück (2011) employ the copula method and find a significant dependence between carbon and stock markets. Koch (2014) considers a multivariate generalized autoregressive conditional heteroskedasticity model to analyze the price interactions among the carbon, energy, and financial markets.

According to the seminal review paper by Babatunde, Begum, and Said (2017), there has been an increasing interest recently in investigating the climate change mitigation policies of the PRC, especially among
academia. Research related to the PRC can be safely divided into three strands, according to the specific aspect the strand focuses on: (i) the reduction of CO$_2$ emission intensity, (ii) the development of renewable energy, and (iii) estimation of the emissions peak.

A vast strand of literature tries to investigate the impact and evaluation of the PRC’s carbon intensity targets, especially using computational general equilibrium models. In a notable proportion of this strand of literature, various optimization methods have been used to look for the optimal mitigation cost path given some carbon intensity targets, treating these targets as emission constraints. In the short run, a noteworthy example is Zhang et al. (2013), who simulated the statistical and economic influences of the PRC’s national emission intensity targets over the sample period of 2010–2015 for the 12th National Five-Year Plan. To their surprise, they can only identify a considerable welfare loss, which ranges from about 1.2% to 1.5% via different disaggregation technologies of reduction targets among various PRC provinces. In contrast, Dai et al. (2011) identified a much smaller (if not trivial) impact. According to the PRC’s Copenhagen commitment, compared with its 2005 level, the PRC will try to reduce its national CO$_2$ emission intensity by 40%–45% before the end of 2020. Dai et al. (2011) concluded that even in the extreme case where the PRC has successfully fulfilled its commitment (which is unlikely given the current situation that its carbon emissions were relatively high by the end of 2016), the PRC’s national GDP loss would only range from about 0.032% to 0.24%.

In the long run, Wang, Wang, and Chen (2009) tried to gauge the economic impact of the PRC’s long-term emission reduction scenario up to 2050. In contrast to the extant literature, they argued that foreseen and unforeseen technological innovations may, to some extent, promote economic growth in the PRC, improve the energy efficiency in the country, and reduce the carbon intensity there. At the same time, Qi et al. (2016) also simulated a long-term scenario for the PRC from 2020 to 2050. They assumed a 3% average annual reduction in CO$_2$ intensity over their sample period, and found that the associated national welfare loss in the PRC would be only 0.9% in 2030, which would increase to 1.6% in 2050. Considering the possible international/national/regional design of the associated carbon ETS in the future, other papers have also tried to gauge the economic impacts of carbon intensity targets in the PRC and obtained different results. For instance, assuming the existence of a counterfactual global carbon emission market, both Zhang et al. (2017) and Qi and Weng (2016) attempted to gauge the economic impact of the PRC’s emission targets by the end of 2020 and 2030, respectively. At the same time, using the sectoral approach as a complement, Mu, Wang, and Cai (2017) went one step further and argued that the
international intended nationally determined contributions could help achieve the committed emission reduction target without suffering from statistically significant economic loss. Focusing on one province, Guangdong, and assuming that the PRC would fully achieve its Copenhagen commitment, Wang et al. (2015) attempted to quantify the economic impacts of the ETS in Guangdong, one of the most developed provinces of the PRC, by the end of 2020. At the same time, focusing on Shanghai city and assuming the PRC’s intended nationally determined contributions commitments will be fully achieved by the end of 2030, Wu et al. (2016) performed an empirical analysis similar to that carried out by Wang et al. (2015). Both Wang et al. (2015) and Wu et al. (2016) identified a negative economic influence in the PRC brought about by achieving the PRC’s emission reduction targets, and provided evidence that the carbon ETS can help mitigate the associated costs when the PRC moves toward its carbon emission reduction target.

Another strand of literature focuses on the social, economic, and environmental impacts of the development of renewable energy in the PRC, especially in the last couple of decades. A notable example is Qi, Zhang, and Karplus (2014), who conjectured that the carbon emission reduction will not be substantial, despite a possibility of many additional renewable power installations in the second decade of the 21st century. Focusing on the impact on the employment rate, Cai et al. (2014) found empirical evidence that more and more jobs would be created due to the development of renewable and/or new energy in the PRC from the second decade of the 21st century. Interestingly, Dai et al. (2016) successfully found more encouraging evidence that carbon emissions would be reduced without incurring substantial macroeconomic costs due to the large-scale development of renewable energy in the PRC. Considering the synergy between emission control and renewable/new energy, Mittal et al. (2016), Cheng et al. (2016), and Duan et al. (2016) provided more affirmative evidence that both the targets of carbon emission reduction and the development of renewable/new energy in the PRC may be achieved at the same time without overly high macroeconomic costs.

To sum up, the extant literature up until now has already documented a range of evidence on the ongoing smooth low-carbon transition of the PRC economy, and most of the evidence is encouraging for both economists and policy makers. However, there is still a gap in the literature, which is the interrelationship among the newly established regional carbon emission exchanges in the PRC, especially considering the exogenous impact from the well-established carbon emission exchanges from outside the PRC. Therefore, we are conducting this project to investigate the potentially existing interrelationship between
arguably the two most important domestic carbon emission exchanges in the PRC, considering the probable exogenous impact from the influential European ETS.

### 5.6 Data and Methodology

#### 5.6.1 Data

This section describes the data in detail. The data set consists of daily prices for the carbon emission markets of the EU\(^1\), Shanghai (SH), and Shenzhen (SZ)\(^2\) over the period of 3 January 2017 to 9 October 2018, since trading volumes in the Shanghai and Shenzhen carbon emission markets were trivial before 2017. All commodity prices are expressed in CNY. Figure 5.1 shows the time series plot for the closing prices in both the Shanghai and Shenzhen carbon emission markets in terms of daily frequency. According to Figure 5.1, there are enough sample variations

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2. We consider the main contract SZA-2016 in the Shenzhen carbon market.
for both the Shanghai and Shenzhen carbon emission markets over our sample period in our data set. Interestingly, the Shanghai carbon emission market has no clear time trend, while that of Shenzhen shows a declining time trend to some extent. We present the summary statistics in Tables 5.1 and 5.2, which further supports our previous observation that there are enough sample variations for both the Shanghai and Shenzhen carbon emission markets over the sample period in our data set.

### Table 5.1: Descriptive Statistics of the Selected Variables (After Logarithm)

<table>
<thead>
<tr>
<th>Variable</th>
<th>Minimum</th>
<th>Maximum</th>
<th>Mean</th>
<th>Standard Deviation</th>
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<td>EU</td>
<td>3.482564</td>
<td>5.299405</td>
<td>4.175204</td>
<td>0.515983</td>
</tr>
<tr>
<td>SH</td>
<td>3.208825</td>
<td>3.749975</td>
<td>3.540342</td>
<td>0.127661</td>
</tr>
<tr>
<td>SZ</td>
<td>2.941804</td>
<td>3.832980</td>
<td>3.389923</td>
<td>0.179254</td>
</tr>
</tbody>
</table>

EU = European Union, SH = Shanghai, SZ = Shenzhen.
Note: This table presents the descriptive statistics of the main variables in the chapter.
Source: Authors.

### Table 5.2: Unit Root Test Results

<table>
<thead>
<tr>
<th>Variable</th>
<th>ADF Unit Root Test</th>
<th>Phillips-Perron Unit Root Test</th>
</tr>
</thead>
<tbody>
<tr>
<td>In levels</td>
<td></td>
<td></td>
</tr>
<tr>
<td>EU</td>
<td>−1.1522</td>
<td>−1.8257</td>
</tr>
<tr>
<td>SH</td>
<td>−1.5387</td>
<td>−2.5868</td>
</tr>
<tr>
<td>SZ</td>
<td>−3.5543***</td>
<td>−4.3637***</td>
</tr>
</tbody>
</table>

| In first-order difference |       |                               |
| EU                      | −7.2938*** | −21.984***                   |
| SH                      | −8.2219*** | −19.829***                   |
| SZ                      | −9.7686*** | −23.093***                   |

ADF = Augmented Dickey-Fuller, EU = European Union, SH = Shanghai, SZ = Shenzhen.
Note: This table presents the results from two-unit root tests (i.e., ADF and Phillips-Perron unit root tests), while *, **, and *** indicate the significance at the 10%, 5%, and 1% level, respectively.
Source: Authors.
To begin our investigation, it is necessary to test the “stationarity of the variables used in this analysis. To determine the order of the “stationarity” of these series, we consider both the Augmented Dickey-Fuller and Phillips-Perron unit root tests. The results in Table 5.2 show mixed findings. The null hypothesis that there is a unit root in levels cannot be rejected for the EU and SH markets but not for the SZ market at the 1% significance level. However, the null hypothesis that there is a unit root in levels is strongly rejected at the 1% significance level for all variables when they are applied to the first-order differences. These findings indicate that our variables have a mixed order of integration and that EU and SH are I (1) and SZ is I (0), which makes our empirical analysis more difficult and hinders many common econometric tools in this project.

5.6.2 Methodology

One aim of this study is to investigate the relationship between the PRC’s carbon emission markets, and between those of the EU and the PRC. Due to the mixed order of integration of our data, we employ the autoregressive distributed lag (ARDL) approach (see, for example, Pesaran et al. 2001) to investigate the impact of the EU carbon emission market on those of the PRC. The ARDL model assumes that the dependent variable is a function of its own past lagged values as well as current and past values of other explanatory variables. For the model, the sample series do not have to be I (1) (see, for example, Pesaran and Pesaran 1997). In addition, the ARDL model in conditional error correction form can integrate short-run adjustment effects with long-run equilibrium without losing information (see, for instance, Jalil and Feridun 2011). Therefore, we can get more information, especially in terms of efficient co-integration relationships, by using this approach (see, for instance, Ghatak and Siddiki 2001). The ARDL \((p, q)\) model can be specified as follows:

\[
y_t = a + \sum_{i=1}^{p} \varphi_i y_{t-i} + \sum_{i=0}^{q} \delta_{1i} x_{1,t-i} + \sum_{i=0}^{r} \delta_{2i} x_{2,t-i} + \varepsilon_t
\]

where \(t = \max(p,q,r)\cdots T\). The optimal lag orders \(p, q,\) and \(r\) can be obtained by minimizing the conventional Bayesian information criterion. For most cases, we find that the optimal number of lags is \(t = 1\), which we use in the analysis. As a result, we restrict all of them that equal one to reduce the noise introduced by the optimal number of lags, and make sure that each term is on the same page.
To investigate the direction of the long-run relationship between the Shanghai and the Shenzhen carbon emission markets in the PRC more effectively, we re-parametrise the above ARDL model in conditional error correction form, which is as follows:

\[
\Delta \ln SH_t = a + \sum_{i=1}^{p-1} \alpha_i \Delta \ln SH_{t-i} + \sum_{i=0}^{q-1} \beta_i \Delta \ln EU_{t-i} + \sum_{i=0}^{r-1} \rho_i \Delta \ln SZ_{t-i} - \gamma (\ln SH_{t-1} - \theta_1 \ln EU_{t-1} - \theta_2 \ln SZ_{t-1}) + \varepsilon_t
\]

\[
\Delta \ln SZ_t = a + \sum_{i=1}^{p-1} \alpha_i \Delta \ln SZ_{t-i} + \sum_{i=0}^{q-1} \beta_i \Delta \ln EU_{t-i} + \sum_{i=0}^{r-1} \rho_i \Delta \ln SH_{t-i} - \gamma (\ln SZ_{t-1} - \theta_1 \ln EU_{t-1} - \theta_2 \ln SH_{t-1}) + \varepsilon_t
\]

where \( \gamma = 1 - \sum_{i=1}^{p} \phi_i \) and the long-run coefficients \( \theta_1 = \frac{\sum_{i=0}^{q} \delta_1 i}{\gamma} \) and \( \theta_2 = \frac{\sum_{i=0}^{q} \delta_2 i}{\gamma} \). In these two formulas above, the shortened forms SH, SZ, and EU stand for futures prices in the Shanghai, the Shenzhen, and the EU carbon emission markets, respectively. These forms of equation provide us with an opportunity to look into both the short-run and long-run relationships at the same time. It is also feasible to include more control variables if necessary.

### 5.7 Empirical Results and Discussion

This section presents the results obtained via the model specified in the previous section. We will empirically examine the impact of the EU carbon emission market on the PRC’s carbon emission markets. The main results of our ARDL model are displayed in Tables 5.3 and 5.4. Several conclusions can be drawn from these two tables. The most important is perhaps that on the interrelationship in the long run as indicated below:

(i) According to Panel A in Table 5.3, a 1% growth in the futures price in the Shanghai carbon emission market increases the futures price in the Shenzhen carbon emission market by 0.5502% in the long run, which is strongly statistically significant at the 5% level.

(ii) According to Panel A in Table 4, a 1% increase in the futures price in the Shenzhen carbon emission market raises the futures price in the Shanghai carbon emission market by 0.4330% in the long run, which is only marginally statistically significant at the 10% level.

The above findings indicate that there is a (at least unidirectional) long-run relationship between the Shanghai and the Shenzhen carbon
emission markets. More specifically, the long-run results for the Shenzhen carbon emission market (Panel A in Table 5.3) show that the Shanghai carbon emission market plays a substantial role in promoting the futures price in the Shenzhen carbon emission market, which is generally positive as expected. In contrast, the EU carbon emission market is not found to have a statistically significant long-run impact on the Shenzhen carbon emission market. However, the long-run results for the Shenzhen carbon emission market (Panel A in Table 5.4) reveal that the Shenzhen carbon emission market is not a statistically significant determinant of the Shanghai carbon emission market at the conventional 5% significance level.

To our surprise, the long-run impact of the EU carbon emission market on the Shanghai and Shenzhen carbon emission markets is also not statistically significant at the conventional 5% significance level. To be specific, for the Shenzhen carbon emission market, the estimated coefficient for $\ln EU$ is $-0.0262$ with a p value of $0.2956$ in Panel A of Table 5.3. For the Shanghai carbon emission market, the estimated coefficient for $\ln EU$ is $0.0035$ with a p value of $0.9202$ in Panel A of Table 5.4.

### Table 5.3: Estimated Autoregressive Distributed Lag Model, Long-Run and Short-Run Coefficients (Dependent Variable: $\ln SZ$, Autoregressive Distributed Lag [1, 5, 1])

| Variable     | Estimate | Std. Err | Z Value | Pr(>|z|) |
|--------------|----------|----------|---------|---------|
| **Panel A: Long-run coefficients** |           |          |         |         |
| $\ln EU$    | $-0.0262$ | $0.0250$ | $-1.0460$ | $0.2956$ |
| $\ln SH$    | $0.5502^{**}$ | $0.2406$ | $2.2870$ | $0.0222$ |
| **Panel B: Short-run coefficients** |           |          |         |         |
| (Intercept) | $1.4073^{***}$ | $0.3346$ | $4.2060$ | $0.0000$ |
| $L\Delta \ln SZ$ | $-0.0483$ | $0.0492$ | $-0.9810$ | $0.3270$ |
| $\Delta \ln EU$ | $-0.0349$ | $0.0395$ | $-0.8840$ | $0.3760$ |
| $\Delta \ln SH$ | $0.0081$ | $0.0996$ | $0.0820$ | $0.9350$ |
| ecm(-1)     | $-0.1102^{***}$ | $0.0241$ | $-4.5710$ | $0.0000$ |

Note: This table reports the estimated coefficients from the autoregressive distributed lag model. The dependent variable is $\ln SZ$. Std.Err denotes the standard error, while Pr(>|z|) denotes the p value. The appropriate lag length was selected based on the Bayesian information criterion, while *, **, and *** indicate the significance at the 10%, 5%, and 1% level, respectively.  
Source: Authors.
The short-run coefficients are shown in Panel B of Tables 5.3 and 5.4. The short-run impact of the EU carbon emission market is also not statistically significant on either the Shanghai or Shenzhen carbon emission market at the conventional 5% significance level. To be specific, for the Shenzhen carbon emission market, the estimated coefficient for $\Delta \ln EU$ is $-0.0349$ with a p value of 0.3760 in Panel B of Table 5.3. For the Shanghai carbon emission market, the estimated coefficient for $\Delta \ln EU$ is $-0.0109$ with a p value of 0.5733 in Panel B of Table 5.4. The reason may be partly to the characteristics of regional trading for carbon emission markets in the PRC, which lead to the strong link between the Shenzhen and Shanghai carbon emission markets, and the insignificant impact of the carbon emission market of the EU on that of Shenzhen and Shanghai.

The short-run coefficients for the key variables are not statistically significant at the conventional 5% significance level, indicating that the short-term fluctuations in the Shenzhen and Shanghai carbon emission markets are irrelevant to each other. To be specific, for the Shenzhen carbon emission market, the estimated coefficient of $\Delta \ln SH$ in Table 5.3 is $-0.0483$ with a p value of 0.3270 while for the Shanghai carbon emission market, the estimated coefficient of $\Delta \ln SZ$ in Table 5.4...
is 0.0035 with a p value of 0.8831. Moreover, in both equations, the estimated coefficient of the error correction term (i.e., ecm[-1] in Tables 5.3 and 5.4) is negative and statistically significant at the 1% level. To be specific, the estimated coefficient of ecm(-1) is –0.1102 and –0.0498 in Tables 5.3 and 5.4, respectively. This negative estimated coefficient of the error correction term implies the speed of adjustment at which a dependent variable returns to equilibrium following a change in the long-run equilibrium relationship (see, for example, Sari et al. 2008). What is more, the results show that the futures price in the Shenzhen carbon emission market, with a larger (in absolute magnitude) estimated coefficient of the error correction term (i.e., ecm[-1] in Tables 5.3 and 5.4), has a faster adjustment rate than the futures price in the Shanghai carbon emission market.

Given the above findings, we argue that no significant short-run relationship exists between the Shanghai and Shenzhen carbon emission markets, but they will influence each other from the standpoint of their long-run relationship. Hence, we suggest that policy makers and government officials enhance the long-term carbon emission market cooperation and linkage in the PRC to deepen the domestic emissions trading system progressively.

Finally, we explore the short-run causalities between the futures prices in the Shanghai and Shenzhen carbon emission markets using the Granger causality test. The results of these short-run causalities are shown in Table 5.5. The research results indicate that no reverse causality exists between the Shanghai and Shenzhen carbon emission markets at the conventional 5% statistical significance level. However, there is a bidirectional relationship between the Shanghai and Shenzhen carbon emission markets at the 10% statistical significance level. For

<table>
<thead>
<tr>
<th>Null Hypothesis</th>
<th>F-Test Statistic</th>
<th>Prob.</th>
</tr>
</thead>
<tbody>
<tr>
<td>The future prices in SZ do not Granger-cause the future prices in SH</td>
<td>0.7870</td>
<td>0.5017</td>
</tr>
<tr>
<td>The future prices in SH do not Granger-cause the future prices in SZ</td>
<td>1.3967</td>
<td>0.2486</td>
</tr>
</tbody>
</table>

SH = Shanghai, SZ = Shenzhen.

Note: This table reports the short-run Granger causalities using one lag in the estimation. We use *, **, and *** to denote the significance at the 10%, 5%, and 1% level, respectively.

Source: Authors.
instance, the p value for rejecting the null hypothesis that the futures prices in Shenzhen do not Granger-cause the futures prices in Shanghai is 0.06766, while the p value for rejecting the null hypothesis that the futures prices in Shanghai do not Granger-cause the futures prices in Shenzhen is 0.05761.

5.8 Concluding Remarks and Policy Implications

In this chapter, we explore the interrelationship between the Shanghai and Shenzhen carbon emission markets using an autoregressive distributed lag model with daily high-frequency data from 3 January 2017 to 9 October 2018. This form of equation provides us with an opportunity to look into both the short-run and long-run relationships at the same time. It is also feasible to include more control variables, if necessary. Using this approach, we provide several new findings.

First, we show that in the long run, there is a unidirectional influence from the futures prices in the Shanghai carbon emission market on the futures prices in the Shenzhen carbon emission market, but not in the other direction. This is statistically significant at the 1% level, and the magnitude is very large. Second, we find a marginal significant short-run influence from the futures prices in the Shanghai carbon emission market on the futures prices in the Shenzhen carbon emission market, and from the futures prices in the Shenzhen carbon emission market on the futures prices in the Shanghai carbon emission market. Finally, to our surprise, in both the long run and short run, we find that the futures prices in the EU ETS market have a very insignificant impact on the futures prices in both the Shanghai and Shenzhen carbon emission markets.

These findings have three important implications. First, our results shed light on the policy formulation of the PRC government on the carbon emission markets to achieve sustainable economic growth and meet carbon reduction targets. Second, our results suggest that the carbon emission markets are, to a very large extent, segmented at the international level; this calls for international cooperation in terms of reducing carbon emission targets. Third, the identification of a significant long-run relationship, but only a marginally significant short-run relationship, suggests that even at the national level the domestic carbon emission exchanges in the PRC are to some extent segmented. This suggests that a unified carbon ETS should be put into effect as soon as possible.

We have to admit that this study has some limitations. Ideally, we should have relied on a well-accepted economic theory to investigate the interrelationship between the Shanghai and Shenzhen carbon
emission markets. Without such a theory, we propose using an ARDL model with daily high-frequency data from 3 January 2017 to 9 October 2018. The choice of carbon emission exchanges is somewhat arbitrary, as we simply choose arguably the two most famous ones, in parallel to the well-established stock trading exchanges in the PRC. Of course, more exogenous variables may be added. However, we doubt that our key results will change, due to the robust nature of our methodology. We believe that it is a fruitful research direction and leave it for future projects.

Further, we make several policy recommendations regarding the efficient functioning of the ETS in Asia with special reference to the PRC, due to its significant contribution to global emissions. More specifically, we suggest that there are several barriers in the Asian region, particularly the weak support for the carbon market. Therefore, it is important to establish a strong regulatory system that should aim to strengthen and support the functioning of the ETS in the region, and perhaps across the major economies. Policy makers and government officials should also give considerable attention to encouraging private investments in the carbon market. There is also a need to support public–private partnership investments in clean energies and energy-efficient technologies, which will have a greater impact on climate change management (Kutan et al. 2018; Paramati et al. 2016; Paramati et al. 2017). Policy makers, with the cooperation of governments, should aim to improve the functioning of the bureaucracy, particularly the departments directly and indirectly linked with climate change initiatives. All these additional policy initiatives will have a greater impact on the functioning and performance of the ETS in the region, as well as in the PRC.
References


6

Valuing the Environment

Euston Quah and Tsiat Siong Tan

6.1 The Need to Value the Environment

Unlike most goods and services, environmental goods and services are not traded in conventional markets due to their public goods characteristics (non-excludable and/or non-rivalry), as well as the fact that they often exist in the form of externalities, beyond the purview of producers and consumers. As such, the environment, which the market cannot directly price, is regarded as non-pecuniary and intangible. Moreover, given that in the absence of intervention no price tag is attached to the environment, market failure ensues, and the environment is inevitably over-consumed and exploited.

While not explicitly priced, intuition points to the fact that people value the environment. We prefer to breathe cleaner air, have access to unpolluted water, live in a predictable global climate, or simply enjoy a scenic view of natural landscapes. Yet, we also apparently are not willing to sacrifice everything else for a better natural environment. Accordingly, it is established that the environment has some positive value but it is not of infinite worth, or “priceless,” as some proponents of the environment might suggest.

Hence, to determine the amount of environmental goods and services to provide or preserve, the question lies in weighing society’s degree of preference for the environment vis-à-vis other goods and services. When constrained by scarce resources, trade-offs must be made. Should we allocate additional resources to improve the environment or invest in economic growth? The underlying objective when making this decision is to ensure that everyone is as satisfied as possible or make as many people better off as possible. In other words, the choices made should ideally maximize society’s well-being. This notion of welfare maximization raises its own set of ethical and philosophical questions, warranting an extensive debate outside the scope of this chapter.
Achieving socially optimal levels of environment and economic efficiency first requires the measurement of preferences for the environment before these can be compared with preferences for alternative goods and services, and whether an informed trade-off can be made may be ascertained thereafter. In this endeavor, money may be used to measure unpriced preferences, as unappealing or even unacceptable as this might sound to some. Valuing the environment in monetary terms allows us to make a quantifiable comparison conveniently between environmental goods and market goods. For example, if a preservation project yields $1 million in environmental and spillover benefits, at a cost of less than $1 million that could otherwise be spent on other market goods, the only logical conclusion would be that this project is desirable for society. A distinction can be drawn between the monetary values of and the monetary values generated from environmental goods and services, as will become clearer in subsequent parts of this chapter.

The most obvious alternative to monetary valuation of the environment would be to conduct a referendum whereby each individual casts one vote according to his or her unobserved preferences. The main shortcoming of such a democratic majority vote, relative to using money as a measuring rod, is that the intensity of preferences is not captured. In the one-man-one-vote referendum, an individual with a high degree of concern for an environmental objective is no different from another individual who cares for the environmental objective but only to a small extent. Monetary values, however, can encompass this vital piece of information.

The need for a value measure is primarily to inform the decision-making and policy-making processes, thereby justifying the allocation of limited resources between competing uses—whether residential, commercial, social, environmental, or otherwise. Putting a value on the otherwise intangible environment at the very least advocates responsibility and precaution when striving for economic development and growth. This importance has been amplified as we increasingly acknowledge the severe impacts of environmental degradation. Quantifying the stock and flow of natural capital, in addition to physical and human capital, allows us to chart our path carefully toward sustainable development. It empowers us to address environmental problems better through policy and management measures (e.g., green taxes), conduct green national accounting, and design compensation schemes in cases of pollution incidents, thus correcting the market failure. Environmental valuation also plays a traditional role in policy impact assessments and has conventionally been incorporated into cost–benefit analysis (see Mishan and Quah, 2007; 2020), cost-effectiveness analysis, environmental impact
assessment, risk–benefit analysis, and so forth. Conducted ex ante, valuation is useful in stipulating and warning of potential losses and required compensation before the damage has occurred. Conversely, ex post valuation advises future policy making.

6.2 Total Economic Value

Total economic value broadly constitutes use value and non-use value. Use value can itself be subdivided into direct use value, indirect use value, and option value, while non-use value comprises existence value and bequest value. When valuing the environment, one must, therefore, first clearly discern if the total economic value is elicited or if only one or some components of the total economic value is/are captured. Note that, in addition to the typology of value illustrated below, there are alternative definitions and classifications of total economic value, such as drawing a distinction between anthropocentric and non-anthropocentric values, and intrinsic and instrumental values (Hargrove 1992), which will not be discussed here.

Direct use value has the most straightforward interpretation: the benefits derived from the actual use of environmental goods and services. Examples include increased tourism revenue as a result of the preservation of natural parks and revenue gains from fishing by improving water quality in fishing grounds. Indirect use value refers to the benefits from ecosystem functions, such as the regulation of the climate or the environment’s role as a waste sink. Lastly, option value (Bishop 1982; Smith 1987) is the potential benefit through options for use in some future period. Individuals might prefer to preserve a specific plant species that has no current use as they believe that it could potentially provide remedies for ailments in the future, subject to some probability of scientific discoveries.

Non-use values are conceptually more abstract, and they are also more difficult to measure empirically. Existence value (Walsh et al. 1984) is the satisfaction derived from knowing that a certain environmental good or service exists. An individual might never have or intend to ever meet a Bornean orangutan. Nevertheless, he/she still hopes to ensure the species’ existence, given the information that they are critically endangered. The individual could be driven by some moral satisfaction of knowing that the animal does not become extinct or of feeling that he/she has fulfilled an innate responsibility to protect wildlife and the natural environment, and as such the society is ameliorated in some way. Bequest value, on the other hand, refers to the utility derived from passing down environmental benefits to future generations, thus ensuring that they will enjoy access to environmental
goods and services. This can be construed as a form of intergenerational altruism. In both existence value and bequest value, the individual attributing his/her value does not use the environmental good in question. In fact, bequest value is a form of existence value.

6.3 Methods of Valuation

The environment can be valued mainly using non-demand curve approaches (Section 3.1) or demand curve approaches (Section 3.2). Non-demand curve approaches have no true welfare measure; the willingness to pay (WTP) for (or willingness to accept [WTA]) an environment good or service cannot be determined without obtaining a demand curve and the area beneath it. While not ideal, non-demand curve approaches nevertheless provide useful information for policy making and will be briefly discussed in the following section. Non-demand curve approaches include replacement cost, mitigation behavior, opportunity cost, and dose–response methods. In contrast, demand curve approaches, as the name suggests, provide welfare measures when environmental improvement or degradation occurs. Demand curve approaches can be further classified into stated (or expressed) preference methods and revealed preference methods.

Stated preference methods measure welfare using income-compensated Hicksian demand curves. Through the use of survey questionnaires, respondents explicitly place values on environmental assets by stating their WTP or WTA. The most commonly stated preference method is the contingent valuation method (CVM). Revealed preference methods elicit consumer surplus welfare measures through uncompensated Marshallian demand curves. Revealed preference methods, unlike stated preference methods, rely on market-based transactions. Environmental preferences are revealed through individuals’ purchase of market goods that are related to the consumption of environmental goods. The value of unpriced environmental attributes can then be measured using methods such as the travel cost or the hedonic pricing method.

Assigning monetary values (also referred to as shadow prices) to environmental goods and services fundamentally involves measuring preferences and utilities through real or hypothetical exchange transactions. This chapter will also discuss two other approaches (Section 3.3)—the Pairwise comparison approach and benefits transfers—which differ from the usual demand and non-demand curve approaches and might prove particularly useful in developing countries. In the following sections, the conduct and applications of each valuation
method will be detailed. It will become apparent that each method has its advantages and limitations, and choosing the one best suited for use is a judgment based on the circumstances and policy objectives at hand. Adopting the most appropriate method of estimation is crucial, with the goal that the estimated value should be as close to the true values as possible.

### 6.3.1 Non-Demand Curve Approaches

**Replacement Cost, Mitigation Behavior, and Opportunity Cost Method**

In deciding whether a project that results in the loss of environmental assets is justified, a blunt tool could measure the cost of replacing, restoring, or maintaining these environmental assets by some other means. This is also known as the replacement cost method, which uses replacement costs as a proxy for the benefit of recovering environmental assets. For example, instead of measuring the widespread impacts of a potential oil spill, this method looks at the cost of cleanup as an indicator of the cost of the oil spill. While replacement costs are theoretically distinct from the benefits of replacement, the method can be useful as a benchmark when environmental assets are deemed too difficult or costly to value directly. This method is also particularly relevant when circumstances dictate that the quality of the environment must be maintained, or environmental preservation must occur for some reason. Some case studies incorporate the replacement cost method to estimate the value of aquatic species and sites of considerable importance to indigenous people in Australia (Jackson et al. 2014), and to assess the cost of soil erosion in the upper Mabaweli watershed of Sri Lanka and the Nyaung Shwe Township in Myanmar (Gunatilake and Vieth 2000; San and Rapera 2010).

The mitigation behavior method is like the replacement cost method, except that it uses expenditures incurred to avert the effects of lower environmental quality as a yardstick for policy making. For example, in the case of air pollution, one might deem expenditures on air purifiers and face masks as mitigation behaviors. In a case where the building of a dam hinders the migration of salmon to their hatchery sites, the cost of building artificial fish ladders to mitigate these impacts and allow the fish to swim upstream to breed could be used.

The opportunity cost method also provides no direct valuation of environmental goods and services and uses the opportunity cost of a proposed project that degrades the environment. The opportunity costs of a project or policy that is environmentally detrimental are
then compared with the necessary benefits to render it efficient. Concomitantly, the social opportunity cost of a given project represents the foregone social gains derived from the next best alternative use.

**Dose–Response Method and Value of Statistical Life**

The dose–response method establishes the relationship between a human’s (or more precisely any living being’s) response and increased exposure to environmental stress, such as an additional dose of pollution. The method has been widely applied to study the impacts of air pollution, with Quah and Tay (2003) providing a good illustration.

Quah and Tay (2003) applied the dose–response method to study the economic cost of particulate air pollution on health in Singapore. The study used PM$_{10}$ (particulate matter with an aerodynamic diameter of 10 micrometers or less) as an indicator of the health risk of air pollution. This pollutant category has previously been identified as responsible for most health problems arising from air pollution, including respiratory symptoms and diseases, carcinogenesis, and premature death. Data on the ambient concentration of PM$_{10}$ were first collected. Air pollution epidemiological literature and further empirical studies were then used to draw connections between pollutant emissions (using PM$_{10}$ as a proxy) and human health effects, with a dose–response function formulated as a result. The dose–response function can be represented using the following equation (Ostro 1994; Rowe et al. 1995; Quah and Tay 2003):

\[
dH_{ij} = a_{ij} \times dA_j \times POP_i
\]

where

- $dH_{ij}$: Change in risk of health impact $i$ due to pollutant $j$
- $a_{ij}$: Slope of the dose response function for health impact $i$ due to pollutant $j$
- $POP_i$: Population at risk of health impact $i$
- $dA_j$: Change in ambient concentration of air pollutant $j$

The coefficient in the dose–response function will likely vary across existing medical studies, and if the policy maker were to draw from the literature, the use of a set of coefficients is recommended, i.e., a lower, central, and higher coefficient. The range of coefficients to be applied could be set at one estimated standard deviation apart.

Next, the impacts on morbidity and premature mortality can be expressed as below:
\[ \Delta \text{Morbidity} = b_i \times \Delta PM10 \times POP_i \]

\[ \Delta \text{Mortality} = c_i \times \Delta PM10 \times POP_i \times 0.01 \times \text{crude mortality rate} \]

where

- \( b_i \): Morbidity coefficient from the dose–response function for health impact \( i \)
- \( c_i \): Mortality coefficient from the dose–response function for health impact \( i \)
- \( \text{POP}_i \): Population exposed to risk of health impact \( i \)

With the estimated changes in morbidity and premature mortality, the final step requires the assignment of monetary values to these changes. This in turn requires a value to be placed on human lives, a rather contentious issue, especially from an ethical or philosophical perspective. Nevertheless, the objective here is simply to list and explain some economic methods to value human lives, which is useful not only to assess environmental impacts and health risks but also in related applications, such as determining tort compensations.

Economists assign monetary values to statistical lives instead of the specific lives of individuals. The value of statistical life (VoSL) involves aggregating a small change in fatal or nonfatal risk across a population. For instance, a 1% reduction in the risk of death that affects 100 individuals is equivalent to one statistical case. This one statistical life is not identified with a particular individual: it is not the intrinsic value of life nor the benefit of saving a specific life with certainty, but the amount an average individual is willing to give up to reduce his or her morbidity or mortality risks. This value could, for example, be used to inform policy makers who are considering a project that on average saves one statistical life within the population.

One method to estimate the VoSL is to survey respondents directly and elicit their WTP for a reduced health risk, or WTA for the opposite, subject to their budget constraints in reality. If an average respondent is willing to pay $10,000 for a 1% reduction in his or her risk of death within the current time period, the VoSL can be computed as being $10,000/1\% = $1 million. The same logic applies to nonfatal risks, which use the value of statistical life year or the value society places on reducing the risk of premature mortality or prolonging life. If the average respondent is willing to pay $10,000 when the expected gain in longevity from reduced pollution is 1 month, then the WTP for 1 full year of life would be $10,000 \times (12/1) = $120,000. The conduct of such surveys is an application of the CVM, which will be elaborated in Section 3.2.2.
Another way to measure the inherent trade-off one makes between risk of death (or illness) with the consumption of other goods and services is to look at the exchange one makes between income earnings and the risk of job-related death (or illness). To illustrate, a job with a 0.2% risk of death might warrant a wage premium of $1,000 over another job with a 0.1% risk of death, all things being equal. This implies that the employee must be compensated $1,000 for taking on the additional 0.1% fatality risk. If this represents the average employee and if labor markets are perfect, then the VoSL can be calculated as being $1,000/0.1% = $1 million. Viscusi and Aldy (2003) have comprehensively reviewed this approach.

Alternatively, one could choose to use medical expenditures (direct costs) and illness-associated lost productivity (indirect costs) as a proxy for the value of life. This is also known as the cost of illness approach, which directly accounts for the human capital value of health, that is, the productivity returns. Furthermore, the money spent on health care could otherwise be spent on other goods and services, costs that are averted in good health. Dividing the cost of illness by mortality rate then gives the cost of life lost. The main shortcoming of this approach, as opposed to the two methods presented before, is that it yields an underestimation, as we omit the intrinsic value of life. In reality, we expect people to value life per se as living life generates happiness and meaningful experiences. We also expect that patients would not completely return to their initial healthy state with medical treatments and would be willing to pay even more to avoid pain and suffering from illness, as well as to remain in a healthy state rather than be cured in an unhealthy state. Note that health care subsidies, which distort prices, or health insurance coverage, which leads to moral hazard problems, further complicate the measurement.

Other similar approaches to valuing a statistical life exist, such as observing revealed behaviors and WTP in insurance markets. One may also look at preventive expenditures instead of medical or insurance expenditures, such as price premiums for residential housing with lower rates of pollution, or WTP for helmets to reduce motorcyclists’ injury risks. Discounted lifetime wages could also be indicative of a life’s worth, at least in terms of its contribution to society from the provision of labor and human capital. Note that instead of using WTP to measure the trade-off between health-risk reductions and the consumption of other goods and services, one may instead measure the trade-off between different health states of varying durations (Hammitt 2002). This is the underlying principle of the nonmonetary measures of disability-adjusted life years and quality-adjusted life years, whereby weights are assigned to life years of discrepant health quality. For example, a year lived in illness might be worth half a year in good health. By extension, using
the various valuation methods, disability-adjusted life years and quality-adjusted life years can be monetized as well (Lvovsky et al. 2000).

### 6.3.2 Demand Curve Approaches

#### Revealed Preference Approaches

**Travel cost method**

Revealed preference approaches rely on observable market-based transactions of goods that are related to the consumption of intangible environmental goods. The travel cost method uses travel costs as a proxy for price, which can be directly observed in monetary terms, to gauge the demand and, hence, the value of recreational sites, including natural parks and reserves. This method assumes that if an individual chooses to incur the cost of visiting a site, then this cost is at least the value the individual attributes to the site, including the environmental goods and services it provides. A relationship can thus be derived from the observed variations in travel costs and frequency of visits. As with the law of demand, a higher travel cost (price) will correspond with a lower number of visits (quantity demanded) (Clawson and Knetsch 1966; Duffield 1984; Randall 1993; Adamwowicz et al. 1994).

Deriving the demand curve of a natural park then allows us to calculate the value visitors place on, say, the natural park by calculating the area under the demand curve. Note that travel costs primarily comprise fuel cost and travel time cost, the latter of which may be estimated using the average wage rate. However, if the journey in itself is an enjoyable activity to the individual, then travel time is not really an incurred cost but an exchange of time for the joy of traveling. Including travel time cost in this case could lead to an overestimation of the site’s value.

The researcher conducts on-site questionnaire surveys to collect information on the round-trip travel cost and number of visits to the amenity. Concentric circles are first drawn to define the different zones around the amenity used by people living at varying distances from the site. Each concentric zone corresponds to a different round-trip travel cost; outer zones have higher travel costs and vice versa. The number of visitors from each zone is then divided by the population in each zone to get the frequency of visits by each zone. The relationship between travel cost and frequency of visits can subsequently be estimated with an econometric model, using frequency of visits as the dependent variable and travel cost as the explanatory variable. The negative coefficient estimated tells us by how much the frequency of visits falls when travel cost rises. Thereafter, a hypothetical range of admission fees to the
amenity is imposed. The hypothetical admission fees serve as a proxy for price and should be in a reasonable range as compared to similar sites. At each alternative admission fee, the schedule of total trips can be easily calculated with the regression coefficient presented above by adding the admission fee to the travel cost. With that, the relationship between the price of the amenity (the hypothetical admission fee) and the quantity demanded (the total number of visits from all zones), which is essentially the demand curve, can be derived. If the data are collected for all visitors within a single day, the area under this demand curve represents the daily value of the site, which can later be augmented to give the annual or even perpetual value of the site.

The steps described above represent the basic zonal travel cost method (Clawson and Knetsch 1966). This valuation approach is relatively inexpensive to apply, has potential for a large sample size, yields results that are relatively easy to interpret and explain, and is based on actual behavior as opposed to stated preference methods. There are also further methodological improvements to be considered. Besides fuel and travel time costs, travel costs should also include accommodation spending, excess meal expenditure, purchases of goods exclusively for the purpose of the trip, and/or existing admission fees to the amenity (in this case, one would be using a hypothetical increase or decrease in admission fee in the steps described above) where appropriate. Instead of the zonal travel cost method, one could also adopt the individual travel cost method or the random utility travel cost method.

For the individual travel cost method, the researcher collects data on travel costs from individual visitors instead of assuming that visitors from the same zone are identical. Therefore, survey questions would include more details, such as the specific residence location of the visitor, the person's income, socioeconomic variables, demographic variables, other locations visited on the same trip, amount of time spent at the site, the level of satisfaction derived from the trip, perceptions of the environmental quality of the amenity, the number of visits per year, the availability of substitute sites that the respondent might have alternatively visited, and so on. This circumvents some of the limitations of the zonal travel cost method, such as assuming equal opportunity cost of time for people in the same zone, or that individuals all react to an increase in travel costs in the same way (i.e., same regression coefficient). If travel costs are incurred not only to visit the amenity but also for other purposes (e.g., a tourist visits multiple attractions on the same trip), then the travel cost incurred should not be the value for one amenity but apportioned across multiple attractions using the time spent on each.

The individual travel cost method yields higher empirical precision with a multi-regression model but requires more extensive data
collection and complicated statistical analysis. This is even more so for the random utility travel cost method. Based on the random utility theory, individuals choose a site to visit out of a set of possible amenities and make trade-offs between the site's quality and the cost of traveling to maximize one's utility. This means that data are required from more than one site, accounting for site-specific characteristics such as environmental quality. The random utility travel cost method utilizes conditional probabilistic models to characterize demand, requires more assumptions, and is more complicated to operationalize.

It is also important to note that the travel cost method only calculates the value of the site to actual visitors who use it and, hence, disregards non-use values. For environmental reserves with endangered species, one should be careful not to conclude that the findings from using the travel cost method are axiomatically the total economic value (see Section 2) as non-use values in this case are likely to be very large. A low frequency of visits (user rate) could also indicate that the inclusion of non-use values might affect the results. People who live near a park and thus incur almost zero travel costs, thanks to a conscious house purchase decision or otherwise, might also value the park more than it seems. Lastly, when differences in travel costs among visitors are not sufficiently large, this method would also prove inapplicable.

**Hedonic pricing method**

The second form of revealed preference approach is the hedonic pricing method, which again uses existing market information to determine the value of environmental goods and services. The basis of this method is that market prices (such as for real estate) are directly affected by their associated environmental characteristics. In purchasing a residential property, the consumer seeks to maximize his/her utility by making a trade-off between the price of the property and the benefits they can derive from it. These benefits not only include factors such as the property's proximity to one's workplace and local amenities, access to the transportation network, and its size and design, but also environmental benefits such as its landscape aesthetics and scenic views. Conversely, one would pay less for houses near sources of pollution or other not-in-my-backyard facilities for the same reason (Quah and Tan 2002).

Knowing that different variables—although not individually priced—are implicitly included in real estate prices allows us to control for these variables. If we were to control for all variables except environmental characteristics, then any remaining price differential between real estate must be due to underlying uncontrolled differences in environmental characteristics (Rosen 1974; Freeman 1993; Smith 1993; Clark and Nieves 1994). Similarly, when controlling for environmental characteristics, the
regression coefficients give the value that consumers implicitly place on these variables. Adoption of the hedonic pricing method has become increasingly feasible with technological advancements, enabling researchers to use geographical information systems to map locations digitally and accurately link physical locations with access to amenities and other variations in desirable and undesirable environmental traits, such as varying exposure to noise pollution from highways. Jiao and Liu (2010) used a geographic field model-based spatial hedonic pricing method, finding that apartments situated close to recreational spaces of the Changjiang River and the East Lake in Wuhan in the People’s Republic of China were significantly more expensive than counterparts that did not meet these criteria.

One major advantage of the hedonic pricing method is that it measures both use and non-use values. However, a prerequisite of the method is a well-functioning property market that readily reacts to changes in demand and supply. This is often not the case, as moving to a new house is a long-term decision hindered by various barriers, many of which cannot be readily observed, and the data are not necessarily collected easily. Income constraints will likely restrict purchase (or rental) decisions of high-cost items like housing. In many states, housing prices are also highly distorted by government taxes and subsidies. Econometrically, not only is it difficult to identify all the relevant explanatory variables that affect real estate prices but problems of multicollinearity also further confound the analysis. For example, the degree of air pollution is expected to be highly correlated to the visibility of scenic views. Finally, the findings are restricted only to (environmental) variables that are identified as varying across properties.

**Stated Preference Approaches**

**Contingent valuation method**

In the 1989 Exxon Valdez incident, the CVM was validated by the United States government and judicial systems to assess the environmental damage of the oil spill and its impacts on beaches, coasts, and wildlife habitats. The CVM has since become the most common and widely adopted environmental valuation method in the literature.

The economic principle of the CVM can be represented as follows (Quah and Tan 1999):

\[
U(Q^0, y^0) = U(Q, y^-) = U(Q^+, y^-) = U(Q^-, y^0 + WTA) = U(Q^+, y^0 - WTP)
\]
where

\[ U(Q^0, y^0) \] is the current utility level without the hypothesized change

\( Q^0 \) is the initial quantity/quality level of environment

\( y^0 \) is the initial income level

- represents decrements

+ represents increments

WTA is willingness to accept

WTP is willingness to pay

As depicted above, given that utility is a function of environment and income, all things being equal, a decrement in environment quality/quantity can be offset by an increment in income, and vice versa. Changes in levels of welfare can be made possible with monetary payments. It is, therefore, possible to create a hypothetical market for intangible environmental goods and services by stipulating an environmental improvement (or degradation) that an individual would be willing to pay for (or willing to accept monetary compensation for) in an attempt to maximize his/her utility.

This stated preference approach requires directly surveying respondents and asking them to state their values for non-market environmental goods and services. To do so, the researcher first provides a detailed description of hypothetical scenarios involving changes in environmental quality or quantity (alternative states of the situation that differ from the status quo), which are the projected effects of a program or policy being considered. If the environmental good or service were to be made available, an individual would be willing to pay an amount of money up to a certain point, which is his/her maximum WTP. The individual is not willing to pay beyond this amount to trade for the environmental good; hence, this is the true value assigned by the individual to the said good. The same argument follows for a removal of an environmental good or service, except that the minimum WTA is to be elicited in this case. With individuals’ WTPs or WTAs, the average WTP or WTA can then be calculated and multiplied by the affected population to give the total value of the environmental asset. Deriving the total economic value (both use and non-use values) becomes possible, but whether the result is the total economic value or not ultimately depends on the design of the survey questions. For example, van Kooten and Bulte (2000) intentionally isolated the elicitation of existence values by only asking respondents their WTP to prevent the extinction of wildlife species. Meanwhile, Quah and Tan (1999) employed the CVM to determine the total economic value (including use value, and
option and existence non-use values) of landscape scenery, specifically
the East Coast Park in Singapore. Use and non-use values are separated
using CVM questions asking about the number of visits to the park,
and respondents that seldom or never visit the park are deemed to be
expressing their non-use values.

The value elicited using the CVM is contingent on the given
hypothetical scenario, as well as how each respondent interprets and
assigns it a value. Therefore, it may prove difficult to elicit a true value;
to do so, it is crucial that the presented hypothetical scenario be realistic,
precise (including information on the duration and scope of impact),
and fully understood by the survey respondents. Using photographs or
audio clips to describe the scenario to the respondents might even be
useful, although the researcher must be cognizant that this can prove
counterproductive if it distorts the scenario or restricts respondents’
understanding.

Designing the survey and how it is to be conducted is paramount
to the success of the CVM. There are various ways that CVM questions
can be asked. The most straightforward method is to use an open-ended
question, i.e., “How much would you be willing to pay for/given that ...
?” This allows the respondent to state any amount he/she wants,
which could result in a high variance in values if respondents in general
have little or no idea as to the value they would attribute to the subject
matter. The researcher can derive the average WTP or directly plot a
downward-sloping demand curve and calculate the area underneath
using a regression or otherwise. Furthermore, control variables such as
demographic characteristics might be added into a multiple regression
model, and the same applies to the other methods described below. We
could, for example, find significant differences in the average WTP of
men versus women.

Open-ended questions are often not advised owing to their lack of
statistical robustness. This has been mentioned as one of the guidelines
of the experts’ panel, including Nobel Laureates Kenneth Arrow
and Robert Solow as chair to the National Oceanic and Atmospheric
Administration (Arrow et al. 1993). Another method is the referendum
(take-it-or-leave-it) approach, in which each respondent receives one
discrete choice question with one predetermined amount of monetary
value, e.g., “Would you be willing to pay $x for ...? Yes/No?” A range
of predetermined amounts, usually at regular intervals, is given across
respondents, such that some respondents are asked in terms of $x, $y,
or $z. The collected data provide information on how many people are
willing to pay at least the given values, and the lower-bound WTP can
be estimated using various estimators, such as a Turnbull estimator
(Turnbull 1976). To illustrate simply, if 20% of the respondents are
willing to pay at least $40, and 80% are willing to pay at least $20, the average WTP at the lower bound works out to be $24. Moreover, the data could be analyzed econometrically using probit/logit models and/or ordered probit/logit models. The demand curve (using survival analysis) can be derived as more people would be willing to make small amounts of payment relative to large amounts. The downside is that the sample size must be larger. See Subade and Francisco (2014) for an application of the referendum approach to elicit the non-use values of the Tubbataha Reefs in the Philippines.

The payment card method is another way of conducting CVM studies. Instead of one predetermined amount per respondent as in the referendum approach, each respondent faces the entire range of predetermined values, i.e., “Would you be willing to pay the following amounts for ...? Option 1: $x. Option 2: $y. Option 3: $z.” Ordered probit/logit regressions are then used for analysis. As with the referendum method or other methods that provide some form of predetermined values, it is vital to note that given values are arbitrary and should be justified with existing studies or theoretical support or, at the very least, are intuitively acceptable. Moreover, one would expect that the payment card method tends to yield results with WTP clustering at the lower amounts, and WTA clustering at the higher amounts. One may refer to Arin and Kramer (2002) for an application of the payment card method to value divers’ WTP to visit marine sanctuaries in the Philippines.

The fourth approach is in the form of auction bidding (Davis 1963; Cummings et al. 1986; Hanemann 1994). For WTP, the respondent is asked whether he/she is willing to pay $x first. If yes, would he/she be willing to pay $(x + \varepsilon)$? If yes again, would he/she be willing to pay $(x + 2\varepsilon)$, and so on, until the respondent answers “no.” This switch point indicates the maximum WTP. For the minimum WTA, the same logic applies, with a downward bidding with successively lower prices instead of an upward bidding. This approach is more cumbersome to conduct and usually requires a face-to-face interview or an online survey, but the demand curve can be directly derived and the results can be analyzed using a multiple regression model. Setting $\varepsilon$ to be a reasonable amount also reduces survey fatigue, as a very small $\varepsilon$ could result in a long survey process and repetitive questions. (See Yu and Abler [2010] for an application of the auction bidding CVM to ascertain the WTP for unpolluted blue skies in Beijing, People’s Republic of China).

Another method proposed by Haneman, Loomis, and Barbara (1991) concerns the use of a double-bounded CVM. The respondent is asked if he/she is willing to pay $x. If yes, would he/she be willing to pay $2x? If no, would he/she be willing to pay $x/2? Each respondent receives one starting price but a range of starting prices (e.g., $x, $y, or $z) is randomly
assigned to different respondents. Again, the arbitrary predetermined starting values must be properly set but, in this case, a larger range of values can be easily tested. We could then collect information regarding the number of people lying within each range of value: whether bid \( < \frac{x}{2}, \frac{x}{2} < bid < x, x < bid < 2x, \) or bid > 2x (and the same for \( y \) and \( z \)). Probabilistic and/or ordered probabilistic regressions are used for analysis.

For the payment card and double-bounded formats, follow-up questions directly asking for the maximum WTP or minimum WTA could always be included, as the exact WTP or WTA is not already elicited. Extreme bids and zero (protest) bids (Portney 1994; Kristrom 1997), which could be due to respondents who were noncooperative or who misunderstood the survey, should be omitted from analysis. Refusing to pay for the environmental damages caused by others or a perception that it is the government’s responsibility to pay for and resolve the issue indicates protest bids.

There have been extensive debates for and against the use of the CVM. Arguments against it are largely due to either the hypothetical nature of the methodology, or respondents’ innate behavioral biases. The former can be addressed or mitigated through the proper design of surveys and choice of research questions. We will, therefore, focus on the behavioral responses in the CVM that could distort expressed preferences.

First, WTP and WTA measures are not equivalent. The WTA has been empirically observed to exceed the WTP. For example, Hanley (1989) found that the average WTA for a ban on the burning of straw is about four times as high as the average WTP against the same ban. Behavioral economists explain this discrepancy as loss aversion. People generally value losses more than gains. The WTP, which measures the benefit of a welfare improvement (a gain), is therefore smaller than the WTA, which measures the cost of a welfare deterioration (a loss). Hence, framing CVM questions as a loss or gain must be carefully considered. The choice of measure should depend on the current assignment of property rights. If the victims of pollution supposedly have the right from pollution, the researcher should be asking the affected general public or local residents their WTA as compensation for a lower environmental quality as a result of the proposed policy change. If instead the polluters have the right to pollute at status quo, CVM questions should ask for the victims’ WTP to avoid a worsened environmental state due to the same policy change. One should also note that the public might be more familiar with and accepting of the concept of the polluter-pay principle (WTA for environmental losses) rather than the victim-pay principle.
Valuing the Environment

Furthering the discussion, negative changes can also occur in the domain of gains and, therefore, should not be treated as a loss but as a reduction in gains. The choice of WTP and WTA thus becomes even trickier. For a detailed discussion on welfare measures concerning gain and loss domains, see Knetsch et al. (2012).

The way in which monetary payments are to be made hypothetically also matters. This is additionally known as the payment vehicle bias. For example, respondents might be willing to pay a smaller amount when raising taxes but willing to pay a larger amount when the money is contributed to a conservation fund for the same purpose. The solution to this discrepancy is to adopt the payment vehicle that is most realistic for the context. On a similar note, respondents may deceive the surveyor if they think that their responses might, for example, affect the actual tax rates they face in the future. This strategic non-revelation of preferences could be an attempt to free-ride on other people’s payments or to profit from compensation schemes. The researcher must assess and make a judgment as to the severity of strategic bias, report any potential overestimation or underestimation, and adjust accordingly. One way to detect strategic bias is to check the distribution of elicited values using the Shapiro-Wilk W statistic (Brookshire et al. 1982; Cummings et al. 1986; Carson and Mitchell 1993; Maddala 1997). Anchoring bias and embedding effects (Knetsch 1998; Chuenpagdee et al. 2001) also constitute a potential problem in the CVM when the final WTP or WTA is centered on the starting bid provided, which respondents use as a mental reference point. Using a set of different starting bids and randomly assigning these starting bids to different respondents resolves this issue.

Finally, people might assign the same monetary value to a part of an environmental good (e.g., saving a specific wildlife species) and the entirety of the environmental good (e.g., preserving the entire forest with various wildlife species, including the above-mentioned species). This irrational behavior, known as the part-whole bias, is explained by mental accounting. People tend to allocate a portion of their income or wealth for different categories of goods, such as a fixed budget for environmental goods and saving wildlife species. One way to mitigate this problem is to have respondents work out their overall budget for the environment before asking for their WTP (Turner and Adger 1995). Alternatively, one could use the CVM to evaluate a basket of environmental goods rather than trying to value each environmental good individually or conduct studies in both manners for comparison.
6.3.3 Other Approaches

Pairwise Comparison Approach
The reliability of the conventional demand curve valuation methods detailed above have been highly debated because these methods are centered on monetary assessments. At times when we might not be as confident of the final monetary values derived, the pairwise comparison approach or the damage schedules approach is preferred. This nonmonetary method has received limited attention and can measure whether one environmental good is worth more than another. It values environmental assets in relative terms rather than in absolute nominal terms. The approach aims to develop an interval ranking of relative importance for a set of intangible environmental issues and policies, derived from respondents’ judgments of environmental degradation. Since it is only an indicator of relative social preferences, it does not face the problems of WTP and WTA non-equivalence and loss aversion (Champ and Loomis 1988; Knetsch 1990; Loomis et al. 1998).

Table 6.1 describes the six main steps of conducting a pairwise comparison study. In each survey question, survey respondents are presented with pairs of environmental losses, alongside some hypothetical descriptions that invoke intrinsic feelings, and must choose the environmental loss they would prefer to suffer. The sets environmental losses are of different types and of different levels of damage. The choices are to be randomized to control for order effects. If the choice set does not contain an excessive number of objects, n, all

<table>
<thead>
<tr>
<th>Step</th>
<th>Description</th>
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<tbody>
<tr>
<td>Step 1</td>
<td>Develop hypothetical but realistic scenarios of different levels of damage to the resources of different levels of activities that can cause such damage.</td>
</tr>
<tr>
<td>Step 2</td>
<td>Use the paired comparison method to present these scenarios (generally as a questionnaire booklet).</td>
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<tr>
<td>Step 3</td>
<td>Conduct the survey, asking the respondents to complete the survey on their own.</td>
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<tr>
<td>Step 4</td>
<td>Analyze the data using the variance stable rank sum method.</td>
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<tr>
<td>Step 5</td>
<td>Test for any significant difference between the rankings of relative importance of resources obtained from various interest groups, and aggregate responses as appropriate.</td>
</tr>
<tr>
<td>Step 6</td>
<td>Suggest policy responses according to the relative importance of the resources.</td>
</tr>
</tbody>
</table>

possible pairs can be presented to each respondent \( n(n-1)/2 \). Repeated measures for each element within the choice set yield more reliable estimates than single-point estimates, as in the CVM (Peterson and Brown 1998). However, one should acknowledge that the method is neither feasible nor satisfactory where the objects are many (David 1998). The rationale behind using the pairwise comparison approach is that respondents are usually more comfortable in comparing objects pairwise and making a discrete choice, rather than stating a monetary value for intangible goods with which they might be unfamiliar. The method is also more intuitively appealing than an ordinal ranking of all objects, especially when differences between objects are subtle and preferences might be conditional on the full set of alternatives that respondents see at one point in time.

Upon completing the survey, binary choices are aggregated across respondents to give an interval scale, an ordering of preferences among the elements within the choice set. This is achieved by giving a preference score for each item, which is the number of times the respondent prefers that item over other items. The preference score is aggregated across respondents and summarized by the variance stable rank method (Dunn-Rankin 1983), i.e., the proportion of times each item is chosen relative to the maximum number of times it is possible to be chosen. With that, we multiply this proportion by 100 and derive a collective judgment scale of the relative importance of all the items (Chuenpagdee et al. 2001) ranging from 0 to 100. Nonparametric statistical tests of significance are used to determine the degree of concordance among individual survey respondents and between different respondent groups. One example is Kendall’s coefficient of concordance, in which a statistically significant coefficient denotes consensus in ranking among the respondents. Correlation across subgroups can also be compared to uncover heterogeneity in preferences.

The pairwise comparison approach is primarily applied to ensure that policies correspond to public values and, hence, can be successfully implemented. Rutherford, Knetsch, and Brown (1998) used the pairwise comparison approach to uncover the expected damage from oil and toxic liquid spills of various types and magnitude. Quah, Choa, and Tan (2006) also used the method to compare four environmental problems perceived by Singaporeans as the most important environmental goods: degradation of the coastal and marine environment, polluted air, ozone depletion, and an unhygienic environment pertaining to food and water. In this study, two levels of environmental qualities, moderate and severe, were tested for each item.

If the findings from the pairwise comparison study show consistent choices, they indicate that decision makers are rational. Inconsistent
choices are detected by circular triads, in which A is preferred to B, B is preferred to C, but C is preferred to A. This could be a result of intransitive preferences, random choice, or respondents’ cognitive limitations. It is also possible that inconsistent choices are meaningful as a result of objects that are multidimensional with different characteristics or different levels (Kahneman et al. 1999). Respondents might perceive these choices as a close call and might be indifferent between the items. To avoid this problem, it is advisable to conduct a pilot survey and check the viability of the survey options. Alternatively, inconsistent choices may be repeated at the end of the survey without an explicit indication to ascertain the preference switches for inconsistent choices. If inconsistent responses are resolved during this retrial, the implication is that intransitive preferences are not the cause of the issue but close calls and indifference between objects. Coefficients of concordance and correlation can be tested for their sensitivity to the inclusion or exclusion of intransitive observations.

The pairwise comparison method yields various advantages (Quah et al. 2006). It is less costly in terms of time and money than other primary research methods and can deal with intractable valuation projects. The method is also flexible, as new scenarios of environment losses can be added by expanding the damage schedule through interpolation and extrapolation from the existing set of environmental goods measured. It effectively compares multiple values of environmental goods without multiple studies in a standardized manner, and the results are easy to interpret. The researcher can also choose to include monetary elements (sums of money) within a choice set to elicit monetary WTP or WTA. As long as some form of comparison in terms of the importance or severity of the objects provided can be made, a scaling can be derived (Sunstein 1994).

**Benefits Transfer**

Benefits transfer (Freeman 1984; Quah and Toh 2011) is another useful valuation approach that involves the adaptation and generalization of information from existing research to a different setting. Existing primary research and studies are referred to as study cases/sites while the setting in which the information is adapted is called the policy case/site. The policy site may differ from the study sites in terms of economic, biophysical, temporal, and/or spatial situation (Freeman 2003; Wilson and Hoehn 2006).

Benefits transfer might be selected because (i) primary valuation is not warranted, (ii) it is too costly to conduct, (iii) there is a lack of expertise to conduct primary data collection, and/or (iv) there is immediate urgency to make a policy decision. To conduct a benefits
transfer, a thorough literature review of relevant studies is crucial. Only with a sufficient number of studies can the adaptation of information yield precise and robust estimates. Given that benefits transfer essentially draws from other valuation studies, it faces the same potential problem of measurement errors. In addition, it is subjected to transfer errors (errors when generalizing across different contexts), especially when adapting information to a setting that is notably different.

Benefits transfer can be categorized into two types: value transfer and function transfer. Value transfer involves a direct application of summary statistics from study cases to the policy case, adjusting when necessary. The summary statistic could be WTP or WTA measures, or even demand elasticities. The adjustments to be made include a discrepancy in environmental impact between study and policy cases, a different affected population, currencies and inflation, and so on. Function transfer (Loomis 1992) involves the application of a statistical function rather than direct use of the summary statistic. Compared to the former, function transfer requires that more extensive adjustments be made through the statistical function to reflect the characteristics of the policy case but yields more precise and robust estimates as the differences in site characteristics are more effectively considered (Brouwer 2000).

Value transfer can be further separated into three types: (i) transfer of point estimates, (ii) transfer of measures of central tendency, and (iii) transfer of administratively approved estimates. A transfer of point estimates typically uses a range of point values from various existing study cases. The shortcoming of this method is that the study and policy sites should ideally be similar in terms of characteristics, including the geographical location; the baseline state of the environment; the degree of environmental change; the composition of the population; and other market, institutional, and cultural characteristics. These assumptions are often not satisfied. Transfer of measures of central tendency uses the mean or medium of the estimates in, or the confidence interval of, study cases. The decision to use the median over the mean is especially apt if study cases have outlier estimates that might skew the latter. Lastly, transfer of administratively approved estimates is the simplest approach in value transfer. However, estimates of these study cases have often (if not always) undergone the government’s evaluation and approval, and so the process by which these estimates are endorsed and published might not be entirely objective. Table 6.2 lists the steps in conducting a transfer of point estimates. The logic of conducting a transfer of measure of central tendency and transfer of administratively approved estimates is the same (refer to Rosenberger and Loomis 2003 for details).
### Table 6.2: Conducting a Point Estimate Transfer

<table>
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<th>Step</th>
<th>Description</th>
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| Step 1 | Define the policy context.  
This definition should include various characteristics of the policy site, what information is needed, and in what units. Policy site characteristics include the location, the type of environmental good, the availability of substitutes, the affected population (which may include both users and non-users and their socioeconomic status), and so forth. The degree, direction, and timing of change in environmental assets must also be quantified, which first requires determination of the baseline state of the environment. |
| Step 2 | Locate and gather original research outcomes. Conduct a thorough literature review and obtain copies of potentially relevant publications.  
The researcher should conduct a keyword search by country or region, type of environmental asset, valuation technique, year, etc. |
| Step 3 | Screen the original research studies for relevance. How well does the original research context correspond to the policy context? Are the point estimates in the right units, or can they be adjusted to the right units? What is the quality of the original research?  
The key is to maximize scientific soundness (the methodology and assumptions), relevance (the similarity in context), and richness in detail (the data description and information collected) (Desvousges et al. 1998). Note that for intranational benefits transfers, adjustments for inflation should be made. For international benefits transfers, adjustments for currency and income elasticity of willingness to pay or willingness to accept should be adjusted. If information on income elasticity is incomplete, a sensitivity analysis of income elasticity between 0.5 and 2.0 could be considered. Otherwise, income elasticity could be assumed to be equal to one. |
| Step 4 | Aggregation over environmental goods and services, affected population, and duration of impact.  
This aggregation, therefore, gives us the valuation estimate in the policy case. |

Source: Largely reproduced and adapted from Rosenberger and Loomis (2003) and Smith (2014).

Function transfer can also be further classified into benefit function transfer and meta-analysis function transfer. Benefit function transfer is straightforward. It applies the regression coefficients from an existing benefit function (from a single study case) to the summary statistic of the policy case. The explanatory variables are the characteristics that affect the value estimate in both the study and policy sites, and this is a judgment to be made by the researcher. Using the same regression coefficient also implicitly assumes that both populations react in the same way toward the value of the environmental asset, which might not be empirically true (VandenBerg et al. 2001). On the other hand, the meta-analysis function transfer uses regression coefficients from
multiple study sites. There is a clear advantage of not being restricted to one study site. The meta-analysis function transfer approach explicitly includes methodological explanatory variables, such as the method of valuation specific to each study, into the regression model, which allows for control of a large number of possible confounding variables. The number of studies to be included is, however, a trade-off between relevance and amount of information. Nelson and Kennedy (2009) comprehensively reviewed over 140 meta-analyses involving the economic valuation of the environment.

6.4 Conclusion

Various tools and methods are used to value the environment, and these are continually being improved and developed. This chapter has highlighted these valuation methods; nonetheless, they are not exhaustive. The choice of valuation method ultimately depends on the researcher’s judgment: certain methods are more feasible and appropriate for certain study conditions, and no method is without its flaws. It is important to think through the motivation and design of the study, and to report the findings and their limitations transparently.

The importance of valuing the environment is compelling, as people become increasingly aware of the environmental challenges we face today. Valuing the environment is even more important in developing countries that possess a large concentration of the Earth’s natural resources and assets. For economic growth to converge with the developed world, developing countries must sustainably tackle the twin goals of rapid development and environmental preservation. Simultaneously faced with greater budgetary constraints than their developed world counterparts, the relative lack of financial resources at governments’ disposal necessitates difficult and prudent trade-offs. Environmental valuation, along with cost–benefit analysis, aids developing countries in making these choices in an informed manner.

The differing circumstances in labor, goods, and financial markets under which developed and developing economies operate have no bearing on the fundamental principles underlying cost–benefit analysis. However, in applying the principles, certain valuation techniques commonly used in developed countries are not appropriate for developing countries (see Quah [2013] for a detailed discussion). Indeed, most revealed preference approaches, including hedonic pricing and the travel cost method, require strong assumptions of rationality, perfect information, and perfect mobility to be valid (Quah and Ong 2009), while stated preference approaches, including the contingent valuation method, are susceptible to many behavioral effects (Kahneman and
Knetsch 1992; Carson et al. 2001) and methodological biases. In the context of a developing nation, such flaws may be magnified. In the example of the national park, if fuel were distributed through a rationing system in a developing country, then the private cost of traveling would be very difficult to determine and the demand curve obtained through typical travel cost techniques would be inaccurate. For stated preference approaches, behavioral effects may be more pronounced in developing economies owing to people’s relative lack of experience of participating in survey research. List (2003) has shown that behavioral effects are, at least in part, brought about by a lack of experience with decision-making circumstances. Therefore, the magnitude of behavioral biases in stated preference approaches is likely to be much more significant in developing nations. Methodological biases in stated preference approaches also tend to be larger in developing nations because of the general lack of trained interviewers (Hanley and Barbier 2009). One common problem is the inability of both interviewers and interviewees to differentiate between willingness to pay and ability to pay. Such misunderstandings are further exacerbated by cultural and linguistic differences. In addition, surveys typically carry significant costs that cash-strapped governments will be hard-pressed to cover. Thus, particularly for developing nations, these two valuation techniques have obvious pitfalls that may render results dubious.

If a primary study is required, the paired comparison approach may prove to be the best solution for developing countries as it avoids the obvious flaws of the other two methodological classes (Quah et al. 2006). Given that a paired comparison uses surveys, like stated preference methods, it avoids the need for strong assumptions as required by revealed preference methods. It also overcomes the key behavioral effect that plagues contingent valuation methods, which is known as the endowment effect. Paired comparison also offers a third reference point: that of the selector. As no real or perceived loss occurs in this case, behavioral effects like loss aversion, which can affect the results of a WTA survey, are avoided (Kahneman et al. 1999). Where a primary study is not needed, benefits transfer can prove to be a low-cost approach in terms of money and time.

When conducting environmental valuation and cost–benefit analysis in developing countries, it is also important to not entirely forego intragenerational and intergenerational equity in the pursuit of efficiency. For intragenerational equity, one should note that developing countries generally lack governmental channels, such as progressive taxation and estate taxes, to redistribute wealth and prevent the income gap from widening too much or too quickly. In fact, prevalent corruption, a chronic problem for most developing nations, specifically
prevents the formation of such channels because it is often in politicians’ interests to line the pockets of their donors from the business sector. Furthermore, income inequality is generally a greater problem for developing nations than for developed nations. When ranked by their Gini coefficients, the 10 countries with the highest income inequalities are all developing nations, while the majority of the 10 countries with the lowest income inequalities are developed nations (United Nations Development Programme 2016). One commonly proposed strategy is to apply weights to costs and benefits to reflect the relative importance of monetary values to different social classes. Benefits or costs accruing to low-income groups may be multiplied and thus magnified, and projects in their favor will thus have a greater likelihood of being approved. For intergenerational equity, there is a tendency for current generations to bias environmental decisions against future generations. Like labor and goods markets, financial markets in developing economies are also weaker than those in developed economies. Private banks in developing countries usually wield considerable monopolistic power, which they may exploit by charging interest rates above what a free market would produce (Yildirim and Philippatos 2007). This implicates the issue of temporal discounting when dealing with future benefits and costs, as the social discount rate should ideally consider both the opportunity cost of capital and a society’s time preference. Intertemporal discounting must be done properly to avoid downplaying the future society’s welfare and to quantify accurately the transfer of resources across generations. We need to consider whether future gains are to be sacrificed for immediate losses or if current costs are to be incurred for future benefits.

Finally, when using environmental values to guide public policy and to devote and divert funds among competing needs, one should always remember that the conversion or destruction of environmental assets is often an irreversible process. Furthermore, the value of the ecosystem in its entirety, along with its life support functions, is not simply the sum of all environmental goods. The dimensions of environmental goods are profound and their relationships with the ecosystem may not be immediately apparent. Nevertheless, despite its limitations, economic valuation of the environment provides useful information, such that the public policy choices made are commensurate with society’s welfare, and it is the first and critical step to guide us. Valuing the environment is never perfect but some valuation is generally better than none.
References


PART III

Globalization and the Environment
7

Trade and the Environment: Recent Evidence and Policy Implications

Brian R. Copeland

7.1 Introduction

The interaction between trade and environmental policy is a continuing source of controversy. International trade provides a channel that can be harnessed to improve environmental outcomes, but it can also exacerbate environmental problems. This chapter provides a short review of recent developments in the trade and environment literature with a focus on what we have learned from the empirical evidence. This is followed by a discussion of the implication for policy in both developed and developing countries.

There have been several reviews of the trade and environment literature (see, for example, Brunnermeier and Levinson 2004; Copeland and Taylor 2004; Levinson 2010; Copeland 2011; Cherniwchan et al. 2017; Cole et al. 2017b; Dechezleprêtre and Sato 2017). Inevitably there will be some overlaps but this review will focus on relatively recent work (although some attention to older work is needed for perspective) and will emphasize available work relevant to Asian economies.

The chapter is organized in three main sections. Section 2 considers what we know about the effect of trade on environmental outcomes. This includes discussion of recent work that studies the environmental effects of firm-level responses to trade. Section 3 reviews empirical evidence on the effects of environmental policy on trade and investment flows. This is followed by a brief summary of what we have learned (Section 4), while Section 5 discusses policy implications. Issues related to trade and climate change are discussed throughout the chapter where appropriate.
7.2 How Does Trade Affect Environmental Outcomes?

Trade affects the environment through many channels. Changes in the level and mix of consumption and production induced by trade affect pollution emissions and the sustainability of renewable resources. Imports of intermediate goods and capital equipment influence production techniques, which in turn affect pollution intensities. Technology transfer induced by trade also affects emission intensities. Trade affects production in different regions of a country differently, which can increase pressure on the environment in some places and reduce such pressure in others. Firms are also affected differently. Some expand and some contract; new firms enter, and some firms exit. All these affect environmental outcomes. Trade affects the demand for transport services, which directly affects pollution emissions; the spread of invasive species is influenced by trade patterns; and the competitive pressures of international trade influence the political process that determines the implementation and enforcement of environmental policy.

7.2.1 Theory

Any change in emission levels can be decomposed into scale (overall level of economic activity), composition (the share of different activities in the economy’s production), and technique (emission intensity) effects (Grossman and Krueger 1993; Copeland and Taylor 1994). If the pattern of production and emission intensities is held constant, an increase in scale will increase emissions. If scale and emission intensities are held constant, a shift in production toward polluting industries at the expense of clean industries will also increase pollution. Finally, if the scale and composition of output are held constant, adoption of cleaner production techniques (lower average emission intensities) will reduce pollution.

Trade affects emissions through all three channels. Much theory and evidence indicate that trade will raise income and stimulate economic growth—that is, trade leads to an increase in the scale of output. However, theory and evidence also suggest that the demand for environmental quality increases with income. If governments are responsive, higher income can lead to more stringent environmental policy that reduces emission intensities (a technique effect). The net effect on pollution could go either way. A large literature on the environmental Kuznets curve (EKC) was stimulated by the Grossman and Krueger (1993) paper. It examined how scale, technique, and composition effects interact to determine the effect of economic growth on environmental outcomes.
The evidence suggests that the effects of growth depend on various factors, such as the stage of economic development, type of government, natural resource abundance, and whether growth is biased toward clean or dirty industries (a composition effect).

For those interested in the interaction between globalization and the environment, one of the challenges is to isolate the effects of trade on the environment from the effects of growth. Economic growth is a policy objective of virtually all governments and so a finding that trade affects environmental outcomes because it affects rates of economic growth has not been the central focus of the literature on trade and environment. The key issue of interest for the purposes of this chapter is whether the growth path for an open economy is more or less polluting than that for a more closed economy—that is, does trade affect the environment differently from economic growth? Since trade and growth both generate scale effects, this leaves composition and technique effects as the two key channels via which trade can yield different types of environmental outcomes from growth. The literature has explored both of these.

7.2.2 Comparative Advantage, Composition Effects, and Industrial Pollution

The role of comparative advantage in influencing the environmental effects of trade was one of the questions addressed many years ago in Copeland and Taylor (1994). They show how trade driven by comparative advantage can lead to effects of trade on the environment that differ systematically from those induced by economic growth. I first illustrate this point using a simple example based on Copeland and Taylor (1994) and then review work on the more general issue of how trade-induced composition effects influence environmental outcomes.

Copeland and Taylor (1994) used a simple Ricardian trade model with a continuum of industries that differ in pollution intensity. Governments are benevolent and richer countries endogenously implement more stringent environmental policies than poor countries because consumer demand for environmental quality increases as income rises. We use a simple specification of preferences that yield an income elasticity of the willingness to pay for environmental quality that is equal to 1.

We first showed that in this model, neutral economic growth (stimulated, for example, by a productivity improvement) has no effect on environmental quality. Growth increases pressure on the environment: pollution will rise if there is no environmental policy. However, because the income elasticity of the demand for environmental quality is 1, governments respond to neutralize the effects of growth on
the environment by tightening up environmental policy. The scale and
technique effects exactly offset one another.

Trade yields a very different outcome. Polluting industry is
concentrated in low-income countries with weaker environmental policy
and clean industry is concentrated in countries with more stringent
environmental policy. Pollution increases in poor countries, and falls in
rich countries; overall, there is a net increase in global pollution because
polluting production shifts to countries where emission intensities are
higher due to weaker environmental policy. The model illustrates a stark
version of the pollution haven hypothesis, which states that trade tends
to exacerbate environmental problems in poor countries with relatively
weak environmental policy.

It is striking that this model predicts that free trade will increase
world pollution, even though neutral economic growth would yield no
increase in pollution. It illustrates a channel via which the growth path
for an open economy could be more polluting than the growth path for
a closed economy.

The model outlined above is obviously very stylized but it provides a
simple illustration of how composition effects induced by international
trade can be important in affecting environmental outcomes. More
generally, if trade is driven by comparative advantage and if pollution
intensities differ across sectors, then trade will affect environmental
outcomes via composition effects. Countries with a comparative
advantage in polluting industries will experience an increase in pollution
and countries with a comparative advantage in clean industries will
experience a fall in pollution. In the simple model outlined above,
income-induced policy differences determine comparative advantage.
In fact, many factors influence comparative advantage—technology
differences, human capital differences, infrastructure, capital
abundance, weather, and so on. But whatever sectors a country has a
comparative advantage in, the effect of trade will depend on whether
they are more or less pollution intensive than average for that economy.
If they are more pollution intensive than the average sector, then trade
will tend to increase pollution emissions. This yields hypotheses that
can be explored with empirical work.

The pollution haven hypothesis is that polluting industry is
concentrated in countries with weak environmental policy—that
is, differences in environmental policy across countries are a key
determinant of trade patterns. If true, then trade will exacerbate
environmental problems in developing countries. As discussed below,
ethe evidence for this hypothesis is weak.

A more robust hypothesis is that composition effects induced
by trade driven by comparative advantage will increase or decrease
pollution in countries based on their comparative advantage. One version of this hypothesis is developed in Antweiler, Copeland, and Taylor (2001) and Copeland and Taylor (2003). They consider the interaction between two sources of comparative advantage—capital abundance and pollution policy. If polluting industries are capital intensive (this is true for pollutants such as sulfur dioxide [SO₂]), then rich capital–abundant countries will tend to have a comparative advantage in polluting industries. However, if rich countries have a relatively stringent environmental policy, it will tend to give them a comparative advantage in clean industries. The two sources of comparative advantage work against each other, and so the pattern of trade depends on which effect is stronger. An important implication is that rich countries may export pollution-intensive goods despite their stringent environmental policy; if this is the case, the pollution haven hypothesis fails.

There has been relatively little work directly testing the pollution haven hypothesis. A large empirical literature studies the effects of environmental policy on trade flows. However, most of that literature addresses the narrower question of whether environmental policy is one of the factors affecting trade and investment flows. Recent evidence (discussed later in this chapter) finds that more stringent environmental policy tends to reduce net exports of pollution-intensive goods. Copeland and Taylor (2004) refer to this as evidence in support of a pollution haven effect. However, this is not enough to provide support for the stronger pollution haven hypothesis. If capital abundance and technology differences are more important than environmental policy in influencing firm location and trade flows, then the pollution haven hypothesis will fail.

Antweiler, Copeland, and Taylor (2001) study the effects on openness to trade on SO₂ concentrations in cities. They use an international panel of data on air quality in cities, similar to the data used by Grossman and Krueger (1995). They estimate a model that studies the role of scale, technique, and composition effects in explaining the variation in SO₂ concentrations across cities and time. This allows them to estimate the pure composition effect of increased openness to trade, holding scale, technique, and other factors that determine the composition of industrial production constant. They obtain several striking results. First, the sign of the composition effect varies across countries. The composition effect of trade tends to increase pollution in some countries and reduce it in others. This is what theory predicts (countries cannot all have a comparative advantage in the same sectors). Second, the sign of the composition effect tends to be positive in high-income countries and negative in low-income countries. This indicates that the pure composition effect of trade tends to raise pollution in rich countries
(with stringent environmental policy) and lower it in poor countries (with weaker environmental policy). This is contrary to the pollution haven hypothesis, which predicts the reverse. Third, the estimated composition effects are small. This indicates that once we control for scale, per capita income, and capital abundance, specialization induced by trade has relatively little effect on pollution. Fourth, estimated technique effects are quite large: high-income countries with a large scale of production also tend (on average) to have stringent environmental policy to counteract the effects of scale. Finally, when scale, technique, and composition effects are aggregated, openness to trade tends to reduce pollution for the average country in the sample.

Why does the pollution haven hypothesis fail in this case? The results of Antweiler, Copeland, and Taylor (2001) are consistent with the hypothesis that capital abundance matters more than pollution policy for trade patterns. Capital-abundant countries have a comparative advantage in SO$_2$-intensive production. Their relatively stringent environmental policy dampens but does not reverse this. Cole and Elliott (2003) provide evidence consistent with this interpretation.

Moving beyond the pollution haven hypotheses, the broader question of how openness to trade affects environmental quality has received considerable attention. Using a similar approach but with emissions data, Cole and Elliott (2003) find that trade tends to reduce organic water pollution (emissions affecting biochemical oxygen demand [BOD]) but increase emissions of nitrogen oxides and carbon dioxide (CO$_2$). Their results for SO$_2$ emissions are inconclusive. Frankel and Rose (2005) account for the endogeneity of trade and find results similar to Antweiler, Copeland, and Taylor (2001): trade tends to reduce SO$_2$ pollution.

Managi, Hibiki, and Tsurumi (2009) use an instrumental variable approach to account for the endogeneity of both trade and income, along with more recent data; they also consider short- and long-run effects. Their key finding is that the effect of trade varies with pollutants and between countries of the Organisation for Economic Co-operation and Development (OECD) and non-OECD countries. As with Cole and Elliott (2003), they find that trade tends to reduce BOD emissions throughout the world. However, they find that the long-run effects of trade on SO$_2$ and CO$_2$ emissions are different for OECD countries and for non-OECD countries. Trade tends to increase SO$_2$ and CO$_2$ emissions in non-OECD countries and reduce emissions in OECD countries. The main reason they find for this is that the net scale and technique effect increases emissions in non-OECD countries and reduces emissions in OECD countries. That is to say, growth leads to a stronger emission-reducing policy response in OECD countries than in non-OECD
countries, a result that is consistent with the EKC literature. The pure trade-induced composition effect of trade on BOD emissions is positive for OECD countries and negative for non-OECD countries. The trade-induced composition effects for SO$_2$ and CO$_2$ are positive for the average OECD and non-OECD countries, but tend to be larger for OECD countries.

As discussed earlier, in assessing the effects of trade on the environment, there is an issue of what the counterfactual is. The result that trade tends to increase SO$_2$ and CO$_2$ emissions for the average non-OECD country is driven largely by the growth-inducing effects of trade. The pure composition effects of trade are small and are found to be smaller for non-OECD countries than for OECD countries. The empirical papers reviewed above suggest that while the rate of economic growth might differ between countries that are more or less open to trade, there is little evidence that the “greenness” of the growth path is significantly affected by how open or closed a country is to trade.

Several other studies estimate the effects of trade on various pollutants but do not attempt to identify the pure composition effects separately. A summary of these studies appears in Shahbaz et al. (2017). Results are mixed and vary with pollutants and the mix of countries studied.

A more robust finding (at least for OECD countries) is that the composition effects of trade tend to be small. Studies that attempt to estimate and identify the pure composition effects of trade (Antweiler et al. 2001; Cole and Elliott 2003; Managi et al. 2009) have all tended to find small composition effects. Another way to assess composition effects is to measure them directly. Levinson (2009) uses data on emissions from United States (US) manufacturing (at the four-digit industry level) and adopts an accounting procedure to calculate scale and composition effects. Technique effects are then calculated as a residual. The scale of manufacturing output increased by about 24% from 1987 to 2001, while air pollution emissions fell. The author looks at several pollutants; here the focus is on SO$_2$ (patterns are similar for other pollutants). SO$_2$ emissions fell by 27%, so there is a 51-percentage-point fall in emissions that must be accounted for by a combination of composition and technique effects. Levinson (2009) finds that the composition effect can account for only about 12% of this fall in emissions and that international trade can account for only 3 of those 12 percentage points—that is to say, the trade-induced composition effect is small. The major explanation for a fall in emissions is a fall in emission intensities; that is, the technique

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1 The following discussion draws partly from Cherniwchan et al. (2017).

There are very few estimates of the composition effects of trade for developing countries. Because such countries undergo structural change during the development process, one might expect that trade-induced composition effects would be larger for such countries. Managi, Hibiki, and Tsurumi (2009) estimate the composition effects of trade for three pollutants and find they are smaller for non-OECD countries than for OECD countries, a result that is surprising. However, Barrows and Ollivier (2016), using an accounting approach in the spirit of Levinson (2009), find that composition effects for carbon emissions are large for India. They find that shifts in the mix of goods production explain more of the change in carbon emissions than do changes in emission intensity. Martin (2012) also finds relatively large composition effects for carbon emissions in India. Carbon emissions differ from many other pollutants in that they are a global pollutant; therefore, in the absence of international coordination, one would expect technique effects to be weaker for carbon emissions because global free rider problems are in abatement. However, carbon emissions are correlated with other local air pollutants for which there are greater incentives for unilateral policy. Further investigation into the role of composition effects in contributing to the path of emissions over time in developing countries would be a fruitful area of work.

Studies that estimate composition effects tend to use country-level data. A finding that these effects are small for countries as a whole may mask important regional variation, especially if regions within a country differ significantly in their comparative advantage and industrial structure. Bombardini and Li (2016) use an innovative method$^2$ to identify the effects of export shocks on pollution in the People’s Republic of China (PRC). They construct a measure of revealed comparative advantage in polluting manufacturing industries for prefectures in the PRC and combine this with measures of sectoral changes of external export market access for the PRC (increases in openness). This allows them to create a set of measures of export shocks for the pollution-

$^2$ Their method is based on the approach used by Autor, Dorn, and Hanson (2013) to study the effects of trade shocks on labor market outcomes in the US. The approach has been used in other contexts in the literature, but this is the first application to environmental outcomes that I am aware of.
intensive manufacturing industry that have regional variation. Their results are quite striking. They find strong evidence that increased exposure to trade results in increased levels of local air pollution in regions with a comparative advantage in the polluting manufacturing industry. Moreover, they are able to identify increases in infant mortality due to the increased pollution.

Bombardini and Li (2016) do not attempt to measure the pure composition effects of trade: their estimates yield the full effect of trade, which includes the effects of scale, technique, and changes in the composition of output. Nevertheless, the work provides quite compelling evidence that regional differences in comparative advantage lead to regional variation in the effects of trade on pollution that can be quite large. For a global pollutant such as carbon emissions, regional variations are not of particular concern because all that matters is the effect of trade on the aggregate flow of emissions (carbon is a globally uniformly mixed pollutant, so the damage from emissions is not much affected by where the emissions come from). However, for local air pollution, regional variation matters. Even if, on average, trade has only a small aggregate effect on local air pollution when averaged across a country, a finding that trade has large effects in certain regions suggests that the effects of trade on pollution can be quite damaging. That is to say, one channel through which trade affects the environment is that regional comparative advantage and agglomeration effects can result in a trade-induced concentration of emissions in particular regions, leading to large declines in air quality and increased health problems in affected locations.

### 7.2.3 Comparative Advantage, Composition Effects, and Natural Capital

Another way that trade driven by comparative advantage can have potentially large regional effects on environmental outcomes is via its effects on the sustainability of renewable resources, or, more generally, on natural capital. Examples include fisheries, forests, water, soil, and habitat to support biodiversity.

Many renewable resources are overexploited because property rights or conditions of access to the resource are not well defined or enforced. The long-run sustainability of the resource is threatened in such cases because of externalities: individuals may lack incentives to conserve the resource because one agent’s decision to refrain from harvesting to protect the resource stock can be undermined by some other individual who increases harvesting.
Trade can significantly increase pressure on resource stocks. The
incentive to harvest depends on demand for the harvested goods. In the
absence of trade, local demand is what matters. However, in free trade,
global demand determines outcomes. This means that a movement
to free trade can have potentially very large effects by concentrating
demand on local resource stocks. For open access resource stocks,
Chichilnisky (1994) and Brander and Taylor (1997) have shown that
trade can lead to welfare losses because it leads to long-run depletion of
renewable resource stocks.

The effects of trade in their models operate via composition effects:
trade shifts economic activity to the harvesting sector in regions with
a comparative advantage in renewable resources. While the role of
composition effects in driving outcomes is one source of similarity in the
effects of trade on renewable resources and industrial pollution, there
are also significant differences.

Industrial pollution is often treated by policy makers as a trade-off between environmental outcomes and income. Although increased
industrial activity may increase emissions, it will also increase income\(^3\) and stimulate economic growth. For developing countries, the trade-off may come down on the side of growth. Moreover, economic growth
can both stimulate the demand for environmental quality and provide
the wealth needed to fund infrastructure and other activities needed
to support effective environmental policy. Increased pollution may therefore be tolerated in the short run, but a shift to increased pollution
abatement is prompted by policy changes in the long run as growth
occurs. This is the idea behind some theories of the EKC (see, for
example, Lopez 1994; Copeland and Taylor 2003; Ch. 3).

In contrast, the effects of trade-induced harvesting can lead to
resource depletion and collapse very quickly. Furthermore, the resource
can take a long time to recover. This means that policy makers may not
have the luxury of waiting for the long-run fruits of economic growth
to provide the means to support conservation policies in the future. If
resource collapse can happen quickly, it may be too late for conservation.

Moreover, while increases in industrial pollution may be
accompanied by both short- and long-run increases in income, this is
not the case when excessive harvesting depletes natural capital. As
a resource is depleted, the long-run income it generates shrinks. This
means that while a trade-induced resource boom may generate short-
run income gains, these gains may be short-lived. Once the resource is

\(^3\) Industrial pollution can yield long-run health effects that may ultimately reduce productivity (and income) if workers get sick. But these effects are often long term
and the worker productivity effects may be difficult to tie directly to pollution.
depleted, the flow of income falls, and less wealth is available to devote to conservation and environmental policies in the future.

Resource depletion is not a necessary outcome when an export shock that stimulates harvesting occurs. It depends on the policy regime. Fisheries in countries such as Iceland and New Zealand are quite healthy despite being exposed to trade-induced demand pressures. This is because they are protected with harvest quotas. Hence, the effects of trade on the sustainability of resource stocks depend on the interaction between comparative advantage and the policy regime. A further complication is that the policy regime may change endogenously in response to the trade regime because changes in the value of the resource stock alter the incentives to invest in management, an idea that goes back to Demsetz (1967). This issue is explored in Copeland and Taylor (2009), who find that trade opportunities that increase the value of the resource stock can induce a transition to better management for some resources but not others (depending on parameters such as the intrinsic natural growth rate of the resource stock and the enforcement power of regulators).

Finally, it is important to note that conservation activity can be a source of comparative advantage, a point emphasized by Brander and Taylor (1997). Consider two countries, one with weak conservation policy and one with strong conservation policy. If demand for the harvested good is strong in the absence of trade, the resource will be severely depleted in the country with weak conservation policy but will be healthy and sustainable in the country with good conservation policy. When trade opens up, the country with good conservation policy will have a comparative advantage in the harvested good: it will have a healthy supply of the resource because the stock has been conserved. However, the harvested good will be scarce in the country with weak conservation policy because the stock is depleted. There are two important lessons here. The first is that there is not always a trade-off between environmental regulation and export success. Countries such as Iceland and New Zealand are successful exporters of fish because of (and not in spite of) their effective conservation policy. The second is that trade can create gains by taking pressure off a depleted resource stock. In our example above, when trade opens up, the country with depleted resources will import harvested goods from the resource-abundant country that has a good conservation policy. This will reduce pressure on the local resource stock and create opportunities for the stock to recover.

Empirical studies on the effects of trade on the sustainability of renewable resources are relatively few. However, a few studies lend support to the hypothesis that in the absence of good conservation policy,
an export shock can lead to fairly rapid depletion of the resource stock.\textsuperscript{4} Rapid depletion of fish stock occurred after the opening of the Estonian coastal fishery to trade in the 1990s (Vetemaa et al. 2006). Taylor (2011) provides evidence that the rapid decline and near extinction of the bison population in the US in 1870–1880 can be attributed to an export boom in buffalo hides triggered by an innovation in tanning technology. Carlos and Lewis (1993; 2010) document how the export of beaver pelts resulted in the depletion of beaver populations in some parts of Canada. Allen and Keay (2001) show how international competition among whalers contributed to the collapse of bowhead whale populations near Greenland in the late 17th and early 19th centuries. Jones and Bixby (2003) attribute the collapse of the abalone fishery off the coast of British Columbia to an export shock induced by the increased availability of air freight services to Asia.

Very few studies have considered the alternative hypothesis that trade can take pressure off a depleted resource and help support its recovery. Some have argued that the implementation of an improved conservation policy for Chinese forests was facilitated partly by the ability to import lumber from neighboring countries. Kjaergaard (1994) argues that Denmark recovered from deforestation and encroachment by sand dunes in the 1700s by importing wood and alternative fuels (and by hiring German engineers to design methods of protecting beaches from erosion, which we can think of as trade in services).

\subsection*{7.2.4 Technique Effects and Trade}

The evidence reviewed above suggests that country-level composition effects of trade liberalization are small, at least for OECD countries. This raises questions about the importance of comparative advantage-induced trade in affecting environmental outcomes. As noted above, the evidence from the PRC suggests that there are important regional variations in the effects of trade on pollution. This implies that comparative advantage is important at the regional level. Autor, Dorn, and Hanson (2013) found significant regional variations within the US in the effects of trade with the PRC on manufacturing output. I am not aware of studies that take the next step and explore whether this also led to regional differences in the environmental impact of increased trade with the PRC in the US, but the variation in output effects suggests such an effect. Nevertheless, the question remains as to whether, on aggregate, the effects of trade induced by the composition effects of comparative advantage-induced trade are small, except via the effects on growth.

\textsuperscript{4} This discussion is influenced by Copeland and Taylor (2017).
Recent work on international trade suggests that there may be other important channels via which trade affects environmental outcomes. This work has recently been reviewed in Cherniwchan, Copeland, and Taylor (2017) and the discussion that follows is influenced by that work. A large portion of international trade is intra-industry trade—trade in similar products. Such trade is not driven by comparative advantage but by a combination of fixed costs, imperfect competition, a consumer taste for product variety, and producer demand for specialized firm-specific inputs. Within any industry, firms produce differentiated products: each firm carves out a market niche and, as a result, countries import and export differentiated products within the same industry. Moreover, the role of trade intermediate inputs has expanded in recent years. Firms increasingly rely on value chains, with different tasks being performed in different places.

A key question is whether firm-level adjustments to trade can play a significant role in affecting environmental outcomes. If firm-level adjustments do affect emissions, they would have shown up in earlier studies as changes in sectoral-level emission intensities—that is, they would be classified as technique effects. As we discuss below, digging deeper into these firm-level effects can reveal firm-level composition effects at work. Moreover, the fragmentation of production and use of value chains that yield firm-level emission intensity reductions may be driven by the forces of comparative advantage that would not have been detected in earlier work—again they would have been measured as technique effects.

**Intra-Industry Trade**

Standard models of intra-industry trade (Krugman 1979; Melitz 2003) are consistent with small industry-level composition effects. These models were developed originally to explain the high volume of trade between similar countries, a stylized fact that was challenging for traditional models of comparative advantage to explain. In the simplest version of an intra-industry trade model (such as Krugman 1979), two identical countries each produce a manufacturing good. However, consumers have a taste for variety, and there are fixed costs associated with producing different varieties. Consequently, each firm produces a different variety of the manufacturing good. Trade occurs because consumers in each country want to consume all varieties, and so trade in similar products (intra-industry trade) results. An important assumption in Krugman’s model is that all existing and potential firms have identical production technology, which means that all firms facing the same environmental regulations will have the same emissions intensity.
This type of model predicts that trade causes a reallocation of production across firms within a sector. In Krugman (1979), trade causes each country to specialize in a smaller number of varieties, and firms expand to serve the export market as well as the domestic market. Once pollution is introduced into such a model, there are no composition effects: the effects of trade on the environment depend on the interaction between scale and technique effects. The outcome depends on the exact specification of the model, but such models tend to predict that trade leads to lower pollution. This is because access to foreign varieties increases real income, which tends to increase the demand for environmental quality more than in proportion to the increase in the demand for the right to pollute arising from scale effects. Anouliès (2016), for example, finds this result in a very simple intra-industry trade model. Roy (2017) uses an instrumental variable approach to identify the effects of increased intra-industry trade on several environmental indicators in a sample of 200 countries, and finds evidence supporting this hypothesis: increased intra-industry trade tends mostly to lead to a fall in pollution and such trade tends to be more pro-environment than trade driven by comparative advantage.

McAusland and Millimet (2013) consider the effects of intra-industry trade within and between countries. They find that international trade allows countries to shift some of the costs of pollution abatement on to foreigners. This implies that international trade leads to a stronger tightening of environmental regulations than does increased trade within a country. They find support for this using data on trade between US states and Canadian provinces.

The importance of intra-industry trade provides one possible explanation for the findings of relatively small trade-induced composition effects discussed above. However, more recent work has focused on intra-industry trade models with heterogeneous firms, and this yields a richer set of predictions.

In the Melitz (2003) model of intra-industry trade, each sector has a continuum of potential firms that differ in productivity. There are fixed costs of producing and fixed costs of exporting. In autarky equilibrium, an endogenous productivity cut-off determines the marginal firm. Only

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5 In other versions of the model, trade causes little or no production changes—all of the effects of trade occur on the consumption side as consumers reduce their consumption of domestic varieties and start including foreign varieties in their consumption basket.

6 See also Bennaroch and Weder (2006) and Bennaroch and Gaisford (2014) for other specifications of the monopolistic competition model, some of which allow for composition effects.
firms above this critical level of productivity survive to produce. Trade has two effects. First is a selection effect: only the most productive firms export. Firms with lower levels of productivity serve only the domestic market; the least productive firms are squeezed out because of increased competition from trade. Then there is a reallocation effect. Surviving domestic firms shrink because of increased competition from trade, and exporting firms expand because they can take advantage of the new export market. Resources are, therefore, reallocated from the less productive to the most productive firms.

Several recent papers have applied this approach to study the effects of trade on the environment. This yields a number of implications for the effects of trade on the environment.

Emission intensity can be expected to vary across firms within the same industry. Most models predict that more productive firms are cleaner (although the reverse can also hold; see Cui 2017). This can come via a couple of channels. More productive firms produce a given level of output with fewer inputs than less productive firms. If pollution ultimately comes from the use of inputs (energy, raw materials, etc.), then emissions per unit input will be lower. Also, if abatement technology has fixed costs, then less productive firms that are operating on a smaller scale will not find it worthwhile to invest in abatement technology. As a result, more productive firms can have lower emission intensities because they are more likely to have made investments in cleaner technology.

If more productive firms are indeed cleaner than less productive firms, then the selection and reallocation effects of trade identified by Melitz (2003) will shift resources to cleaner firms—that is, trade will induce a composition effect across firms within a sector. Sectoral emission intensities will fall because trade has caused the composition of production within a sector to shift toward cleaner firms. Although this is a composition effect, it will show in industry-level decompositions as a technique effect. This channel was highlighted by Kreickemeier and

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8 This will be true if production technologies are homothetic. With non-homothetic technology, the more productive firms could be more capital or energy intensive, and it is possible for their emission intensity to be higher than less productive firms. This is ultimately an empirical question.

9 On this point, see Girma et al. (2008), Bustos (2011), Batrakova and Davies (2012), and Forslid et al. (2015). Cui (2017) considers the case where more productive firms could pay fixed costs to invest in a more productive technology that need not necessarily be cleaner.
Richter (2014). Cherniwchan, Copeland, and Taylor (2017) refer to this as the pollution reduction by rationalization (PRR) hypothesis.

Holladay (2016) finds some evidence consistent with the PRR hypothesis. Using US data, he finds that exporting firms have lower emission intensities than non-exporting firms (conditional on industry, state, and time fixed effects). Cui, Lapan, and Moschini (2016) also find that US exporters tend to be cleaner than non-exporters, and Forslid, Okubo, and Ulltveit-Moe (2015) find a similar result for Swedish manufacturing firms.

Martin (2012) finds support for the PRR hypothesis using data from India. She finds evidence that reallocation effects across firms within industries lead to a within-industry composition effect that tends to lower carbon emissions. Jinji and Sakamoto (2015) find that Japanese exporters have lower CO$_2$ emission and energy emission intensities in most, but not all, industries: the emission and energy intensities are higher for exporting firms in the iron and steel industry.

Trade will also induce technique effects that vary across firms even if there are no policy changes. Because trade increases the scale of more productive firms, those firms may find it cost-effective to spend the fixed costs to upgrade their abatement technology. This enhances the PRR effect discussed above. However, the other side of this is that trade also causes a contraction of output for domestic firms that are not productive enough to export. Those firms will find reduced incentives to pay the fixed costs for abatement technology. Cherniwchan, Copeland, and Taylor (2017) refer to this as the distressed and dirty industry (DDI) hypothesis.

Holladay (2016) finds evidence that could be interpreted as support for the DDI hypothesis: he finds that firms facing increased import competition in the US market tend to have higher emission intensities. Gutiérrez and Teshima (2016) also find evidence that supports some aspects of the DDI hypothesis alongside evidence that suggests more subtle effects. They find that trade agreements caused Mexican firms facing increased import competition to make fewer investments in environmental protection. This is consistent with the DDI hypothesis. However, they also find that local environmental quality around such plants improved—that is, something offset the potential adverse consequences of the DDI channel. They find that plants exposed to more import competition had reduced expenditures on energy and fuel. One hypothesis is that this, combined with improved air quality, was due to a reduction in output induced by import competition. That is to say, a negative scale effect may have offset the reduced investment in abatement. However, the authors also find evidence that energy intensity (energy per unit output) declined: they suggest that increased
competition created incentives to look for ways to reduce energy inputs. A technique effect may, therefore, also have been a contributing factor.

**Intermediate Inputs and Offshoring**

Another channel by which trade can affect firm-level emission intensities is through increased access to imported intermediate goods and via opportunities for outsourcing some production tasks to foreign markets (which we refer to here as offshoring).

The effects of increased access to imported intermediate goods can raise or lower pollution emissions, depending on whether the intermediates are relatively clean or dirty. Benarroch and Weder (2006) consider intra-industry trade in intermediate goods and show how trade affects firm-level emission intensities; depending on assumptions about technology, pollution could rise or fall. Martin (2012) finds that reduced tariffs on imported intermediates in India led to significant improvements in firm-level energy efficiency. Cherniwchan (2016) found that the North American Free Trade Agreement reduced emission intensities in US manufacturing firms; his evidence suggests that access to imported intermediate goods was partially responsible.

There is as yet relatively little work on the effects of offshoring on pollution. Cole, Elliott, and Okubo (2014) model offshoring by assuming that firms can either pay for abatement or pay a fixed cost to offshore all of their polluting activity. This yields the prediction that large firms will offshore polluting activity to avoid the costs of dealing with domestic environmental policy. Cherniwchan, Copeland, and Taylor (2017) sketch an offshoring model based on Feenstra and Hanson (1996). The production of final goods requires assembling a continuum of intermediate goods that can be produced in-house or offshored to foreign producers. Intermediates vary in pollution intensity. If the home country has a sufficiently more stringent environmental policy than foreign countries, then trade liberalization causes the most pollution-intensive intermediate stages of production to be offshored. This model is interesting in that it generates an outcome similar to that of the pollution haven hypothesis (trade causes pollution-intensive activity to concentrate in places with relatively weak environmental policy), and yet in the countries with a relatively stringent environmental policy, it would show up in the data as reduced firm-level emission intensities. The classic pollution haven hypothesis operates via an industry composition effect, as dirty industries migrate to places with weak environmental policy. In an offshoring model, industries (and firms) do not migrate and so industry-level composition effects are small; rather, firms fragment, with the dirtier stages of production moving to places with weak environmental standards. This yields a fall in emission intensities that
is driven by within-firm composition effects; these effects are driven by comparative advantage but embedded in a model of intra-industry trade. Cherniwchan, Copeland, and Taylor (2017) refer to this as the pollution offshoring hypothesis.

There is a small but growing literature aimed at evaluating the pollution offshoring hypothesis empirically. Michel (2013) uses a decomposition analysis and finds evidence that offshoring has contributed to a fall in various types of pollution emissions in Belgium. Cole et al. (2017a) use Japanese manufacturing data from 2009 to 2013 and find that firms have a significantly lower rate of CO₂ emission intensity growth after they offshore some of their production activity. This does not happen when they outsource production to other Japanese firms. This is consistent with the hypothesis that firms are offshoring relatively pollution-intensive stages of production. Li and Zhou (2016) find evidence that US environmental policy has caused US manufacturing firms to offshore some of their pollution-intensive activity. They find that when a firm increases its share of imports from low-wage countries, the emission intensity of its US plants falls. They also find evidence that this is accompanied by a fall in abatement spending by the US plants. Cherniwchan (2017) finds evidence that the fall in emission intensities in US manufacturing firms as a result of North American Free Trade Agreement was partly due to offshoring pollution-intensive stages of production.

### 7.2.5 Transport

The literature reviewed above studies the effects of trade on pollution generated during production. International transport is another important source of emissions. Cristea et al. (2013) report that international transport accounts for 3.5% of global carbon emissions. However, when they sum up carbon emissions generated during the production and transport of traded goods, they find that international transport accounts for about 33% of trade-related emissions. Emissions vary by mode of transport, with emissions per kilogram of traded goods being substantially higher for air than for sea transport. This also means that the contribution of transport to trade-related emissions varies a lot by country. For US exports, transport accounts for more than half of trade-related emissions because a relatively large value share goes by air transport. In contrast, in the PRC, production accounts for a large fraction of its trade-related emissions because much of its export trade goes by ship. Cristea et al. (2013) also do simulations that suggest that emissions from transport are likely to grow faster than the growth of trade, suggesting that trade leads to increased carbon emissions.
Shapiro (2016) also investigates the effects of trade on carbon emissions, paying careful attention to the carbon intensity of different modes of transport. He estimates a structural model of global trade that accounts for emissions generated during production and transport. He finds that trade increases global carbon emissions by about 5%, and that this is roughly evenly divided between increases in emissions from production and transport.

### 7.3 How Does Environmental Regulation Affect International Competitiveness?

#### 7.3.1 Domestic Pollution Policy

I now consider the effects of environmental policy on international competitiveness. Environmental policy aimed at polluting firms works through two major channels. It creates incentives to reduce emission intensities, that is, to reduce pollution generated per unit of output. This can be achieved by changes in inputs (such as switching from fossil fuels to renewable energy), changes by in production techniques, investment in abatement technology, and in the long run by creating incentives for innovation. Environmental policy also raises production costs or constrains the scale of output; that is to say, it can reduce output, employment, and productivity. If affected firms produce non-traded goods, some of the compliance costs can be passed on to consumers; but with higher costs and prices, output is nevertheless likely to fall (depending on demand elasticities) as consumers switch to alternative products and services. If affected firms produce tradable goods, then international competitiveness in the affected sectors is eroded. Environmental policy targeting producers of polluting goods affects the costs of domestic suppliers but not foreign suppliers. Hence, there is less scope to pass costs on to consumers, and so the negative output and employment effects in the affected industry will likely be higher than in the non-traded sector.

Both channels outlined above are part of the path toward a more efficient economy that internalizes pollution externalities. A past legacy of weak environmental policy means that polluting sectors have received an implicit subsidy for the use of environmental services. So, an efficient policy response is to shift the economy toward greener production activities. This will imply output reductions in polluting sectors and output expansion in cleaner sectors.

However, adverse effects on output, employment, and competitiveness create adjustment costs that result in income and
employment losses. The costs of more stringent environmental policy are often concentrated among a relatively small group of workers and firms, while the benefits of improved environmental quality are spread widely. This can create political pressure that makes it difficult to tighten up environmental policy. Moreover, in the tradable sector, opponents of more stringent policy can raise the issue of the non-level playing field. Domestic producers bear the cost of more stringent policy, while foreign firms exporting to the domestic market will not bear similar costs unless their own government has a similar policy agenda.10

These concerns have stimulated a large literature that assesses the effects of environmental policy on international competitiveness. The key questions are whether and to what extent a relatively stringent environmental policy leads to a reduction in net exports, net foreign direct investment (FDI) inflows, and increased offshoring of polluting activities. This issue applies to regulations affecting the full spectrum of pollutants but has loomed large in the climate change literature, where concerns about carbon leakage resulting from the loss of competitiveness have played a major role in policy debates. Several reviews of this literature exist.11 I will be brief and discuss mainly work focused on Asia.

The literature has faced two main challenges: the availability of data on the stringency of environmental regulations and issues affecting identification, such as the endogeneity of environmental policy.

In some cases, data on a pollution tax or charge are available (for example, Levinson 1999); however, this is rare. Proxies for the stringency of environmental regulations are often used, such as measures of abatement costs or qualitative measures of policy stringency. Abatement cost measures typically come from survey data and can be problematic because they are endogenous. The application of the US Clean Air Act has been used as an indicator of the stringency of environmental regulation in several studies. Compliance with national air quality standards was enforced at the county level. Counties not in compliance had to implement more stringent regulations than those in compliance, this leading to a simple measure of the variation in policy stringency across counties. Brunel and Levinson (2016) provide a good overview of issues related to the measurement of the stringency of environmental policy.

10 Policies aimed at controlling pollution-generated consumption will not be subject to this problem because governments can regulate the emission intensity of any goods consumed within their jurisdiction, regardless of where they were produced. See Copeland (2011) for more discussion and a review of issues surrounding consumption-generated pollution.

Endogeneity of environmental policy is also a key challenge. For example, political pressure may result in industries subject to strong import competition facing weaker environmental policy than other sectors. A researcher may then find that net exports are positively correlated with the stringency of environmental policy. However, in this case, it is net exports that drive environmental policy rather than the reverse. Similarly, countries that have a comparative advantage in polluting sectors (because of natural resource deposits, capital abundance, access to technology, etc.) will face significant pressures on their environment. An effective government will implement a stringent environmental policy in response. Again, a researcher will find that net exports are positively correlated with the stringency of environmental policy but, in this case, it is export success that drives environmental policy, not the policy that is causing export success. The availability of panel data that allow researchers to use fixed effects to deal with omitted variable bias and the use of strategies such as instrumental variables to deal with endogeneity have allowed researchers to address these challenges, with varying degrees of success.

Early work failed to find a link between environmental policy and various measures of sectoral-level competitiveness (Jaffe et al. 1995). More recent work that accounts for policy endogeneity or exploits panel data has tended to find support for the hypothesis that more stringent environmental policy reduces net exports (Levinson and Taylor 2008; Broner et al. 2016), reduces net inflows of FDI (Keller and Levinson 2002; Millimet and Roy 2016), induces multinationals to shift some of their production out of the US (Hanna 2010); and reduces new plant births in US counties with stringent regulation (Becker and Henderson 2000). The magnitude of the effect is, however, still a matter of controversy. It varies across industries (Becker and Henderson 2000) and across studies. Many studies find a relatively small effect; however, some, such as Levinson and Taylor (2008) and Broner, Bustos, and Carvalho (2016), find quite large effects.

Most of the work in this literature has used US or European data. However, a small but growing number of studies have used data from Asia. Most find some evidence in support of the hypothesis that more stringent environmental policy reduces net exports or incoming FDI in affected sectors.

Chung (2014) finds that outbound FDI from the Republic of Korea (henceforth Korea) is influenced by environmental regulations in the host country. He uses a qualitative measure of the stringency of environmental regulations in host countries and industry-level data disaggregated to the four-digit level. The policy measure is interacted with the pollution intensity of the industry. Controlling for other sources
of comparative advantage, he finds that weak environmental regulation is both statistically and economically significant as a factor that attracts Korean investment in polluting industries. Using more recent data for Korea’s outbound FDI, Yoon and Heshmati (2017) find similar results.

A series of papers have found evidence that more stringent environmental policy in the PRC reduces incoming FDI in pollution-intensive industries but that the effect varies with the source country.

Dean, Lovely, and Wang (2013) study the effects of environmental policy on foreign investment (via joint ventures) in the PRC for the period 1993–1996. They exploit variation across provinces in water pollution charges as their measure of environmental policy stringency. They find that the effects of environmental policy on inbound investment vary both by industry and by source economy. Significant effects are found only in heavily polluting industries and for investors from Hong Kong, China; Macau, China; and Taipei, China. Investment from non-ethnic Chinese source countries was not significantly affected by environmental policy. Di (2007) uses data on firms in four industries in the PRC (two highly polluting and two not) and finds that more stringent environmental policy deters FDI in polluting sectors.

Cai et al. (2016) use a much larger and more recent set of data on inbound FDI to the PRC. To help facilitate identification, they exploit the national two control zones (TCZ) policy introduced in 1998, in which TCZ cities are subject to more stringent environmental regulation than non-TCZ cities. They find that more stringent environmental regulation deters FDI in pollution-intensive industries; however, this effect only appears for countries with weaker environmental regulation than the PRC. For source countries with more stringent environmental regulation, Chinese environmental policy did not have a deterrent effect on FDI.

Hering and Poncent’s (2014) study is one of the few to examine the effects of environmental policy on trade flows using data from Asia. They use PRC’s TCZ policy to provide a measure of the stringency of environmental policy. They study the effects of PRC’s environmental policy on exports from Chinese cities, some of which have more stringent regulations than others because of the TCZ policy. They find that more stringent environmental policy reduces exports from polluting sectors, but that the effect varies according to whether firms are privately owned or state owned. While policy had significantly affected privately owned firms, a similar effect was not identifiable for state-owned firms.

7.3.2 Carbon Leakage

The potential effect of a carbon tax on the international competitiveness of affected industries has been a major issue in debates on climate
policy. The issue arises when there are sub-global agreements. If the shadow price of carbon emissions varies across countries (because some countries take much more aggressive action to reduce carbon emissions than others), then the concern is that carbon-intensive production will shift to countries with a lower effective carbon price, and that this will undermine efforts by other countries to reduce emissions.

Although the underlying mechanism is essentially the same as in the literature discussed above, it has loomed much larger in the climate change literature. A major reason for this is that carbon emissions are a global pollutant and are, thus, subject to global free rider issues. If a country tightens up regulations aimed at particulate matter or SO$_2$ emissions, there may be some leakage if their regulations induce some shifting of production to other countries. This will generate some local opposition to the policy but the benefits will be clear: local air quality will improve. Moreover, although pollution may increase elsewhere because of competitiveness effects, that pollution will typically either not affect the jurisdiction tightening up its policy or will have only a small effect. Any environmental policy is accompanied by costs. The competitiveness effect will just be one of the costs, and it may be deemed tolerable because of the clear environmental benefits.

Leakage induced by more stringent regulations aimed at reducing carbon emissions, on the other hand, will have much different effects. The increase in pollution that occurs elsewhere will affect the global environment and, hence, harm the country that tightens up its regulations. Moreover, if leakage is significant, then those in a country considering tightening up its policy will be concerned that leakage will undermine the effectiveness of the policy. They will incur compliance costs but see a relatively small reduction in overall global emissions. Finally, the environmental benefits will be long term and will spread among all countries. This free rider problem implies that the specter of carbon leakage is a powerful argument to undermine local support for emission reduction policies unless such policies are accompanied by a mechanism to counteract leakage.

To date, there is relatively little empirical evidence on the magnitude of carbon leakage mainly because policies targeting carbon emissions are relatively recent (for a recent review, see Carbone and Rivers 2017). Most of the estimates come from computable general equilibrium models and they tend to range from 5% to 25%,\textsuperscript{12} although some models have predicted leakage in excess of 100% (Babiker and Rutherford 2005).

There is a small econometric literature. Aichele and Felbermayr

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\textsuperscript{12} A 25% leakage estimate implies that one-quarter of the drop in emissions in the policy-active country will be offset by an increase in emissions elsewhere.
(2015) use a gravity model of trade to estimate the carbon leakage induced by countries that ratified the Kyoto Protocol. Their measure of the stringency of regulation is simply whether or not the country ratified the protocol. Their approach allows them to identify leakage as a result of trade between bilateral country pairs. They find that embodied carbon imports into ratifying countries from countries that did not ratify the Kyoto Protocol increased by about 8%, suggesting that leakage occurred.

Aldy and Pizer (2015) study the effects of fuel price changes on US manufacturing, including potential competitiveness effects, which are captured by increases in net imports in fuel-intensive industries induced by higher fuel prices. They then use their estimates to simulate the effect of a $15 per ton carbon tax imposed by the US. They find that a US carbon tax would reduce output in energy-intensive industries, with estimates of output reductions as high as about 5% in the most affected industries. However, only about one-sixth of this output reduction is due to displacement by imports—that is, they conclude that a carbon tax would induce some leakage but that it would be relatively small.

Another approach to quantifying potential leakage is to estimate a structural model of the affected industry. Fowlie, Reguant, and Ryan (2016) do this for the cement industry in the US. Cement production is a major source of carbon emissions and, although cement is heavy, ocean shipping is feasible. There is also trade along the borders with Canada and Mexico. Since 1980, imports into the US have grown and, by 2006, imports had about a 25% market share. Moreover, the cement industry is characterized by large fixed costs, which means market power is considerable. The authors estimate the effects of various carbon taxes and find that both leakage and the exacerbation of the market power distortion induced by the exit of the marginal producer would seriously undermine the effects of a unilateral carbon tax. In their simulations, they impose a carbon tax equal to the social cost of carbon and find welfare losses for the US when implementing carbon taxes for cases where the social cost of carbon is less than about $40 per ton. It is interesting that, in their model, leakage offsets the market power problem—import competition mitigates consumer losses as marginal US producers exit in response to a carbon tax—but undermines the benefits of lower carbon emissions.

### 7.4 Summary: What Have We Learned from the Recent Literature?

Before considering policy, it is useful to review a few key things we have learned from the literature reviewed above.
First, the effects of trade on aggregate environmental outcomes for a country are relatively small. Evidence suggests that industry composition effects are small (although we need more evidence for developing countries) so that trade-induced growth effects tend to dominate. But given that growth is an objective for most countries, it is not obvious that the environmental policy response to trade-induced growth should be any different than that to other sources of growth.

Second, behind small aggregate effects can lurk large effects on particular regions or sectors. Trade can concentrate global demand on a particular production activity in an economy. This can lead to rapid trade-induced collapses of renewable resource stocks and large increases in local air and water pollution in regions experiencing export booms (Bombardini and Li 2016).

Third, trade can have important positive or negative consequences via its influence on technique effects. Trade can facilitate access to clean inputs and green technology that contributes to lower emission intensities. Furthermore, trade-induced increases in income can lead to increased political pressure for more effective environmental policy.

Fourth, heterogeneous firm-level responses to trade can affect environmental outcomes. Offshoring parts of a production process can lead to lower domestic emission intensities while shifting polluting activities to other countries; firm-level selection and reallocation effects will also affect sectoral emission intensities. Evidence on the magnitude of these effects, however, remains the subject of ongoing research.

Fifth, international transport accounts for a significant fraction of trade-related pollution emissions—between 33% and 50% of such emissions (Cristea et al. 2013; Shapiro 2016). Emissions vary with the mode of transport (emissions per ton of traded goods are massively higher from air transport than from ocean shipping); and the contribution of transport to trade-related emissions varies by country, being more substantial for high-income than for low-income countries because higher-value goods are more likely to be shipped by air.

Sixth, there is evidence that more stringent environmental policy reduces international competitiveness in affected sectors, although the magnitude of the effect is the subject of ongoing research. Most evidence suggests that the effect is relatively small. Other factors are much more important than pollution policy in determining trade and investment flows.

Finally, while numerical simulations suggest that carbon leakage rates could range from 5% to 25%, still very few econometric studies identify actual carbon leakage. Nevertheless, the competitiveness effects of carbon policy will likely continue to attract much more attention from
policy makers than the competitiveness effects of policies targeting pollutants that affect local environmental quality and human health.

7.5 Policy Implications

7.5.1 Domestic Environmental Problems

A standard prescription is that trade policy should target trade issues and environmental policy should target environmental problems. This is the classic targeting principle. However, this only works when there are no other distortions. If, for example, polluters have market power or if multilateral collective action to deal with transboundary pollutants fails, then matters are more complex. Moreover, the infrastructure and administrative capability to develop, implement, and enforce environmental policy can be costly and is not always available, especially in developing countries.

To the extent that effective environmental policy is available, then in the case of domestic pollution problems, a key implication of the trade and environment literature is a need for policy coordination. If trade liberalization will stimulate polluting industries and will have large effects on local air and water quality in booming regions or put increased pressure on natural capital (soil, water, wildlife habitats, and renewable resources), then environmental policy needs to be monitored and adjusted to mitigate the potential for increased environmental degradation. This is especially critical in the case of local renewable resource stocks that can collapse quite quickly as a result of global demand pressure. The sequencing of policy reforms is an important issue in this context. In some cases, the view is that trade liberalization and pro-growth policies should come first to move people out of poverty and generate the income and wealth needed to support policies that will yield sustainable environmental outcomes. However, there are cases of effective irreversibilities (that is, where recovery from environmental degradation will take a great many years), especially when renewable resource stocks are vulnerable to collapse. Moreover, many sources of natural capital support income and employment, especially among the poor, so an export boom may not generate an increase in long-run income. In such cases, there is a strong case for ensuring that protection is adequate for natural capital before opening up an affected region to global demand pressure from trade and investment.

When effective environmental policy is not available, then we are much more firmly in a second-best world where the environmental implications of trade and investment liberalization should be considered. The piecemeal policy reform literature shows that a path of trade policy
reform that improves welfare always exists (see Copeland 1994) but such a package of reforms requires considering the environmental consequences of increased trade. Selective increases in market access, such as reduced tariffs on green intermediate inputs, are good candidates for policy reform.

In some cases, export restrictions can be a second-best conservation policy. External demand pressures on a resource need not cause environmental problems if there are effective restrictions on harvesting or resource extraction. However, if conservation policy is not effective, then an alternative is to use policies that reduce external demand pressure, such as export restrictions. Rules of the World Trade Organization (WTO) are more flexible in allowing export restrictions than import restrictions, although in some cases export restrictions are constrained by WTO accession agreements or regional trade agreements. The PRC has explicitly argued that some of its export restrictions and value-added tax rebate rules are designed to improve conservation and environmental quality. However, such restrictions can also be used to improve the PRC’s terms of trade or to protect downstream industries by redirecting raw material production to the domestic market. Eisenbarth (2017) explores these issues using PRC data. She finds some support for both channels; in particular, the structure of value-added tax tax rebates discourages the export of some emission-intensive activities and so does have aspects of a second-best environmental policy.

The evidence that environmental policy reduces international competitiveness in polluting sectors—but that most estimates of these effects are small—has implications for both developed and less-developed countries. The possibility that environmental policy may reduce the competitiveness of an affected sector is not in itself an argument against implementing the policy. Many changes that yield long-run benefits for an economy yield short-run adjustment costs as the structure of production adjusts to the new market reality: technological change favors some sectors and disadvantages others, and trade agreements can lead to output and employment reductions in import-competing sectors. However, policy makers need to design policies that are effective in achieving environmental targets but also minimize compliance costs. Where feasible and enforceable, policies that allow firms to choose the least costly way of achieving environmental targets are likely to minimize adverse competitiveness effects.

It is tempting to think that the competitiveness effects of environmental policy might provide an opportunity for countries to attract industry by keeping environmental standards weak. However, there is little evidence to support this as an effective strategy for growth. Although evidence exists that increasing the stringency
of environmental policy can lead to some relocation of production elsewhere, there is no strong evidence that weak environmental policy is a key factor in determining the location decisions of firms. That is to say, the evidence does not support the pollution haven hypothesis: polluting industry does not tend to concentrate in countries with weak environmental regulation. Other factors—agglomeration effects, quality of infrastructure, proximity to trading partners, government stability, characteristics of the labor force, taxes, etc.—play a significant role in determining firm location. Environmental policy is just one of a great many factors that matter.

7.5.2 Climate Change

The trade and environment literature also has several implications for climate policy. First, however, it is important to keep in mind that trade-related carbon emissions account for less than 10% of all carbon emissions. Shapiro (2016) calculates that trade-related emissions account for about 7.6% of carbon emissions, with these roughly evenly divided between emissions from transport and the production of traded goods. This is significant but it does imply that emissions generated by carbon leakage will be only a very small fraction of global emissions. Nevertheless, trade-related aspects of climate change policy loom large in policy matters because of the interaction between global free rider problems and domestic politics.

One important channel via which climate policies will impact countries is in terms of trade effects. This is especially important in considering the effects of policies aimed at international transport and for understanding the incidence of border carbon adjustments.

An efficient policy aimed at dealing with carbon emissions from global transport would require a uniform carbon tax on all forms of transport. This would require a global agreement and would ideally be accompanied by carbon taxes applied to domestic transport to avoid diversion effects. Shapiro (2016) considers both unilateral carbon taxes on international shipping applied by large trading blocs, such as the European Union (EU) and the US, and a global carbon tax on shipping. A carbon tax on shipping acts like a tax on global trade. As a result, unilateral taxes on global shipping tend to improve the terms of trade for countries implementing such a tax. For example, a unilateral carbon tax imposed by the EU would lower global carbon emissions but it would also transfer income from the rest of the world to the EU by means of trade improvements for the EU. Shapiro (2016) also finds that a global carbon tax on international transport would reduce welfare for the poorest two-thirds of countries. That is to say, while a global carbon tax
on shipping would raise global welfare (by 0.18%, according to Shapiro’s estimates), the incidence of the tax would be uneven across countries and it would generate implicit income transfers from poor countries to rich countries.

The welfare effects of border carbon adjustment (BCA) taxes are also influenced in terms of trade effects. In fact, such policies work by reducing the price of carbon-intensive goods received by exporters from countries that have relatively lax carbon emission policies.

The theoretical case for BCAs was developed by Markusen (1975), who showed that in the absence of a cooperative agreement to target global pollutants, a country that unilaterally reduces such pollutants can indirectly influence the level of emissions from its trading partners by targeting imports from those countries with a tariff. Copeland (1996) showed that a pollution content tariff (a tariff based on the pollution content of traded goods) would be a more finely tuned instrument to target foreign emissions. Keen and Kotsogiannis (2014) showed that a BCA policy could be part of a global cooperative climate agreement if some countries lack the resources or infrastructure to implement and enforce their own climate policy.

Nevertheless, BCA is a blunt instrument: it targets a very small fraction of foreign emissions (recall that trade-related carbon emissions are only about 7.5% of global emissions), and it does so indirectly. A BCA can induce trade diversion, as firms in an exporting country will shift some of their trade to other countries; this implies that, to be effective, a BCA should be implemented by a large coalition of countries. The effectiveness of BCAs has been explored mainly through the use of computable general equilibrium models. Most studies suggest that such policies would have some success in curtailing carbon leakage, although the effectiveness varies across studies (see, for example, McKibben and Wilcoxen 2009; Böhhringer et al. 2012a). One main benefit of a BCA system is that it would likely increase the political feasibility of the introduction of an effective carbon emission reduction policy by sub-global coalitions.

A result that is quite robust across studies is that the terms of trade effects induced by a BCA result in income transfers (Böhhringer et al. 2012a; 2012b). A BCA is a tax on imports, and it tends to improve the terms of trade of the country imposing the tax. This implies that it worsens the terms of trade for the countries it targets. These typically are lower-income countries. Mattoo et al. (2009) suggest that countries that would potentially be targeted by a BCA would be better off if they either imposed an export tax on carbon-intensive exports or negotiated voluntary export restraints. This would avoid some of the adverse redistributive effects of such policies, while still dampening leakage.
While the theme of most of the literature on carbon leakage is that trade creates challenges for dealing with climate change, trade can be part of the solution to climate change problems in many ways. Copeland and Taylor (2005) show that, in theory, international trade leads to lower pollution abatement costs because it eases substitution possibilities for goods that become scarcer via climate policy. This is more likely to be important for small countries than large countries because large countries can adjust via increased regional trade. Burgess and Donaldson (2010) provide evidence from India on how regional trade reduced the impact of weather shocks. Costinot, Donaldson, and Smith (2016) use a quantitative model to study the global effects of climate change on agriculture. Their simulations suggest that domestic reallocation of production and regional trade will play a bigger role in facilitating adjustment to climate change than would international trade.

Lovely and Popp (2011) highlight another channel through which trade can help support better policies to deal with climate change. They provide evidence that the availability of cleaner technologies (inputs) via trade makes it more likely that countries will adopt more stringent pollution regulations to lower compliance costs.

Trade can also provide export markets that facilitate the development of green technology. Acemoglu et al. (2012) study the role of directed technical change as a part of the transition to a low-carbon economy. They emphasize the role of knowledge spillovers and network effects in the innovation process. This means there are failures in the market for innovations, in addition to the market failures from pollution externalities. This suggests that carbon taxes (and equivalent policies) will not be enough to deal with climate change; they need to be supplemented by policies to promote innovations in cleaner technology. Export opportunities create larger markets that increase the incentives for investments in innovation.

Harrison, Martin, and Nataraj (2017) provided a recent review of green industrial policy. They highlight examples of cases where countries have been successful in developing renewable energy, such as the PRC’s support of the solar cell industry and the growth of the wind turbine industry in Germany. Harrison and her colleagues emphasize two aspects of a successful green industrial policy. First, there needs to be a demand for the products or processes that receive government support. Demand can be induced by domestic and foreign government policies that create incentives for substitution by cleaner technology, and this is where trade plays a role. Second, there needs to be a latent comparative advantage. Not every country will be successful in developing a solar cell industry, for example. There was strong demand for the technology in Europe and the US, but the PRC developed a large market share because of its latent
comparative advantage in the sector. Trade creates many opportunities for the development of new technologies. Governments have a role to play on both the demand side (by enacting policies that create a demand for clean technology) and the supply side (by providing support for research and development). However, policies aimed at the supply side are more challenging to implement because of the usual problem of identifying those sectors that will be successful in the long run.

7.6 Conclusion

The evidence suggests that the aggregate effects of trade on the environment are relatively small. However, aggregate measures mask regional and sectoral variations. Trade can concentrate global demand for a region's output or renewable resources, and this can lead to large adverse effects on environmental outcomes if appropriate policies are not in place. There is much scope for further research to identify the heterogeneous effects of trade on the environment within countries.

Recent work has focused on firm-level responses to trade liberalization and this is beginning to influence the trade and environmental literature. Evidence indicates that reductions in pollution in manufacturing in high-income countries have come from a fall in emission intensities rather than from a shift in the composition of production toward cleaner industries. However, much is still unknown about how trade affects emission intensities, especially via offshoring of polluting tasks within a firm. There is also scope for more work that assesses the role of trade-induced changes in the composition of production in affecting pollution outcomes in developing countries.

The targeting principle suggests that environmental policy should deal with the environment and that there is no need to adjust trade policy to deal with environmental problems. However, when there are fixed costs of implementing environmental policy and when there are structural distortions in economies, then inevitably trade and environmental policy will be linked. This is of particular concern in the case of climate change policy because the challenges of addressing global collective action problems mean that issues such as carbon leakage will loom large.

It is important, however, not to overemphasize the challenges created by linkages between trade and the environment. Trade can be critical in helping us deal with both local and global environmental problems. Trade facilitates the transfer of green technologies; it provides large markets to cover the fixed costs of innovation; and it allows consumers to more easily substitute away from dirty to clean goods and services. Policy should focus on helping facilitate these channels.
References


8

International Outsourcing, Environmental Costs, and Welfare

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8.1 Introduction

In recent decades, amid the increasing trend of globalization resulting from the rapid advancement in communication and transportation technology, the outsourcing of intermediate and/or finished goods or services to firms in foreign countries to lower production costs and increase production efficiency has become prevalent in world trade. For example, client firms in developed countries in the North (i.e., the United States [US] and the European Union [EU]), while maintaining management bases and conducting research and development at home, shift their manufacturing activities to developing countries in the South where labor costs are lower (e.g., the People’s Republic of China [PRC], India, Malaysia, the Philippines, Thailand, and Viet Nam), and/or buy a substantial amount of parts or services from local Southern firms.

It is noteworthy that, while the focus on international outsourcing (also known as “offshoring or fragmentation”) has been mainly placed on North–South outsourcing, a firm’s decision to outsource can actually be driven by various factors, including but not limited to the lowering of labor costs. Such factors can be related to capital, technology, and organizational competency. The paramount aim is to enhance the operational capability and profitability of the firm’s production. To illustrate, both the US and EU countries outsource products to each other. The PRC outsources various intermediate goods (such as crude petroleum, integrated circuits, and iron ore) from Australia; Germany; Hong Kong, China; Japan; and the Republic of Korea (ROK), among others, while itself shifting the outsourcing of goods (of garments,
apparel, toys, footwear, and tools) to other developing countries in Asia, Latin America, and Africa. The evolution of these outsourcing patterns over recent years reveals two important facts: (i) outsourcing can occur universally among countries and it can be of any direction (i.e., from North to South, from North to North, from South to North, and from South to South); and (ii) each time goods and services are imported, the importing country has possibly outsourced a portion of economic activity from abroad—that is, all international trade will likely involve some outsourcing of intermediate inputs.

Consequently, international outsourcing has become a major topic in the study of international trade and stimulated a spate of intellectual contributions to the literature from economists in the field (e.g., Chao and Yu 1993; Bhagwati et al. 2004; Kohler 2004; Long 2005; Görg and Hanley 2005; Jones 2005). Among the rich and still-growing body of literature studying various aspects of international outsourcing are two notable works, both of which use the neoclassical Heckscher-Ohlin general equilibrium framework: one by Bhagwati et al. (2004) and the other by Batra and Beladi (2010). In the former, Bhagwati et al. (2004) argue that trade opportunity increases as a result of advances in information technology, which converts the traditional non-tradable service into a so-called Mode 1 service. Analytically this is equivalent to a growth agent. Further, Bhagwati et al. (2004), without formally integrating outsourcing into the Neoclassical growth model, examined the terms of trade and the welfare effects of outsourcing.

Later, Batra and Beladi (2010), inspired by Mankiw’s argument for offshoring, explored several major properties of outsourcing, particularly the welfare effect of international trade, by incorporating a labor-augmenting feature of outsourcing into the neoclassical Heckscher-Ohlin model of general equilibrium. Here, it is worthwhile to note that (i) the argument of Bhagwati et al. (2004) for the growth effect of outsourcing is confined to the case of a Mode 1 service-type of outsourcing, which anticipates the study of Batra and Beladi (2010) on the labor-augmenting effect of outsourcing; and (ii) both Bhagwati (2004) and Batra and Beladi (2010) do not expound the equivalency of the effects of various types of outsourcing on economic growth. More recently, Choi and Beladi (2012) extended the work of Batra and

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2 See Mankiw et al. (2004) and Mankiw and Swagel (2006) for their extensive account of the debate around Mankiw’s 2004 statement about welfare gains from outsourcing.
Beladi (2010) by exploring the internal and external gains for a small outsourcing country in the presence of variable returns to scale.3

Over recent decades, the dramatic expansion of outsourcing by Northern firms to the South has inflicted massive damage on the once-fertile air and soil of the South. A marked example is the PRC, the so-called “world factory,” where the production of a myriad of outsourced goods and services (i.e., garments, apparel, toys, footwear, tools, light machinery, electronics, and information-technology products) has contaminated the air, water, and soil; depleted labor and material pools; triggered deforestation/desertification and global warming; and (more seriously) endangered public health. However, the vendor countries in the South are mostly developing countries whose primary goal is economic development. Hence, despite the environmental damage, they opt (in their early period of international offshoring) to provide a hospitable business environment to Northern outsourcing firms. However, as outsourcing activities expanded, the resulting environmental costs became too heavy for the vendor countries to bear. Therefore, since 2014, major vendor countries (notably the PRC) have begun to account for or internalize the environmental costs of outsourcing by enacting environmental regulations and taxes. This policy shift increases the prices of their outsourced goods and services to firms in the North. However, the extent of such a move is still considerably below the level necessary to bring the South back to an acceptable environmental quality, as specified by the standards of the World Health Organization. This argument is supported by numerous observations on the presently stagnating or regionally deteriorating environmental quality of the PRC and its neighboring countries, including the ROK, Cambodia, India, the Lao People’s Democratic Republic, Malaysia, and the Philippines. To quote from a few recent reports by mass media:4

“The cost of environmental degradation in [the People’s Republic of] China was about $230 billion in 2010, or 3.5% of the nation’s gross domestic product—three times that in 2004, in local currency terms.”

3 Batra and Beladi (2010) and Choi and Beladi (2012) deal with the factor-augmenting effects of outsourcing without environmental consideration.

4 See internet articles—Global Warning (2009) and Wong (2013)—in the References section for these reports. Other reports on outsourcing and environmental degradation in the South include Greenpeace International (2014), BBC News UK (2014), Kelly (2014), and Gracie (2015), as listed in the References.
“The acidic deposition damages buildings, degrades the environment and reduces crop yields. In India, wheat growing near a power plant suffered a 49% reduction in yield compared with that grown 22 kilometers away.”

Another development in international outsourcing is that many Northern firms have adopted new strategies to counter the rising costs of outsourcing, as instigated by Southern regulations. Firms have resorted to vendor-country diversification, partial outsourcing, insourcing, or resourcing. Since the mid-1990s, there have been numerous reports about the changing patterns of outsourcing by Northern firms.5

This chapter aims to investigate the welfare consequences via the environmental impact of international outsourcing in a three-stage model of North–South trade. In stage I, the outsourcing firms of the North inflict environmental damage on the vendor country in the South, but the South (pursuing economic development as its primary objective) is willing to bear the burden of environmental spillovers. In stage II, when the environmental damages become substantial, the vendor country begins to internalize the environmental costs by enacting regulations. However, the regulations in stage II and the resulting increase in the price (level) of outsourced goods and services are far below the level that can bring the Southern environment back to the internationally acceptable level. This chapter assesses the gains and losses of the South as caused by outsourcing from the North, explicitly expounding upon the environmental costs to the South. The policy implication of our finding is clear: further environmental regulations by the South and/or international collaboration are required to account fully for the environmental costs to the South, as in stage III.

8.2 Assumptions and the Model

We deploy a three-stage 2x2 model to depict an outsourcing country in the North, which has two sectors under perfect competition: sector 1 producing good 1 in the amount of \( X_1 \), and sector 2 producing good 2 in the amount of \( X_2 \). Each sector utilizes two factors, labor \( (L) \) and capital \( (K) \), that are fixed in supply and perfectly mobile between the two sectors. Flexible wages and rental ensure the full-employment of labor \( (L) \) and capital \( (K) \). Since nonconstant returns to scale can obscure the cost effect of outsourcing, the present analysis is confined to a constant

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5 For media reports on the recently changing patterns of outsourcing, see internet articles Frew (2011), Hohner et al. (2011), Heineman (2013), LeBeau (2013), and Northam (2014).
returns to scale case. Further, for mutual benefits, the outsourcing firms and the foreign vendor firms are assumed to share technologies in producing the outsourced goods and services. The factor-augmenting effects of outsourcing differ depending on the direction and type of the outsourcing. However, this chapter will focus on the labor-augmenting case because (i) it is the major reason behind the North–South trade causing the environmental problems in the South, and (ii) once its effects are identified, the effects of other types of factor-augmenting outsourcing can be similarly deduced. In the presence of the labor-augmenting effect of outsourcing, the production functions of the two sectors can be written as

\[ X_i = F_i(A_i, L_i, K_i) \quad i = 1, 2 \quad (1) \]

where \( L_i \) and \( K_i \) are the labor and capital employed by sector \( i \), \( F_i \) is linearly homogeneous in its inputs and subject to the law of diminishing returns, \( A_i \) is a labor-augmenting factor derived from the outsourced goods and services in sector \( i \), and \((A, -1)L_i\) is the effective quantity of labor services supplied by the outsourcing activity. Equation (1) shows that, in sector \( i \), \( A_i \) is initially equal to unity in the absence of outsourcing. As the firms in the sector engage in outsourcing, \( A_i \) rises above its value of unity and renders an automatic reduction in the use of domestic labor via a labor-augmenting effect on the sector’s productivity. That is, \( A_i \) is a choice variable to the \( X_i \) producers that increases with the volume of outsourcing. Further, it cannot be less than unity because that would mean that outsourcing raises firms’ unit costs. Further, \( dA_i = 0 \) in the absence of outsourcing, and \( dA_i = d(A_i - 1) > 0 \) in the presence of additional outsourcing in sector \( i \).

Let \( p_i \) denote the price of good \( i \) (\( i = 1, 2 \)), and \( p = p_1 / p_2 = p \), the relative price of good 1 in terms of good 2 setting \( p_2 = 1 \) initially. Then, the marginal revenue product (or the value of the marginal product) of the labor-augmenting factor \( A_i(A_x) \) can, respectively, be written as \( a_i = p_i(\partial F_i / \partial A_i) = p(\partial F_i / \partial A_i L_i)(\partial A_i L_i / \partial A_i) = p F_{A_i L_i} L_i > 0 \) and \( a_2 = (\partial F_2 / \partial A_2) = (\partial F_2 / \partial A_2 L_2)(\partial A_2 L_2 / \partial A_2) = F_{A_2 L_2} L_2 > 0 \) where \( \partial F_i / \partial A_i L_i = F_{A_i L_i} \), and \( \partial^2 F_i / \partial (A_i L_i)^2 < 0 \) due to diminishing returns (\( i = 1, 2 \)).

Assuming that the presence of many outsourcing firms and vendor firms renders the international outsourcing market as competitive as all other input markets, outsourcing firms are thus the price takers of the labor-augmenting factor \( (A) \) in each stage of outsourcing. In stage I, outsourcing by the Northern firms inflicts environmental damages.

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6 The assumption of competitive international outsourcing markets eliminates some trivial outcomes in our comparative static analyses and allows us to focus on the environmental costs of outsourcing.
on the South, but the vendor country (with its primary objective
being economic growth) is willing to bear the environmental costs of
outsourcing. Therefore, the actual price of the labor-augmenting factor
\((A_i)\) in this stage does not include the environmental cost. However,
moving into stage II, when the environmental damage become
significant, the vendor country begins to internalize the environmental
costs by enacting regulations. Consequently, vendor firms begin to
raise the prices of outsourced goods and services due to higher costs. Let
\(b_i\) and \(b_i^-\) respectively denote the actual price (reflecting the marginal
cost) level of the outsourced goods and services \((A_i)\) in terms of good
2 in stages I and II. Assuming a competitive international outsourcing
market in each stage, \(b_i(b_i^-)\) is perceived as a given parameter by
both the outsourcers and the vendors. Then \(b_i^- - b_i > 0\) can be shown
graphically as the upward shift in the perfectly elastic supply curve of \(A_i\)
to the outsourcing firms resulting from the regulatory measures of the
environmental costs in stage II. Since the firms engage in outsourcing
mainly to lower their costs of production, we postulate:

\[
A_i = 1, \text{ if } a_i \leq b_i \text{ in stage I (or if } a_i \leq b_i^- \text{ in stage II)}
\]

for all levels of \(A_i\), and \(A_i > 1\) if \(a_i > b_i\) in stage I

(or if \(a_i > b_i^-\) in stage II) for some levels of \(A_i\). \(2\)

In other words, if \(a_i \leq b_i\) in stage I (or if \(a_i \leq b_i^-\) in stage II) for all levels
of \(A_i\), there is no outsourcing in that stage and, hence, \(A_i = 1\). However,
if \(a_i > b_i\) in stage I (or if \(a_i > b_i^-\) in stage II) for some levels of \(A_i\), the
outsourcing firms in sector \(i\) engage in outsourcing and \(A_i\) rises above its
initial value of unity. Further, an increase in the volume of outsourcing
by the outsourcing firms elevates their production efficiency and raises the
value of the labor-augmenting factor \((A_i)\). Thus, the value of \(A_i\) implicitly
measures the volume of outsourcing.

Since outsourcing firms make profit-maximizing decisions based
on the marginal revenue product \((a_i = pF_{Ai, i}A_i)\) and the costs of the
outsourced goods and services \((b_i)\), we postulate \(A_i = A_{i,a} + A_{i}(a_i, b_i)\),
where \(A_{i,a}\) is an exogenous (or autonomous) component of outsourcing
and \(A_{i}(a_i, b_i)\) is an endogenous component responding to \(a_i\) and \(b_i\). This
is similar to the well-known macroeconomic procedure used to obtain
consumption or import multipliers. Note that additional outsourcing
lowers the firm’s unit cost only when \(a_i > b_i\) and, hence, \(A_i > 1\). For all
values of \(A_i\) for which \(a_i \leq b_i\), the firms would not undertake additional
outsourcing (\(A_i = 1\)).

In factor markets, perfect factor mobility ensures identical factor
prices equal to the value of the marginal product of each factor:
\[ w = p F_{A1L1} A_1 = F_{A2L2} A_2 \]
\[ r = p F_{K1} = F_{K2} \]  

(3)

where \( w \) and \( r \), respectively, denote the wage and rental rate in terms of the second good, \( F_{Li} \) and \( F_{Ki} \), respectively, stand for the partial derivatives of \( F \), with respect to labor and capital \((i = 1, 2)\).

The full employment condition in the factor markets implies

\[ L_1 + L_1 = L, \]
\[ K_1 + K_1 = K. \]  

(4)

The system of equations consisting of (1)–(4) represents the production side of the general equilibrium model of outsourcing. To facilitate the ensuing analysis, a comparative static analysis must be conducted to identify two basic properties of the present model, namely, sectoral output response to outsourcing and sectoral output response to price change.

Since the derivation of these effects is quite lengthy, we relegate it to the Appendix and state only the summary results in the following proposition:\(^7\)

**Proposition 1:** (i) For a given commodity’s prices and the environmental cost of outsourcing, outsourcing gives rise to an ultra-biased effect on sectoral outputs (i.e., it increases the output of the outsourcing sector at the expense of that of the other); and (ii) For a given level of outsourcing and the environmental cost of outsourcing, sectoral output responds positively to its relative price.

To be specific, the ultra-biased effects on sectoral outputs implies \( dX_i / dA_i > 0 \) and \( dX_j / dA_i > 0 \) \((i, j = 1, 2; i \neq j)\), and the positive price-output response in each sector means \( dX_1 / dp > 0 \) and \( dX_2 / dp > 0 \).

Meanwhile, the demand side of the outsourcing country is represented by the expenditures function

\[ E(p, U) = \min (pD_1 + D_2), \]

---

\(^7\) Choi and Beladi (2012) conducted a similar comparative static analysis under variable returns to scale. To focus on the environmental costs of outsourcing, our analysis is confined to a constant returns to scale case.
where \( D_i (i = 1, 2) \) is the consumption of the two goods, and \( U \) is a strictly quasi-concave utility function, \( U(D_1, D_2) \geq U \). Note that the expenditure function is derived by minimizing the expenditure subject to a utility constraint.

### 8.3 Outsourcing, Environmental Costs, Internalization, and Welfare

The economy’s budget constraint stipulates that the total value of expenditure is determined by the value of national income:

\[
E(p, U) = I, \tag{5}
\]

where \( I = pX_1 + X_2 - b_1(A_1 - 1) - b_2(A_2 - 1) \) denotes the national income expressed in terms of good 2. Free trade is assumed so that the domestic prices and the international prices of the traded goods are synchronized at the ratio of \( p \). Totally differentiating (5), we obtain the expression for the welfare effect of outsourcing occurring in the two sectors \( dW \).

\[
dW = E_v dU = pdX_1 + dX_2 - E_p dp - b_1 dA_1 - b_2 dA_2 - (A_1 - 1) db_1 - (A_2 - 1) db_2 \tag{6}
\]

where \( E_p = \partial E / \partial p = D_1 \), and \( E_i = D_i - X_i \) denotes the excess demand in sector 1.

Now suppose that an autonomous outsourcing takes place via the firms in sector 1, i.e., \( A_1 > 1 \), \( dA_1 > 0 \), \( A_2 = 1 \) and \( dA = 0 \).\(^8\) Note that the outputs of the two industries are directly affected by the outsourcing and indirectly affected by the adjustment in the terms of trade, as consequent to the outsourcing. We may postulate that

\[
X_i = X_i(A_i, p)
\]

so that

\[
\frac{dX_i}{dA_i} = \frac{\partial X_i}{\partial A_i} + \frac{\partial X_i}{\partial p} \frac{dp}{dA_i}, \quad i = 1, 2. \tag{7}
\]

\(^8\) This assumption is made solely for simplicity’s sake. The analysis of outsourcing in sector 2 simply involves the same procedure as that in the present case of outsourcing in sector 1. When outsourcing occurs in both sectors 1 and 2, the total effect of outsourcing is the summation of the relevant effects of the two sectors.
The first term (on the right-hand side) captures the direct output effect of outsourcing, and the second term indicates the indirect output effect caused by a change in the terms of trade.

Now, we are ready to analyze the welfare effect of outsourcing in sector 1. Differentiating (6) with respect to \( A_1 \) and using (7), we obtain the expression for the welfare effect of the outsourcing occurring in sector 1:

\[
\frac{dW}{dA_1} = \frac{pdX_1}{dA_1} + \frac{dX_2}{dA_1} - E_i \frac{dp}{dA_1} - b_1 - (A_1 - 1) \frac{db_1}{dA_1} \\
= [p \frac{\partial X_1}{\partial A_1} + \frac{\partial X_2}{\partial A_1} - b_1 - (A_1 - 1) \frac{db_1}{dA_1}] + (p \frac{\partial X_1}{\partial p} + \frac{\partial X_2}{\partial p} - E_i) \frac{dp}{dA_1}.
\] (8)

Since the terms of the trade effect could distract the present analysis from focusing on the environmental effects of outsourcing, we assume that the international commodity markets are competitive and, hence, the outsourcing country (North) is a price taker of the two final goods, i.e., \( dp / dA_1 = 0 \). Then, (8) reduces to

\[
\frac{dW}{dA_1} = \frac{pdX_1}{dA_1} + \frac{dX_2}{dA_1} - b_1 - (A_1 - 1) \frac{db_1}{dA_1}
\]

which can be rewritten as

\[
\frac{dW}{dA_1} = p \left[ (F_{A_1 L_1} A_1 dL_1 + F_{A_1 L_1} L_1 dA_1 + F_{K_1} dK_1) \right] \\
+ \left[ (F_{L_2} dL_2 + F_{K_2} dK_2) \right] \frac{dA_1}{dA_1} - b_1 - (A_1 - 1) \frac{db_1}{dA_1} \\
= \left( wdL_1 + a_1 dA_1 + rdK_1 \right) - (wdL_1 + rdK_1) - b_1 - (A_1 - 1) \frac{db_1}{dA_1} \\
= a_1 - b_1 - (A_1 - 1) \frac{db_1}{dA_1} = a_1 - b_1^*.
\] (9)

Equation (9) shows the various effects of outsourcing on the welfare of the country: the marginal revenue product of the outsourced goods

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9 This model can be extended to a large country case where the terms of trade are variable. See, for example, Choi and Yu (1985).
and services \((a_1)\) minus the price of the outsourced goods and services \((b_1)\), as coupled with the change in the price of the outsourced goods and services \((\Delta a_1)[db_1/da_1]\). Without fully internalizing environmental costs, outsourcing depletes environment quality during stage I. However, in stage II, the South enacts environmental regulations to account for the environmental costs, thereby increasing the price of outsourced goods and service \((\Delta a_1)[db_1/da_1] > 0\). Hence, in stage II, \(b_1' = b_1 + (\Delta a_1)(db_1/da_1)\) is regarded as the new exogenously given parameter by the Northern outsourcing firms.

Equation (9) shows that, in the absence of environmental costs, \((\Delta a_1)[db_1/da_1] = 0\), \(b_1 = b_1\). Therefore, outsourcing should occur until \(a_1 = b_1\) (or \(a_1 = b_1'\)), where \(dW/da_1 = 0\). This implies that outsourcing necessarily enhances the welfare of the outsourcing country and attains the optimum resource allocation for the world as well. Hence, the following proposition:

**Proposition 2:** In the absence of induced environmental cost, outsourcing enhances welfare up to the optimal level of both the country and the world.

However, when environmental costs exist, \((\Delta a_1)[db_1/da_1] > 0\), the actual cost of outsourcing paid by the Northern firms in stage II \(b_1'\) depends on the degree of internalization of the environmental costs, as subject to the regulations of the South. Specifically, based on the extent of internalization, the marginal cost of outsourcing to the outsourcing country \(b_1'\) in stage II can be positioned somewhere between the two extreme values, namely, \(b_1\) under no internalization and \(b_1'\) under full internalization, i.e., \(b_1 \leq b_1' \leq b_1\).

Considering stage III, there will be a full internalization of the environmental costs, \(b_1^* = b_1 + (\Delta a_1)(db_1/da_1) = b_1'\). Outsourcing occurs up to the extent where \(a_1 = b_1^*\) and \(dW/da_1 = 0\) in equation (9). In this case, the volume of outsourcing obviously shrinks compared with the case of stage I (where, due to no internalization of the environmental costs, outsourcing occurs up to where \(a_1 = b_1\)). However, outsourcing under the full internalization in stage III determines the optimal level of outsourcing for the world, ensuring efficient worldwide resource allocation (including attaining a quality environment). In contrast, when the environmental costs are not accounted for (as in stage I), \(b_1^* = b_1\) and, hence, outsourcing occurs up to the level of \(a_1 = b_1\), implying over-outsourcing (vis-à-vis the optimum level of outsourcing that occurs at \(a_1 = b_1^*\)). Here, it is our view that the over-outsourcing (under no internalization) in stage I is a major cause of massive environmental damage to the South (e.g., the PRC). Then, what is the current state of internalization in the North–South outsourcing? The answer can be
inferred from the presently stagnating or even regionally deteriorating environmental quality in the South, as frequently reported by the mass media (see Note 4); that is, North–South outsourcing is presently in a state of partial internalization of its environmental costs. Under such a partial internalization (in stage II), the actual price of the outsourced goods and services ($b_1^+$) is higher (lower) than those under no internalization (full internalization), i.e., $b_1^+ < b_1^c = b_1^f$.

In the present case, over-outsourcing still occurs and creates an environmental burden for the South, although the degree of over-outsourcing is less than that under no internalization. Then, another question may arise: why does North–South outsourcing remain at a level of partial, rather full internalization (under which the environmental problem would be resolved)? Several explanations are possible. First, the economies of the vendor countries in the South are still in the infant stages of development and, hence, need international liquidities and Northern technologies to pursue their primary goals of economic development. Therefore, they are reluctant to employ a potent policy that could totally discourage outsourcing by the North. Second, as the prices of outsourced goods and services go up due to the environmental regulations implemented by the South in stage II, the Northern firms resort to countermeasures to lower the costs of outsourcing, such as vendor-country diversification, partial outsourcing, insourcing, or resourcing. Thirdly, in stage II, the increasing price of outsourced goods and services induces the entry of new vendor firms into the outsourced industry; thus, the increased competition among vendors hinders prices from rising in the outsourcing market. Hence, we can state the following proposition:

---

10 When facing rising outsourcing costs due to the internalization of environmental costs, as in stage II, the outsourcing firms of the North opt to search for a new optimal strategy to lower outsourcing costs. Among a few alternative strategies are (i) shifting their outsourcing post(s) to other vendor countries in the South where the costs of outsourcing are lower than in the current vendor countries. This strategy, leading to vendor country diversification, as well as slowing the rate at which the price of outsourced goods and services is rising, can be observed in the migration of US and EU firms from the PRC to other vendor countries in the South—i.e., Cambodia, Dominican Republic, Ghana, India, Indonesia, Malaysia, the Philippines, Rwanda, and Viet Nam where the environmental problems are less severe and the costs of outsourcing are lower than in the PRC; (ii) bringing their outsourcing post(s) back to the home country in the North through insourcing or resourcing; and (iii) adopting a partial outsourcing strategy by mixing outsourcing and insourcing strategies. The choice of strategies should obviously determine the future pattern and geography of global outsourcing.
Proposition 3. In the presence of the environmental costs of outsourcing, optimal outsourcing, as ensuring an environment acceptable to the world community, can be obtained under the full internalization of the environmental costs. However, if the environmental costs are not fully internalized, over-outsourcing and environmental deterioration occur in the vendor country. In this case, the level of over-outsourcing and the degree of environmental deterioration vary inversely with the degree of the internalization, as bounded by no internalization versus full internalization.

Figure 8.1 shows the gains and losses from outsourcing for the North and the South in the presence of environmental costs. In the graph, \( A_1 - 1 \) (or \( A_1 \)) denotes the level of outsourcing, and \( a_1 = pFA1L1L1 \).
indicates the demand for (or the marginal revenue product of) $A_1$. Since the outsourcing firms are the price takers of $A_1$, they face a perfectly elastic supply curve of $A_1$ at $b_1 (b_1')$ when the environmental costs are not (are) internalized by the South in stage I (stage II). In stage I, when outsourcing between the South and the North takes place at the price of $b_1$, equilibrium outsourcing occurs at $g$ and the level of outsourcing is $A_1^\pi - 1$. The North clearly gains by the area of $b_1 b_1'^fdg$ because it pays no environmental cost, and the South loses by the area of $b_1 b_1'^fhg$ because it bears the whole burden of the environmental costs. Therefore, the welfare loss occurs to the world by the area of $dhg$, which represents the deadweight loss caused by the misallocation of resources due to over-outsourcing; that is, in stage I, the South subsidizes the North in terms of enduring the environmental costs of outsourcing.

In stage II, the actual price of the outsourced goods and services ($b_1'$) depends on the degree of the internalization of the environmental costs. Under the full-internalization scheme in stage III, the price is $b_1'^c$. Thus, the transition from stage I through stage II to stage III entails movement of the equilibrium point from $g$ to $d$ and, hence, decreases the level of outsourcing from $A_1^\pi - 1$ to $A_1'^c - 1$. As a result, the North loses by $b_1 b_1'^fdg$, the South gains $b_1 b_1'^fhg$, and the world gains by $dhg$ owing to the recovery of the deadweight loss of stage I. This confirms that the optimum level of outsourcing to the world community occurs when the environmental costs of outsourcing are fully internalized.

Suppose, as of now, the North–South outsourcing industry is in a state of partial internalization (as in stage II), such that, in Figure 8.1, the price of the outsourced goods and services ($b_1'^c$) is situated between $b_1$ and $b_1'^f$ (i.e., $b_1 < b_1'^c < b_1'^f$), and the equilibrium outsourcing point is $e$. Then, at the outsourcing level $A_1'^c - 1$ (as compared with the no-internalization case in stage I), the South (North) clearly gains (loses) by the area $b_1 b_1'^cig (b_1 b_1'^ceg)$, and the world gains $eig$; whereas, compared with the case of full internalization, the South (North) loses (gains) by the area $b_1'^c b_1'^fje (b_1'^c b_1'^fde)$. Here, our finding is that, as the level of internalization increases from stage I toward the full internalization of stage III, the level of outsourcing decreases. The South (North) gains (loses) more, and the welfare gain for the world increases. Therefore, to resolve the environmental problem, strong Southern environmental regulations and/or international collaboration are highly desirable. The summary results are presented in Tables 8.1 and 8.2.
Table 8.1: Gains and Losses from No or Full Internalization

<table>
<thead>
<tr>
<th>Policy</th>
<th>Equilibrium</th>
<th>South</th>
<th>North</th>
<th>World</th>
</tr>
</thead>
<tbody>
<tr>
<td>(a) Outsourcing costs with full environmental costs → No internalization (stage I)</td>
<td>d→g</td>
<td>-b₁b₂h₁</td>
<td>+b₁b₂d₂</td>
<td>-dh₂</td>
</tr>
<tr>
<td>(b) No internalization → Full internalization (stage II)</td>
<td>g→d</td>
<td>+b₁b₂h₁</td>
<td>-b₁b₂d₂</td>
<td>+dh₂</td>
</tr>
<tr>
<td>(a)+(b) Net welfare change</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

Source: Authors.

Table 8.2: Gains and Losses from Partial Internalization

<table>
<thead>
<tr>
<th>Policy</th>
<th>Equilibrium</th>
<th>South</th>
<th>North</th>
<th>World</th>
</tr>
</thead>
<tbody>
<tr>
<td>(c) No internalization → Partial internalization at the current price of b₁</td>
<td>g→e</td>
<td>+b₁b₂e₁</td>
<td>-b₁b₂e₂</td>
<td>+e₂</td>
</tr>
<tr>
<td>(d) Partial internalization at the current price of b₁ → Full internalization</td>
<td>e→d</td>
<td>+b₁b₂e₁j₁</td>
<td>-b₁b₂e₂d₂</td>
<td>+d₂j₁</td>
</tr>
<tr>
<td>(c)+(d) Total welfare change</td>
<td>g→e→d</td>
<td>+b₁b₂e₁je₂</td>
<td>-b₁b₂d₂</td>
<td>e₂ + d₂j₁</td>
</tr>
</tbody>
</table>

Source: Author.

8.4 Conclusions

In recent decades, it has become prevalent in world trade that firms in one country outsource intermediate and/or finished goods or services from vendor firms in other countries to increase production efficiency and lower production costs. While public focus on international outsourcing is mainly directed on North–South outsourcing, a firm’s decision to undertake outsourcing can be driven by a host of factors (besides reducing labor costs). The ultimate aim thereof is to enhance the profitability and capability of the firm’s operation. To illustrate, both the US and EU countries have outsourced materials and goods from each other. The PRC outsources various intermediate goods (such as crude petroleum, integrated circuits, iron ore, gold, and cars) from Australia; Germany; Hong Kong, China; Japan; and the ROK, among others, while shifting its outsourcing of garments, apparel, toys, footwear, and tools to other developing countries in Asia, Latin America, and Africa.

The evolution of these outsourcing patterns reveals that (i) outsourcing can occur universally among trading countries whereby
its direction can be of any type (North to South, North to North, South to North, and South to South); and (ii) a generalized theory of international outsourcing covering all possible patterns will be the topic of future research.

This chapter has investigated the environmental effects of international outsourcing for both the North and the South using a three-stage general equilibrium model of international trade. We argue that the degree of internalization remains substantially below the level that can bring the South back to an internationally acceptable level of environmental quality. Thus, the tightening of environmental regulations and the fostering of international cooperation are warranted to promote better welfare for all.
References


Appendix

This appendix derives the effects of labor-augmenting outsourcing on sectoral outputs and the price-output response, which are essential to the analyses in the text. Suppose that autonomous outsourcing takes place by the firms in sector 1 only, i.e., $A_1 > 1$, $dA_1 > 0$, $A_1 = 1$, and $dA_1 = 0$. Differentiating equations (1) yields

$$dX_1 = F_{A1L1}A_1dL_1 + F_{A1L1}L_1dA_1 + F_{K1}dK_1$$
$$dX_2 = F_{L2}dL_2 + F_{K2}dK_2$$  \hfill (A-1)

Using (A-1), (2) and (3), we obtain

$$\hat{X}_1 = \theta_{L1}\hat{L}_1 + \theta_{K1}\hat{K}_1 + \theta_{A1}\hat{A}_1,$$
and $$\hat{X}_2 = \theta_{L2}\hat{L}_2 + \theta_{K2}\hat{K}_2,$$  \hfill (A-2)

where the circumflex, “$\hat{}$,” indicates the relative change of the variable (e.g., $\hat{X}_i = dX_i / X_i$) except $\hat{A}_j = dA_j / (A_i - 1)$, and $\theta_{ij}$ ($j = L, K; i = 1, 2$) represents the share of the $j$th factor in the total value of $i$th good, and $0 \leq \theta_{ai} = a_i(A_i - 1) / p_iX_i < 1$ denotes the share of the effective labor-augmenting factor ($A_i - 1$) in the total value of good 1. The determinant of $\theta_{ij}$ matrix is positive (negative) according to whether good 1 is labor-intensive (capital-intensive) relative to good 2 [i.e., $k_i < (>)k_j$], where $k_i = K_i / L_i$ (j = $L, K; i = 1, 2$). Perfect competition in the two traded goods markets ensures average-cost-pricing in each of the final goods industries, i.e., $wL_1 + rK_1 + b_1(A_1 - 1) = pX_1$ and $wL_2 + rK_2 = X_2$.

Defining the elasticity of factor substitution of the $j$th sector as

$$\sigma_j = \frac{\hat{K}_j - \hat{L}_j}{\hat{w} - \hat{r}} > 0_i \hfill (A-3)$$

where each $\sigma_j$ is positive. Substituting (A-3) in (A-2) and using the average-cost-pricing condition, we obtain

---

11 Since the outsourcing firms in sector 1 pay $b_1$ for the price of the labor-augmenting factor in stage 1, average-cost-pricing implies $wL_1 + rK_1 + b_1(A_1 - 1) = pX_1$ and $\theta_{L1} + \theta_{K1} + \theta_{A1} = 1$ where $\theta_{A1} = b_1(A_1 - 1) / pX_1$. Then, $\sigma_i = \theta_{L1} - \theta_{L2}(1 - \theta_{L1}) = \theta_{K2}(1 - \theta_{L1}) - \theta_{K1}$. 
\( \hat{L}_1 = [\hat{X}_1 - \theta_{K1}\sigma_1(\hat{w} - \hat{r}) - \theta_{A1}\hat{A}_1] / (1 - \theta_{b1}), \hat{L}_2 = \hat{X}_2 - \theta_{K2}\sigma_2(\hat{w} - \hat{r}), \)

\( \hat{K}_1 = [\hat{X}_1 - \theta_{L1}\sigma_1(\hat{w} - \hat{r}) - \theta_{A1}\hat{A}_1] / (1 - \theta_{b1}), \hat{K}_2 = \hat{X}_2 + \theta_{L2}\sigma_2(\hat{w} - \hat{r}). \) (A-4)

where \( 0 \leq \theta_{b1} = b(A_1 - 1) / pX_1 < 1 \) is the share of the actual payment for the effective labor-augmenting factor \((A_1 - 1)\) in the total value of good 1.

Since \( p_2 \) is set to unity initially, using (A-3), (A-4), and the average-cost-pricing condition, we obtain

\[
\theta_{A1}\hat{w} + \theta_{K1}\hat{r} = (\theta_{A1} - \theta_{b1})\hat{A}_1 - \theta_{b1}\hat{p} + \hat{p}
\]

and \( \theta_{L2}\hat{w} + \theta_{K2}\hat{r} = 0. \) (A-5)

Note that \( \theta_{A1} > \theta_{b1} \) in the presence of outsourcing because outsourcing occurs only if \( a_i > b_i \).

Since the international outsourcing market is competitive, the outsourcing firms are price takers of the outsourced goods and services, i.e., \( b_i = 0(b^*_i) \) in stage I (stage II).

Solving (A-5) for \( \hat{w} \) and \( \hat{r} \), we derive

\[
\hat{w} = (1 / |\theta|)[\theta_{K2}(\theta_{A1} - \theta_{b1})\hat{A}_1 + \theta_{K2}\hat{p}]
\]

\[
\hat{r} = (1 / |\theta|)[\theta_{L2}(\theta_{A1} - \theta_{b1})\hat{A}_1 - \theta_{L2}\hat{p}]
\]

\[
\hat{w} - \hat{r} = (1 / |\theta|)[(\theta_{A1} - \theta_{b1})\hat{A}_1 + \hat{p}]. \) (A-6)

Differentiation of the full-employment conditions (4) yields

\[
\lambda_{L1}\hat{L}_1 + \lambda_{L2}\hat{L}_2 = \hat{L},
\]

\[
\lambda_{K1}\hat{K}_1 + \lambda_{K2}\hat{K}_2 = \hat{K}. \) (A-7)

where \( \lambda_{ji} \ (j = L, K, i = 1, 2) \) is the proportion of the \( j \)th factor in the \( i \)th industry.

Setting \( \hat{L} = \hat{K} = 0, \) and substituting (A-3), (A-4), and (A-5) into (A-7), we get a system of equations in matrix form:

\[
\begin{bmatrix}
\lambda_{L1} & \lambda_{L2} \\
\lambda_{K1} & \lambda_{K2}
\end{bmatrix}
\begin{bmatrix}
\hat{X}_1 \\
\hat{X}_2
\end{bmatrix}
= \begin{bmatrix}
\delta_L(\theta_{A1} - \theta_{b1})\hat{A}_1 / |\theta| + [\lambda_{ji}\theta_{A1} / (1 - \theta_{b1})\hat{A}_1 + (\delta_L \hat{p} / |\theta|)] \\
-\delta_K(\theta_{A1} - \theta_{b1})\hat{A}_1 / |\theta| + [\lambda_{ji}\theta_{A1} / (1 - \theta_{b1})\hat{A}_1 - (\delta_K \hat{p} / |\theta|)]
\end{bmatrix} \) (A-8)
where,
\[ \delta_L = \lambda_{L1} \theta_{K1} \sigma_1 (1 - \theta_{b1}) + \lambda_{L2} \theta_{K2} \sigma_2, \]
\[ \delta_K = \lambda_{K1} \theta_{L1} \sigma_1 (1 - \theta_{b1}) + \lambda_{K2} \theta_{L2} \sigma_2. \]

Here, \( \delta_j > 0 \) \( (j = L, K) \) denotes the change in the use of the \( j \)th factor per unit of output that occurs in both industries due to the change in wage-rental ratio.

### A.1 Outsourcing and Sectoral Outputs

#### Effect of Outsourcing on Sectoral Outputs

To identify the effects of outsourcing on sectoral outputs at constant commodity prices (i.e., \( \hat{p} = 0 \)), we solve the system of equations in (A-8) using the Cramer rule; thus, we obtain

\[
\frac{\dot{X}_1}{\dot{A}_1} = \frac{1}{|\lambda|} \left\{ \lambda_{K2} \left[ \frac{\delta_L (\theta_{d1} - \theta_{b1})}{|\theta|} + \frac{\lambda_{L2} \theta_{K2}}{(1 - \theta_{b1})} \right] - \lambda_{L2} \left[ \frac{-\delta_K (\theta_{d1} - \theta_{b1})}{|\theta|} + \frac{\lambda_{K1} \theta_{d1}}{(1 - \theta_{b1})} \right] \right\}
\]

\[
\frac{\dot{X}_2}{\dot{A}_1} = \frac{1}{|\lambda|} \left\{ \lambda_{L1} \left[ \frac{-\delta_L (\theta_{d1} - \theta_{b1})}{|\theta|} + \frac{\lambda_{K1} \theta_{d1}}{(1 - \theta_{b1})} \right] - \lambda_{K1} \left[ \frac{\delta_K (\theta_{d1} - \theta_{b1})}{|\theta|} + \frac{\lambda_{L2} \theta_{K2}}{(1 - \theta_{b1})} \right] \right\}
\]

\[
\frac{\dot{X}_2}{\dot{A}_1} = \frac{-1}{|\lambda|} \left( \lambda_{L1} \delta_L + \lambda_{K1} \delta_K \right) (\theta_{d1} - \theta_{b1})
\]

where \( |\lambda| = \lambda_{L1} \lambda_{K2} - \lambda_{K1} \lambda_{L2} \) is the determinant of the \( \lambda_{ji} \) matrix. Note that \( |\lambda| = \lambda_{L1} \lambda_{K2} - \lambda_{K1} \lambda_{L2} = (L_1 L_2 / LK)(k_2 - k_1) \), which is positive (negative) according to whether good 1 is labor-intensive (capital-intensive) relative to good 2 (i.e., \( k_1 < (> k_2 \)). Therefore, \( |\lambda| \) has the same sign as \( |\theta| \). Furthermore, \( \theta_{d1} - \theta_{b1} > 0 \) for outsourcing to occur. Therefore, \( \dot{X}_1 / \dot{A}_1 > 0 \) and \( \dot{X}_2 / \dot{A}_1 > 0 \). That is, outsourcing creates “ultra-biased” effects on sectoral outputs, i.e., the output of the outsourcing sector increases at the expense of the other sector.
A.2 Price–Sectoral Outputs Response

We obtain the expression for commodity price–output response at a given level of outsourced goods and services \( \hat{A}_i = 0 \) by solving the system of equations in (A-8) using the Cramer rule:

\[
\frac{\hat{X}_1}{\hat{p}} = \frac{1}{|\lambda|} \left[ \lambda_{x2} \delta_L + \lambda_{x3} \delta_K \right] > 0
\]

\[
\frac{\hat{X}_2}{\hat{p}} = \frac{1}{|\lambda|} \left[ \lambda_{x1} \delta_K + \lambda_{x2} \delta_L \right] > 0.
\]

That is, in a stable system, a rise in the relative price of a good always increases the output of the good and decreases that of the other good. Note that this result is consistent with the proposition of Neary (1978) that the price–output response is always normal (i.e., positive) under a stable system. Therefore, proposition 1 in the text follows from the results in A.1 and A.2.
9

Does Trade Openness Change the Environmental Kuznets Curve? Evidence from the People’s Republic of China

Zheng Fang, Bihong Huang, and Zhuoxiang Yang

9.1 Introduction

The relationship between economic growth and the environment has remained a debated issue over the last 2 decades. Grossman and Krueger (1995) brought forward the environmental Kuznets curve (EKC) hypothesis, which refers to an inverted U-shaped relationship between environmental degradation and per capita income, i.e., the environmental quality initially deteriorates but then improves as income rises. However, most empirical studies on the EKC hypothesis have presented mixed results (Shafik and Bandyopadhyay 1992; Antweiler et al. 2001; Cole 2004; Stern 2004; Jalil and Feridun 2011).

Some studies have tried to incorporate openness into the EKC analysis. In particular, the impact of trade openness/liberalization on the environment can be decomposed into scale, technology, and composition effects (Grossman and Krueger 1991; Copeland and Taylor 1994; Cole and Elliott 2003). The scale effect refers to the likely increase in emissions resulting from the overall economic growth generated by trade openness. The technology effect accompanying trade liberalization is expected to decrease the emission intensity through the usage of clean production technology. The composition effect refers to the change of economic structure that may occur when countries specialize in the production in which they have a comparative advantage. Therefore, the aggregate effect of trade on the environment is uncertain.

Cities in the People’s Republic of China (PRC) provide a unique setting for studying this issue because both trade and pollution have
Does Trade Openness Change the Environmental Kuznets Curve?  
Evidence from the People’s Republic of China

The income gap between coastal and inland PRC cities is remarkable. For instance, GDP per capita in Shenzhen amounted to $25,135 in 2015, while Dingxi in Gansu Province only had a per capita GDP of $1,748 in the same year. The huge regional gap that the shape of the EKC will vary dramatically across different cities. Surprisingly, few researchers have investigated the EKC at the city level in the PRC. Among them, Zheng et al. (2013) are the first to assess how public concern and the local leadership’s characteristics influence the EKC across PRC cities.

Using panel data covering 261 PRC cities for 2004–2013, this chapter complements the existing literature by examining how trade openness changes the relationship between income and the environment in the PRC. We attempt to answer the question: what does trade liberalization/openness bring to PRC cities and how does it facilitate the movement of cities on the EKC curve? Specifically, we analyze the impact of trade openness on the EKC using the emission of industrial pollutants, industrial wastewater, and sulfur dioxide (SO2) in cities. The empirical evidence supports the EKC relationship. The turning points of the inverted U-shaped curve are found to be around CNY31,849–CNY49,446 per capita GDP for wastewater and CNY9,274–CNY10,103 for SO2. In addition, trade openness is found to have a negative impact

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1 GDP data are taken from the CEIC China Premium Database and the exchange rate is obtained from the Bank for International Settlements.
on wastewater but a positive effect on SO$_2$. Furthermore, results from the regional analysis show that the EKC relationship also holds in different regions. However, if the cities are grouped by the level of openness, we find that the EKC hypothesis is only true in cities with a high level of trade openness, suggesting that trade openness is a key determinant of the relationship between the environment and income.

We contribute to the ongoing debate in several respects. First, we infer the effect of income on pollutant emissions with the state-of-the-art empirical method. The reduced-form models applied to the empirical study of the EKC reflect correlation rather than the causal mechanism (Cole et al. 2004; Kijima et al. 2010) due to the potential feedback effect from environmental quality to income growth. Different econometric methods have been adopted in the recent EKC literature to solve this concern about reverse causality, including the system generalized method of moments (GMM) (Li et al. 2016), the spatial panel model (Kang et al. 2016), and fully modified ordinary least squares (Kasman and Duman 2015), among others. In this chapter, we employ the recently developed continuously updated fully modified (Cup-FM) estimates (Bai et al. 2009) to address the endogeneity concern and cross-sectional dependence. We first assess the cross-sectional dependence of the panel and employ the cross-sectionally augmented IPS (CIPS) test (Pesaran 2007) to examine the presence of unit roots in the panel. For comparison, we also conduct fixed-effects regressions, and the difference and system GMM estimation. The empirical results indicate that Cup-FM estimates generate more robust results than other estimations.

Second, our empirical analysis on the growth–trade–environment nexus in the PRC is performed on a large scale of prefectural-level data over 10 years. Most studies on the EKC in the PRC focus on the provincial level. For example, Kang, Zhao, and Yang (2016) adopt a spatial panel model to examine the EKC hypothesis on carbon dioxide (CO$_2$) in 30 PRC provinces. Li, Wang, and Zhao (2016) use the system GMM in a dynamic panel model and an autoregressive distributed lag (ARDL) model to investigate the EKC hypothesis in 28 PRC provinces. Bearing in mind the huge income gap across cities, it is important to investigate the EKC hypothesis using city-level data.

Finally, this chapter divides the city sample into three regions—eastern PRC, central PRC, and western PRC—to examine the robustness of the EKC hypothesis. In addition, we classify all the cities into high openness and low openness by comparing their openness index to the median value. We find that the EKC hypothesis only holds for the high-openness cities and is invalid for the low-openness cities. The in-depth
analysis of the EKC relationship in various dimensions enhances our understanding of the potential drivers of the curve.

The remainder of this chapter is organized as follows. Section 2 connects this study with previous literature. Section 3 discusses the data and the econometric methodology. Section 4 presents the empirical results and Section 5 concludes the chapter with policy implications.

9.2 Literature Review

The existing literature has extensively studied the link between growth, trade, and the environment and indicated that the EKC hypothesis is more likely to hold for pollutants like \(\text{SO}_2\) (Grossman and Krueger 1995; Copeland and Taylor 1994; Suri and Chapman 1998; Cole 2004; Jalil and Mahmud 2009). However, due to the adoption of different samples, pollutants, and methodologies, conclusions vary and the causal linkage remains unclear.

Shafik and Bandyopadhyay (1992) found that all indicators of trade policy have insignificant impacts on water shortage and sanitation and municipal waste, but more open countries put less pressure on forest resources and tend to have lower levels of \(\text{SO}_2\) and carbon emissions. Using pooled cross-country and time series data, Suri and Chapman (1998) show that exporting manufactured goods by industrialized countries is an important factor in generating the upward-sloping portion of the EKC while importing by industrialized countries contribute to the downward slope.

Antweiler, Copeland, and Taylor (2001) investigated the effects of openness to trade on \(\text{SO}_2\) concentrations and estimated three trade-induced effects. They found that fre trade appears to be good for the environment. Dasgupta et al. (2002) cast doubt on the “race to the bottom” scenario, which indicates that outsourcing dirty production to developing countries where the environmental regulations are less strict lowers the pollutant emissions in developed countries. They argued that liberalization is good for the environment. Using the ARDL methodology, Jalil and Feridun (2011) and Jalil and Mahmud (2009) conclude that a quadratic relationship exists between income and \(\text{CO}_2\) emissions, and trade openness has a positively significant impact on carbon emissions in the PRC.

However, there are some criticisms of the EKC. For example, Harbaugh, Levinson, and Wilson (2002) reexamined the EKCHypothesis on a panel of worldwide cities and found that the results are sensitive to changes in model specifications. Stern (2004) summarized four
main econometric criticisms of the EKC hypothesis: heteroskedasticity, simultaneity, omitted variables bias, and cointegration issues. Perman and Stern (2003) argue that cointegration analysis is imperative for testing the validity of the EKC given that the data usually contain stochastic trends. Wagner (2008; 2015) also pointed out the inadequate application of unit root and cointegration techniques in several empirical studies. He noted that standard panel cointegration tests are not appropriate when there are short time dimensions, nonlinear transformations of integrated variables, or cross-sectional dependence in the data. In this chapter, we employ the Cup-FM method developed by Bai, Kao, and Ng (2009) to deal with endogeneity and cross-sectional dependence issues.

Regarding the turning point, results in several studies showed a wide range. For instance, Grossman and Krueger (1995) indicated that the turning point of the EKC curve was around $8,000 per capita for most pollutants. The turning point for SO2 emissions estimated by Stern and Common (2001) was over $100,000. This may arise from the difference in samples, pollutants, models, and econometric methods.

The earliest models were simple quadratic functions of per capita income. Then some researchers included a GDP-cubed term and found that the relationship is of an N (or S) shape, implying that environmental degradation accelerates again after decreasing to a certain level (Kijima et al. 2010). Zheng et al. (2013) showed an S shape of the linkage between GDP and particulate matter (PM10) using PRC city-level data and got the peak point at around $4,900 per capita. In this chapter, we adopt the quadratic function of per capita income due to a relatively short time span.

9.3 Data and Methodology

9.3.1 The Empirical Model

We consider the following EKC model:

\[
\ln Pollution_{it} = \alpha + \beta_1 \ln GDP_{it} + \beta_2 (\ln GDP_{it})^2 \\
+ \beta_3 \ln Trade_{it} + \beta_4 \ln Electricity_{it} \\
+ \beta_5 \ln Density_{it} + \beta_6 \ln Manufacturing_{it} + \beta_7 \ln Service_{it} + \epsilon_{it} \]  (1)

where \( \ln Pollution, \ln GDP, \ln Trade, \) and \( \ln Electricity \) are logarithms of pollution, real GDP, total trade, and industrial electricity consumption, all expressed in per capita terms. The variable \( \ln Density \) is the logarithm
of population density. The variables *Manufacturing* and *Service* are the ratio of output in the second and third industry to gross output, measuring the composition effect. The superscripts *i* and *t* denote prefecture city and year, respectively.

In this chapter, we examine two types of pollutants, industrial wastewater and industrial SO$_2$ emissions, because they are among the most commonly used indicators of pollution in cities, and their severe effects on the environment and human health have been recognized for a long time (Grossman and Krueger 1995). In particular, due to the heavy reliance on coal for electricity generation, SO$_2$ emissions are the source of many serious environmental problems in the PRC. According to the EKC theory, an inverted U-shaped relationship exists between income and pollution; thus, we expect $\beta_1$ to be positive and $\beta_2$ to be negative. Electricity consumption tends to cause more pollution; therefore, $\beta_4$ is likely to be positive. Since population density and the structure of the economy are also likely to affect the level of pollution through the scale and composition effect (Grossman and Krueger 1995; Cole 2004), they are included in the regression specification as well. The relationship between trade and environment is the focus of this chapter. Besides using the measure of *lnTrade* as a robustness check, we also use alternate proxies such as the ratio of total trade to GDP (*trade share*) and the pair of *lnExport* and *lnImport* to explore further how the openness of the economy is related to the environment in a city. While no existing studies have considered the impact of export and import separately, these two factors may affect the pollution in local cities quite differently. Cities with huge export of manufacturing goods tend to be more polluted than those with huge import of manufacturing goods, even if the total trade is assumed to be the same. We therefore conduct another regression including both export and import simultaneously. The analysis for the whole of the PRC is repeated for three geographical regions: eastern, middle, and western PRC. Eastern PRC accounts for the largest share of the trade and includes cities in the provinces of Hebei, Liaoning, Shandong, Jiangsu, Zhejiang, Fujian, Guangdong, and Henan, as well as the municipalities of Tianjin, Beijing, and Shanghai. Central PRC includes cities in Shanxi, Jilin, Anhui, Jiangxi, Henan, and Hubei provinces. The rest belongs to western PRC.

All data are obtained from the China City Statistical Yearbook (China National Bureau of Statistics, various years) and the CEIC China Premium Database. The definitions of variables and the summary statistics are given in Table 9.1.
### Table 9.1: List of Variables and Summary Statistics

<table>
<thead>
<tr>
<th>Variables</th>
<th>Definitions</th>
<th>Mean</th>
<th>Std. Dev.</th>
<th>Min.</th>
<th>Max.</th>
<th>Obs.</th>
</tr>
</thead>
<tbody>
<tr>
<td>lnWater</td>
<td>Logarithm of per capita industrial wastewater (kg)</td>
<td>4.940</td>
<td>0.908</td>
<td>1.136</td>
<td>8.581</td>
<td>2610</td>
</tr>
<tr>
<td>lnGas</td>
<td>Logarithm of per capita industrial SO$_2$ (kg)</td>
<td>2.490</td>
<td>1.084</td>
<td>–7.097</td>
<td>5.553</td>
<td>2610</td>
</tr>
<tr>
<td>lnGDP</td>
<td>Logarithm of per capita real GDP (yuan in constant 2002 price)</td>
<td>9.837</td>
<td>0.681</td>
<td>7.727</td>
<td>11.855</td>
<td>2610</td>
</tr>
<tr>
<td>lnGDP squared</td>
<td>Square of the logarithm of per capita real GDP</td>
<td>97.223</td>
<td>13.465</td>
<td>59.703</td>
<td>140.551</td>
<td>2610</td>
</tr>
<tr>
<td>lnTrade</td>
<td>Logarithm of per capita real total trade (used in constant 2002 price)</td>
<td>5.462</td>
<td>1.953</td>
<td>–0.751</td>
<td>11.783</td>
<td>2610</td>
</tr>
<tr>
<td>Trade share</td>
<td>Ratio of total trade to GDP</td>
<td>0.231</td>
<td>0.413</td>
<td>0.001</td>
<td>4.622</td>
<td>2610</td>
</tr>
<tr>
<td>lnExport</td>
<td>Logarithm of per capita real export (used in constant 2002 price)</td>
<td>4.924</td>
<td>1.970</td>
<td>–4.785</td>
<td>11.219</td>
<td>2610</td>
</tr>
<tr>
<td>lnImport</td>
<td>Logarithm of per capita real import (used in constant 2002 price)</td>
<td>4.168</td>
<td>2.387</td>
<td>–5.296</td>
<td>10.942</td>
<td>2610</td>
</tr>
<tr>
<td>lnElectricity</td>
<td>Logarithm of per capita industrial electricity consumption (million kWh)</td>
<td>0.837</td>
<td>1.231</td>
<td>–3.868</td>
<td>4.267</td>
<td>2610</td>
</tr>
<tr>
<td>lnDensity</td>
<td>Logarithm of population per square kilometer (people/km$^2$)</td>
<td>5.782</td>
<td>0.901</td>
<td>1.562</td>
<td>7.840</td>
<td>2610</td>
</tr>
<tr>
<td>Manufacturing</td>
<td>Share of output from the second industry in total output (%)</td>
<td>49.798</td>
<td>10.721</td>
<td>15.7</td>
<td>90.97</td>
<td>2610</td>
</tr>
<tr>
<td>Service</td>
<td>Share of output from the third industry in total output (%)</td>
<td>35.740</td>
<td>8.550</td>
<td>0.0</td>
<td>76.85</td>
<td>2610</td>
</tr>
</tbody>
</table>

GDP = gross domestic product, kg = kilogram, km$^2$ = square kilometers, kWh = kilowatt hour, SO$_2$ = sulfur dioxide.

Source: Authors.
9.3.2 Econometric Methods

Different econometric methods have been used in the EKC literature to estimate the coefficients in the equations. Li, Wang, and Zhao (2016) use system GMM in a dynamic panel model and an ARDL model to investigate the EKC hypothesis in 28 PRC provinces. Kang, Zhao, and Yang (2016) adopt a spatial panel model to examine the EKC hypothesis on CO$_2$ in 30 provinces. Kasman and Duman (2015) employ the fully modified ordinary least squares method to study the dynamic causal relationship between CO$_2$, GDP, energy use, trade openness, and urbanization in 15 European countries for 1992–2010. In this chapter, we use an innovative estimation, namely, the Cup-FM method proposed by Bai, Kao, and Ng (2009). The Cup-FM estimator has recently been used in the energy–growth nexus literature (Fang and Chang 2016; Fang and Chen 2017; Chen and Fang 2017). Compared to other estimation methods, the Cup-FM estimator allows for cross-sectional dependence and endogeneity. Furthermore, it also has good finite sample properties. So, it fits the city panel in this chapter well.

To use the Cup-FM estimate, we conduct a few pretests, including a panel unit root test and a panel cointegration test. First, the cross-sectional dependence tests proposed by Frees (1995) and Pesaran (2004) are employed to determine the appropriate panel unit root tests. Frees’s (1995) test and Pesaran’s (2004) cross-sectional dependence test are selected because they are valid for samples with large $N$ and small $T$. Both tests have the null hypothesis of independence across units, and the test statistics follow the standard normal distribution asymptotically.

If the data are cross-sectional dependent, the second-generation panel unit root tests that allow for interdependence across units (Phillips and Sul 2003, Bai and Ng 2004, Pesaran 2007) should be employed. This chapter employs the popular CIPS proposed by Pesaran (2007) to test for the presence of unit roots. The CIPS test considers the following specification:

$$
\Delta y_{it} = \alpha_i + \rho_i y_{i,t-1} + \beta_0 \bar{y}_{t-1} + \sum_{j=0}^{p} d_{j+1} \Delta \bar{y}_{t-j} + \sum_{j=1}^{p} c_j \Delta y_{i,t-j} + \epsilon_{it} \quad (2)
$$

where $\alpha_i$ is the deterministic term, $\bar{y}_t$ is the mean of the dependent variable $y_{it}$ for all $N$ observations at time $t$, and $\rho$ is the lag order selected according to an information criterion. Let $t_i$ be the t-statistics of the coefficient estimates of $\rho_i$ for unit $i$. The CIPS test statistic is then defined as $CIPS = \frac{1}{N} \sum_{i=1}^{N} t_i$. For data with $T$ less than 20, a truncated CIPS test is proposed to avoid the problem of oversize, and the truncated values for the t-statistics vary for various deterministic terms (Pesaran 2007). Since we consider the annual data from 2004 to
2013, a truncated CIPS test statistic is used in this chapter to examine whether the series are stationary.

If the series are not stationary, we proceed to test for a cointegrating relationship. To account for the cross-sectional dependence in the data, panel cointegration tests such as that of Westerlund (2007) are preferred. However, it is not applicable due to the short time span considered in this chapter. Following Chen and Fang (2017), the traditional seven statistics of Pedroni (1999; 2004) are considered instead to get a clue about the cointegrating relationship among the variables. The first step is to estimate the residuals based on the model

\[ y_{it} = \alpha_i + \sum_{j=1}^{k} \beta_{ij} X_{jit} + \epsilon_{it} \quad (3) \]

where \( y \) and \( X \) are assumed to be integrated of order one, the superscripts \( t \) and \( i \) are the cross-sectional unit and time period, respectively, and \( k \) denotes the number of regressors. The second step is to test for the presence of unit roots in the residuals \( r_{it} = \rho_i r_{it-1} + u_{it} \). Under the null hypothesis of no cointegration, we shall get \( \rho_i = 1 \). For the four panel statistics, the alternative hypothesis is \( \rho_i = \rho < 1 \) for all units; and for the other three group statistics, the alternative hypothesis is \( \rho_i < 1 \) for all \( i \) where \( \rho_i \) can be heterogeneous across units. Among the seven statistics, we rely more on the two Augmented Dickey-Fuller (ADF) statistics that perform well in the small \( T \) panel (Wagner and Hlouskova 2009).

After confirming the existence of a cointegrating relationship among the set of variables, we estimate the coefficients using the Cup-FM method. Following Bai, Kao, and Ng (2009), we consider the model

\[ y_{it} = \alpha_i + \sum_{j=1}^{k} \beta_{ij} X_{jit} + \epsilon_{it}, \]

where the error term \( \epsilon_{it} \) follows a factor model \( \epsilon_{it} = \lambda_i F_t + u_{it} \). The common factor is to model the cross-sectional dependence. The Cup-FM estimator is then given by

\[ (\hat{\beta}_{Cup}, \hat{F}_{Cup}) = \text{argmin} \frac{1}{nT^2} \sum_{i=1}^{n} (y_i - x_i \beta)'M_F (y_i - x_i \beta) \quad (4) \]

where \( M_F = I_T - T^{-2} F F' \), and \( I_T \) is the identity matrix of dimension \( T \). To get the estimator, an initial value is assigned to \( F \) and estimations are repeated until convergence. The Cup-FM estimator obtained from the iterated procedure is proved to be \( \sqrt{nT} \) consistent. Moreover, it allows for cross-sectional dependence and endogeneity, and has good small-sample properties.

For comparison, we also conduct fixed-effects regressions, and the difference and system GMM estimation. The GMM method is appropriate for the dynamic panel with small \( T \) and large \( N \) (Arellano and Bond 1991; Arellano and Bover 1995; Blundell and Bond 1998).
Consider the following reduced-form dynamic panel model:

\[ \ln y_{it} = \alpha \ln y_{it-1} + \beta X_{it} + \nu_i + \varphi_t + \epsilon_{it} \]  

(5)

where \( \ln y \) is the logarithm of pollutant; \( X \) is the vector of explanatory variables including \( \ln GDP, \ln GDP \text{ squared}, \text{ openness}, \ln Electricity \); the share of output from the manufacturing and service sector in gross output; and \( \nu_i \) and \( \varphi_t \) represent the individual fixed city effect and time effect. Therefore, \( \alpha \) captures the dynamic effect and \( \beta \) is a vector of coefficients associated with the explanatory variables. The difference GMM eliminates the endogeneity in the dynamic model by employing the lagged variables as the instruments (Arellano and Bond 1991). The system GMM augments the difference GMM by adding the level equation and additional instruments to solve the issue that lagged variables perform poorly for first-differenced variables (Blundell and Bond 1998).

Besides estimating the coefficients, we perform the panel Granger non-causality test proposed by Dumitrescu and Hurlin (2012). It is selected because of its good properties in the small sample and it allows for heterogeneity and cross-sectional dependence using a block bootstrap procedure. Consider the panel model:

\[ y_{i,t} = \alpha_i + \sum_{k=1}^{K} \rho_{i,k} y_{i,t-k} + \sum_{k=1}^{K} \beta_{i,k} x_{i,t-k} + \epsilon_{i,t} \]  

(6)

To test whether the series \( \{x_{i,t}\} \) Granger-cause \( \{y_{i,t}\}, \) Dumitrescu and Hurlin (2012) propose following these steps:

(i) Get the standardised test statistics \( Z_{N,T} = \frac{\sqrt{N}}{2K} (W_{N,T} - K) \) and \( \tilde{Z}_N = \frac{\sqrt{N} \{W_{N,T} - E(\tilde{W}_{i,T})\}}{\sqrt{\text{var}(\tilde{W}_{i,T})}} \) where \( W_{N,T} \) is the average of \( N \) Wald statistics \( \tilde{W}_{i,T} \) obtained from the estimation for each cross-section unit.

(ii) To get the empirical critical values, assume \( \beta_{i,k} = 0 \) and get the estimates \( \hat{\alpha}_i, \hat{\rho}_{i,k}, \) and residuals \( \hat{\epsilon}_{i,t}. \) Next, construct a new series \( \{\tilde{y}_{i,t}\} \) where \( \tilde{y}_{i,t} = \hat{\alpha}_i + \sum_{k=1}^{K} \hat{\rho}_{i,k} y_{i,t-k} + \hat{\epsilon}_{i,t}, \) and compute the test statistics using the new series \( \{\tilde{y}_{i,t}\}. \) Here \( \tilde{\epsilon}_{i,t} \) is resampled from the residual series with a replacement. Repeat this step several times and we can get the distribution of test statistics, and therefore the empirical critical values at a given significance level.

(iii) Compare the test statistics with empirical critical values and draw a conclusion regarding the hypothesis.
9.4 Results and Discussion

9.4.1 Panel Unit Root and Panel Cointegration Test

Table 9.2 shows Frees’s (1995) Q statistics and Pesaran’s (2004) test statistics for both industrial wastewater and SO₂ as the dependent variable. Each column uses one measure of openness in the regression specification; a total of three different measures are used in this chapter. The test results show that no matter which measurement of openness is employed and which type of pollutant is considered, there is a strong indication that the data are cross-sectionally dependent.

<table>
<thead>
<tr>
<th>Test</th>
<th>lnTrade</th>
<th>Trade Share</th>
<th>lnExport, lnImport</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Pollutant: Industrial wastewater</strong></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Frees (1995) Q</td>
<td>32.768***</td>
<td>32.783***</td>
<td>32.784***</td>
</tr>
<tr>
<td><strong>Pollutant: SO₂</strong></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Frees (1995) Q</td>
<td>38.409***</td>
<td>37.390***</td>
<td>37.894***</td>
</tr>
<tr>
<td>Pesaran (2004)</td>
<td>33.576***</td>
<td>33.574***</td>
<td>31.061***</td>
</tr>
<tr>
<td>N</td>
<td>2610</td>
<td>2610</td>
<td>2610</td>
</tr>
</tbody>
</table>

SO₂ = sulfur dioxide.

Notes:
1. The null hypothesis of both tests is cross-sectional independence;
2. *** denotes a significance level of 1%;
3. Trade share is (export+import)/gross domestic product;
4. The pair of lnExport and lnImport are included in the regression as a measure of openness.

Source: Authors.

Given the interdependence across cities, Pesaran’s (2007) CIPS panel unit root test is applied. Table 9.3 reports the test results for variables both at level and after first difference. Two specifications are considered, one with only the intercept and the other with both the intercept and trend pattern. The results show that the variables of lnWater, lnGas, lnTrade, Trade share, lnExport, lnDensity, Manufacturing, and Service are not stationary at levels, but are stationary after first difference. So, these variables are integrated of order one. Other variables—lnGDP, lnGDP squared, lnImport, and lnElectricity—show some evidence of being stationary at levels; but they are more stationary
after first difference. In general, at a 1% significance level, all variables can be considered as an integration of order one. So, we proceed with the panel cointegration test in the next step.

As explained in the methodology, Westerlund’s (2007) panel cointegration test that allows for cross-sectional dependence is preferred. However, due to the short time span, it cannot be applied in this context. Alternatively, Pedroni’s (1999; 2004) traditional seven test statistics are used to test for panel cointegration. The two ADF statistics are reported in Table 9.4 because they are found to perform well in the panel with a short time span (Wagner and Hlouskova 2009). As shown in Table 9.4, the null hypothesis that there is no cointegrating relationship is rejected at the 1% significance level in all specifications of both pollutants. This suggests that under the assumption of no cross-sectional dependence, the seven variables—pollutant, GDP, GDP squared, openness measure, electricity consumption, population density, and economic structure in terms of output share of the secondary and tertiary industries—are cointegrated.

Subsequently, we estimate the regression equation using different econometric methods that are appropriate under different assumptions.
9.4.2 Estimation Results

We first compare the estimation results from fixed-effect regressions, difference GMM, system GMM, and the Cup-FM method. Coefficient estimates using various methods for the industrial wastewater specification where total trade is incorporated as the measure of openness are reported in Table 9.5. Columns 2 and 3 show the coefficients of the fixed-effects estimation. It is noted that the coefficient of $\ln GDP$ is positive while the coefficient of $\ln GDP$ squared is negative, both at the 10% significance level. This is consistent with the EKC hypothesis, implying that industrial wastewater emissions first increase with economic development, and then start to decrease after GDP reaches a turning point. It can be calculated from the fixed-effect estimates of $\ln GDP$ and $\ln GDP$ squared that the turning point appears when per capita GDP reaches CNY12,274 (in constant 2002 prices). This is much lower than the turning points found in previous studies. The coefficient estimates from fixed-effects regressions are biased because they do not consider the endogeneity and interdependence across units.

In the difference and system GMM estimation in columns 4 to 7, the lagged dependent variable has a significant and positive impact on the dependent variable $\ln Water$, confirming a dynamic relationship. The test AR(1) is rejected and AR(2) cannot be rejected, suggesting that the assumption that residuals are not serially correlated at the second order is satisfied. Hansen test results show that the second assumption of

### Table 9.4: Pedroni’s Panel Cointegration Test

<table>
<thead>
<tr>
<th>Pollutant: Industrial wastewater</th>
<th>(\ln \text{Trade})</th>
<th>(\text{Trade Share})</th>
</tr>
</thead>
<tbody>
<tr>
<td>Panel ADF</td>
<td>-17.800***</td>
<td>-22.930***</td>
</tr>
<tr>
<td>Group ADF</td>
<td>-19.220***</td>
<td>-29.380***</td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>Pollutant: SO₂</th>
<th>(\ln \text{Trade})</th>
<th>(\text{Trade Share})</th>
</tr>
</thead>
<tbody>
<tr>
<td>Panel ADF</td>
<td>-18.880***</td>
<td>-23.730***</td>
</tr>
<tr>
<td>Group ADF</td>
<td>-21.590***</td>
<td>-29.380***</td>
</tr>
</tbody>
</table>

ADF = Augmented Dickey-Fuller, SO₂ = sulfur dioxide.

Notes:
1. The lag order is determined by the Schwarz information criterion;
2. *** denotes a significance level of 1%;
3. Results are not available for the specification with export and import.

Source: Authors.
overidentification restriction is satisfied, suggesting that the instruments are valid. However, the coefficient estimates for $\ln{\text{GDP}}$ and $\ln{\text{GDP squared}}$ using the difference GMM method are insignificant. While there may be a weak instrumental variable problem in the difference GMM estimator (Arellano and Bover 1995; Blundell and Bond 1998), the coefficients of the two GDP variables in the system GMM method are also insignificant, and the signs are even unexpected. Results from both the difference and system GMM methods suggest that the inverted U-shaped EKC is not supported. This is inconsistent with Li, Wang, and Zhao (2016), who used the system GMM estimator for a panel of 28 provinces in the PRC and supported the EKC hypothesis. This is likely to be caused by the large N and small T panel and the presence of cross-sectional dependence because the underlying assumption that the residuals are uncorrelated across units is violated and, therefore, the results are not reliable.

As explained in the methodology part, this chapter prefers the Cup-FM estimator that allows for both endogeneity and cross-sectional dependence. As reported in the last two columns of Table 9.5, all coefficient estimates are significant at 1%. The EKC is well supported in line with findings in existing literature in the Chinese context (Kang et al. 2016; Li et al. 2016). The results imply that an inverted U-shaped relationship exists between industrial wastewater and income for PRC cities, and the turning point is estimated to happen at around CNY49,446 (in constant 2002 prices). Furthermore, trade is found to reduce the industrial wastewater pollution. Specifically, a 1% increase in total trade is associated with a 0.04% reduction of wastewater pollution. This may suggest that the technical effect dominates in the trade–environment nexus (Grossman and Krueger 1993).

Next, we use the preferred Cup-FM estimator for all the model specifications to examine the relationship between the environment, development, and openness in the whole PRC and in its different regions.

The Cup-FM estimates of key variables for various specifications using $\ln{\text{Trade}}$, $\text{Trade share}$, and the pair of $\ln{\text{Export}}$ and $\ln{\text{Import}}$ as the measure of openness are reported in Table 9.6. The results support the EKC hypothesis for both pollutants of industrial wastewater and $\text{SO}_2$ emissions. Industrial wastewater is found to increase with per capita GDP until it reaches CNY31,849–CNY49,446 (in constant 2002 prices), depending on which openness measure is considered. As for the relationship between trade and pollution, as explained earlier, the coefficient estimates suggest that a 1% increase in total trade is

---

2 Due to space considerations, results for other control variables are not reported but are available upon request from the authors.
<table>
<thead>
<tr>
<th></th>
<th>Fixed Effect</th>
<th>Difference GMM</th>
<th>System GMM</th>
<th>Cup-FM</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Coefficient</td>
<td>Robust SE</td>
<td>Coefficient</td>
<td>Robust SE</td>
</tr>
<tr>
<td>lnGDP</td>
<td>1.111*</td>
<td>(0.670)</td>
<td>0.377</td>
<td>(0.900)</td>
</tr>
<tr>
<td>lnGDP squared</td>
<td>-0.059*</td>
<td>(0.033)</td>
<td>-0.003</td>
<td>(0.045)</td>
</tr>
<tr>
<td>lnTrade</td>
<td>0.003</td>
<td>(0.037)</td>
<td>0.083</td>
<td>(0.052)</td>
</tr>
<tr>
<td>lnElectricity</td>
<td>0.046*</td>
<td>(0.024)</td>
<td>0.018</td>
<td>(0.026)</td>
</tr>
<tr>
<td>lnDensity</td>
<td>0.082</td>
<td>(0.462)</td>
<td>0.294</td>
<td>(0.545)</td>
</tr>
<tr>
<td>Manufacturing</td>
<td>-0.002</td>
<td>(0.007)</td>
<td>-0.013*</td>
<td>(0.008)</td>
</tr>
<tr>
<td>Service</td>
<td>-0.000</td>
<td>(0.007)</td>
<td>-0.003</td>
<td>(0.005)</td>
</tr>
<tr>
<td>L.lnWater</td>
<td>0.518***</td>
<td>(0.050)</td>
<td>0.849***</td>
<td>(0.024)</td>
</tr>
<tr>
<td>AR1</td>
<td>[0.000]</td>
<td></td>
<td>[0.000]</td>
<td></td>
</tr>
<tr>
<td>AR2</td>
<td>[0.369]</td>
<td></td>
<td>[0.332]</td>
<td></td>
</tr>
<tr>
<td>Hansen test</td>
<td>[0.678]</td>
<td></td>
<td>[0.996]</td>
<td></td>
</tr>
</tbody>
</table>

# of observations                      2610  2088  2349  2610

Cup_FM = continuously updated fully modified, GDP = gross domestic product, GMM = generalized method of moments, SE = standard error.

Notes:
1. The result is for the specification of the industrial wastewater and variables including lnGDP, lnGDP squared, openness, lnElectricity, lnDensity, Manufacturing, and Service;
2. L.lnWater is one year lag of lnWater;
3. AR(1) and AR(2) are tests for autocorrelation in differenced residuals; the Hansen test is for testing for overidentification restrictions;
4. ***, **, and * denote a significance level of 1%, 5%, and 10%, respectively.

Source: Authors.
associated with a 0.04% reduction in industrial wastewater. That trade helps improve the environment is also evidenced in the second model, while the negative coefficient is only significant at the 10% significance level. Similar conclusions are found by Birdsall and Wheeler (2001), Ferrantino (1997), and Grether et al. (2007). However, it seems that export and import alone do not matter too much in reducing industrial water pollution. If we look at SO₂ emissions, while the EKC hypothesis still holds, the turning point is estimated at around CNY9,274–CNY10,103 per capita income (in constant 2002 prices). On the other hand, total trade is found to be positively related to SO₂ emissions at the 5% significance level. Specifically, if total trade increases by 1%, SO₂ emissions in that city tend to increase by 0.03%. This is consistent with findings in Ang (2009), Jalil and Feridun (2011), and Nasir and Rehman (2011). It should be noted, however, that export and import affect SO₂ emissions in a different way than industrial wastewater.

Table 9.6: Continuously Updated Fully Modified Estimation Results

<table>
<thead>
<tr>
<th></th>
<th>Pollutant: Industrial Wastewater</th>
<th>Pollutant: SO₂</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>(1)</td>
<td>(2)</td>
</tr>
<tr>
<td>lnGDP</td>
<td>1.751***</td>
<td>1.641***</td>
</tr>
<tr>
<td>lnGDP squared</td>
<td>-0.081***</td>
<td>-0.076***</td>
</tr>
<tr>
<td></td>
<td>(-10.813)</td>
<td>(-11.008)</td>
</tr>
<tr>
<td>lnTrade</td>
<td>-0.044***</td>
<td></td>
</tr>
<tr>
<td></td>
<td>(-3.473)</td>
<td></td>
</tr>
<tr>
<td>Trade share</td>
<td>-0.086*</td>
<td></td>
</tr>
<tr>
<td></td>
<td>(-1.925)</td>
<td></td>
</tr>
<tr>
<td>lnExport</td>
<td>0.001</td>
<td></td>
</tr>
<tr>
<td></td>
<td>(0.130)</td>
<td></td>
</tr>
<tr>
<td>lnImport</td>
<td>0.004</td>
<td></td>
</tr>
<tr>
<td></td>
<td>(0.660)</td>
<td></td>
</tr>
<tr>
<td>No. of observations</td>
<td>2610</td>
<td>2610</td>
</tr>
</tbody>
</table>

GDP = gross domestic product, SO₂ = sulfur dioxide.

Notes:
1. Other controls include lnElectricity, lnDensity, Manufacturing, and Service;
2. ***, **, and * denote a significance level of 1%, 5%, and 10%, respectively.
Source: Authors.
emissions differently. If the total trade increase is from export, SO$_2$ emissions could be even worse; if the total trade increase is due to increased import, SO$_2$ emissions may be slightly reduced. Partly due to the offsetting effect, trade share is found to be insignificant in line with Li, Wang, and Zhao (2016) using PRC provincial data.

### 9.4.3 Estimation Results by Region

We repeat the analysis by region. Specifically, the cities are grouped according to their geographical locations and three subsamples are formed. They are eastern PRC, central PRC, and western PRC, which comprise 101, 97, and 63 cities, respectively. Appendix Table 9.A1 shows the cross-sectional dependence test results for the three regions in various specifications. The null hypothesis of independence across cities is rejected by Frees’s (1995) statistics in all specifications. Pesaran’s (2004) test statistics also reject the null hypothesis in most specifications (except for the specification of wastewater in central PRC and western PRC when trade is used to measure the openness). Overall, the test results support interdependence across cities in eastern, central, and western PRC in the context of the environment, development, and openness.

Next, Appendix Table 9.A2 reports the panel unit root test results using Pesaran’s (2007) CIPS statistics for all variables considered in this chapter for the three regions. Similar to the tests for the whole PRC, two model specifications (i.e., one with only the intercept and another with both the intercept and trend) are considered for the level variable and the differenced variable. It is observed that the null hypothesis that unit root exists is rejected in all the tests when the variables are first differenced and the model contains only the intercept, at least at the 10% significance level. To conclude, evidence is strong that all variables (except lnGDP and Service in the eastern PRC) show the property of integration of order one. Table 9.A3 shows the panel cointegration test results using Pedroni’s (1999, 2004) panel and group ADF statistics, which are appropriate when the T is small. Though they do not consider the cross-sectional dependence, the results at least lend some support to a long-run cointegrating relationship among the variables. So, our discussion will subsequently focus on the coefficient estimates reported in Table 9.7.

There are two noteworthy findings from Table 9.7. First is that no matter which measure of openness is employed in the model and which pollutant is considered, the coefficients of lnGDP are all positive, and the coefficients of lnGDP squared are all negative. Therefore, the EKC hypothesis is not only supported using city data from the whole of the PRC but also by city data in different regions.
Table 9.7: Continuously Updated Fully Modified Estimation Results by Region

<table>
<thead>
<tr>
<th>Pollutant: Industrial wastewater</th>
<th>Eastern PRC</th>
<th>Central PRC</th>
<th>Western PRC</th>
</tr>
</thead>
<tbody>
<tr>
<td>lnGDP</td>
<td>3.181***</td>
<td>1.396***</td>
<td>0.794***</td>
</tr>
<tr>
<td></td>
<td>(11.324)</td>
<td>(3.137)</td>
<td>(5.898)</td>
</tr>
<tr>
<td>lnGDP squared</td>
<td>-0.165***</td>
<td>-0.081***</td>
<td>-0.018**</td>
</tr>
<tr>
<td></td>
<td>(-11.754)</td>
<td>(-3.451)</td>
<td>(-2.342)</td>
</tr>
<tr>
<td>lnTrade</td>
<td>-0.233***</td>
<td>0.008</td>
<td></td>
</tr>
<tr>
<td></td>
<td>(-9.025)</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Trade share</td>
<td>-0.037</td>
<td>0.274**</td>
<td></td>
</tr>
<tr>
<td></td>
<td>(-0.868)</td>
<td>(2.275)</td>
<td></td>
</tr>
<tr>
<td>lnExport</td>
<td>-0.243***</td>
<td>0.014</td>
<td></td>
</tr>
<tr>
<td></td>
<td>(-9.258)</td>
<td>(1.136)</td>
<td></td>
</tr>
<tr>
<td>lnImport</td>
<td>0.029</td>
<td>0.021**</td>
<td></td>
</tr>
<tr>
<td></td>
<td>(1.623)</td>
<td>(2.100)</td>
<td></td>
</tr>
</tbody>
</table>

continued on next page
Table 9.7 continued

<table>
<thead>
<tr>
<th>Pollutant: $\text{SO}_2$</th>
<th>Eastern PRC</th>
<th>Central PRC</th>
<th>Western PRC</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>(1)</td>
<td>(2)</td>
<td>(3)</td>
</tr>
<tr>
<td>InGDP</td>
<td>1.081***</td>
<td>1.160***</td>
<td>0.899***</td>
</tr>
<tr>
<td>lnGDP squared</td>
<td>-0.049***</td>
<td>-0.054***</td>
<td>-0.036***</td>
</tr>
<tr>
<td></td>
<td>(-5.181)</td>
<td>(-5.858)</td>
<td>(-3.893)</td>
</tr>
<tr>
<td>lnTrade</td>
<td>-0.026</td>
<td>0.050***</td>
<td>-0.012</td>
</tr>
<tr>
<td></td>
<td>(-0.828)</td>
<td>(3.224)</td>
<td>(-0.641)</td>
</tr>
<tr>
<td>Trade share</td>
<td>-0.002</td>
<td>-0.006</td>
<td>0.466**</td>
</tr>
<tr>
<td></td>
<td>(-0.032)</td>
<td>(-0.058)</td>
<td>(2.230)</td>
</tr>
<tr>
<td>lnExport</td>
<td>0.204***</td>
<td>0.030***</td>
<td>0.057***</td>
</tr>
<tr>
<td></td>
<td>(6.235)</td>
<td>(3.924)</td>
<td>(4.660)</td>
</tr>
<tr>
<td>lnImport</td>
<td>-0.129***</td>
<td>-0.052***</td>
<td>-0.032***</td>
</tr>
<tr>
<td></td>
<td>(-5.329)</td>
<td>(-6.658)</td>
<td>(-2.993)</td>
</tr>
<tr>
<td>No. of observations</td>
<td>1,010</td>
<td>970</td>
<td>630</td>
</tr>
</tbody>
</table>

Cup-FM = continuously updated fully modified, GDP = gross domestic product, $\text{SO}_2$ = sulfur dioxide.

Note: Other controls include lnElectricity, lnDensity, Manufacturing, and Service.

Source: Authors.
Second, trade has a heterogeneous effect on the environment by region. For the industrial wastewater, cities with a high level of openness (in terms of total trade) are found to be associated with less water pollution in the eastern PRC but more industrial water pollution in central PRC. Specifically, coefficient estimates of $\ln{\text{Trade}}$ and $\ln{\text{Export}}$ are found to be significantly negative in eastern PRC, while the coefficients of $\text{Trade share}$ and $\ln{\text{Import}}$ are significantly positive in central PRC. This seems to reflect that in the eastern PRC where the economy is more developed, the environmental regulations are more rigorous, there are more high-tech industries, and trade has a technical and composite effect on the environment in the sense that it helps reduce industrial wastewater pollution. On the other hand, in central PRC, the economic structure is yet to be upgraded and there are still some high-polluting industries; trade seems to exhibit a scale effect on the environment that more trade brings more wastewater pollution. In western PRC, while $\ln{\text{Trade}}$ and $\text{Trade share}$ are not found to affect the water pollution significantly, export is observed to have an adverse impact on the environment. In the least-developed regions, the scale effect of trade will likely dominate. If we look at the results for the pollutant $\text{SO}_2$ emissions, what seems to be striking is that in all the regions, export and import affect the $\text{SO}_2$ emissions in a similar way. More export tends to increase $\text{SO}_2$ emissions, while more import tends to decrease $\text{SO}_2$ emissions. So, for the pollutant $\text{SO}_2$, more regulations for the trade industries are necessary.

How will trade openness likely affect our results? To investigate this question, we calculate the city openness index by averaging the trade shares for 2004–2013 for each individual city and separate the cities into high openness and low openness by comparing their openness index to the median. As a result, we get 131 cities in the high-openness group and 130 cities in the low-openness group. The Cup-FM estimation results are reported in Table 9.8. It is interesting to find that the EKC hypothesis only holds for the high-openness cities and is invalid for the low-openness cities. Furthermore, in the sample of high-openness cities in Table 9.8, $\ln{\text{Trade}}$ and $\ln{\text{Export}}$ reduce industrial wastewater pollution significantly. Combining the results with those by region confirms that high openness is associated with less industrial water pollution.

### 9.4.4 Granger Causality Results

Lastly, we conduct the panel Granger noncausality test to confirm whether a causal relationship exists. The results of the panel Granger noncausality test employing Dumitrescu and Hurlin’s (2012) method are shown in Table 9.9. For the whole PRC, all the trade openness measures Granger-cause the pollutant variable, be it industrial wastewater or
<table>
<thead>
<tr>
<th>Pollutant: Industrial wastewater</th>
<th>High Openness</th>
<th>Low Openness</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>(1)</td>
<td>(2)</td>
</tr>
<tr>
<td>lnGDP</td>
<td>2.479***</td>
<td>2.379***</td>
</tr>
<tr>
<td></td>
<td>(15.734)</td>
<td>(17.113)</td>
</tr>
<tr>
<td>lnGDP squared</td>
<td>-0.130***</td>
<td>-0.131***</td>
</tr>
<tr>
<td></td>
<td>(-15.536)</td>
<td>(-17.234)</td>
</tr>
<tr>
<td>lnTrade</td>
<td>-0.185***</td>
<td></td>
</tr>
<tr>
<td></td>
<td>(-9.058)</td>
<td></td>
</tr>
<tr>
<td>Trade share</td>
<td>-0.010</td>
<td></td>
</tr>
<tr>
<td></td>
<td>(-0.248)</td>
<td></td>
</tr>
<tr>
<td>lnExport</td>
<td>-0.110***</td>
<td></td>
</tr>
<tr>
<td></td>
<td>(-6.236)</td>
<td></td>
</tr>
<tr>
<td>lnImport</td>
<td>-0.012</td>
<td></td>
</tr>
<tr>
<td></td>
<td>(-0.832)</td>
<td></td>
</tr>
</tbody>
</table>

*continued on next page*
Table 9.8 continued

<table>
<thead>
<tr>
<th>Pollutant: SO₂</th>
<th>Low Openness</th>
<th>High Openness</th>
<th></th>
<th></th>
<th></th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>(1)</td>
<td>(2)</td>
<td>(3)</td>
<td>(1)</td>
<td>(2)</td>
<td>(3)</td>
</tr>
<tr>
<td>lnGDP</td>
<td>1.231***</td>
<td>1.667***</td>
<td>1.127***</td>
<td>-0.471</td>
<td>-0.617**</td>
<td>-0.491*</td>
</tr>
<tr>
<td>lnGDP squared</td>
<td>-0.064***</td>
<td>-0.093***</td>
<td>-0.059***</td>
<td>0.013</td>
<td>0.020</td>
<td>0.013</td>
</tr>
<tr>
<td>lnTrade</td>
<td>0.008</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>0.038***</td>
</tr>
<tr>
<td>Trade share</td>
<td>-0.002</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>1.210***</td>
</tr>
<tr>
<td>lnExport</td>
<td></td>
<td>0.107***</td>
<td></td>
<td></td>
<td></td>
<td>0.039***</td>
</tr>
<tr>
<td>lnImport</td>
<td></td>
<td>-0.050***</td>
<td></td>
<td></td>
<td></td>
<td>-0.028***</td>
</tr>
<tr>
<td>No. of observations</td>
<td>1,310</td>
<td>1,300</td>
<td>1,300</td>
<td>1,300</td>
<td>1,300</td>
<td>1,300</td>
</tr>
</tbody>
</table>

Cup-FM = continuously updated fully modified, GDP = gross domestic product, SO₂ = sulfur dioxide.
Note: Other controls include lnElectricity, lnDensity, Manufacturing, and Service.
Source: Authors.
SO₂ emissions. The Granger causal relationship running from trade to pollutant also holds in eastern PRC and central PRC (except for the pair of lnTrade and lnGas). In western PRC, only trade share is found to Granger-cause both pollutants, while there is also some evidence that total trade Granger-causes industrial wastewater. To conclude, the Granger causal relationship running from openness measure to industrial pollutants is generally confirmed, and the sign of the relationship should be inferred from the estimation discussed in the last subsection.

Table 9.9: Panel Granger Noncausality Test Results

<table>
<thead>
<tr>
<th>Test</th>
<th>PRC Test statistic</th>
<th>p-value</th>
<th>Eastern PRC Test statistic</th>
<th>p-value</th>
<th>Central PRC Test statistic</th>
<th>p-value</th>
<th>Western PRC Test statistic</th>
<th>p-value</th>
</tr>
</thead>
<tbody>
<tr>
<td>lnTrade→lnWater</td>
<td>3.858</td>
<td>0.000</td>
<td>2.428</td>
<td>0.015</td>
<td>2.5798</td>
<td>0.010</td>
<td>1.5776</td>
<td>0.115</td>
</tr>
<tr>
<td>lnTrade→lnGas</td>
<td>10.490</td>
<td>0.000</td>
<td>0.537</td>
<td>0.592</td>
<td>1.1097</td>
<td>0.267</td>
<td>1.9529</td>
<td>0.051</td>
</tr>
<tr>
<td>Trade share→lnWater</td>
<td>1.970</td>
<td>0.049</td>
<td>1.846</td>
<td>0.065</td>
<td>2.3250</td>
<td>0.020</td>
<td>7.1891</td>
<td>0.000</td>
</tr>
<tr>
<td>Trade share→lnGas</td>
<td>5.999</td>
<td>0.000</td>
<td>2.562</td>
<td>0.010</td>
<td>1.8523</td>
<td>0.064</td>
<td>2.1021</td>
<td>0.036</td>
</tr>
<tr>
<td>lnExport→lnWater</td>
<td>6.098</td>
<td>0.000</td>
<td>3.626</td>
<td>0.000</td>
<td>2.1089</td>
<td>0.035</td>
<td>1.0226</td>
<td>0.307</td>
</tr>
<tr>
<td>lnExport→lnGas</td>
<td>10.101</td>
<td>0.000</td>
<td>2.600</td>
<td>0.009</td>
<td>4.6078</td>
<td>0.000</td>
<td>0.5059</td>
<td>0.613</td>
</tr>
<tr>
<td>lnImport→lnWater</td>
<td>3.756</td>
<td>0.000</td>
<td>10.242</td>
<td>0.000</td>
<td>6.5316</td>
<td>0.000</td>
<td>0.278</td>
<td>0.781</td>
</tr>
<tr>
<td>lnImport→lnGas</td>
<td>5.498</td>
<td>0.000</td>
<td>2.840</td>
<td>0.005</td>
<td>6.1030</td>
<td>0.000</td>
<td>1.042</td>
<td>0.298</td>
</tr>
</tbody>
</table>

PRC = People’s Republic of China.
Note: The null hypothesis of the Dumitrescu and Hurlin (2012) test is no Granger causality.
Source: Authors.

9.5 Conclusions with Policy Recommendations

This chapter has examined the EKC hypothesis and investigated the extent to which trade has contributed to the EKC shape across cities in 2004–2013 in the PRC. To address criticisms on the methodology of the EKC’s empirical analysis raised in some recent studies (Stern 2004; Wagner 2008; Kijima et al. 2010), we adopt an innovative estimation, namely, the Cup-FM method, to deal with cross-sectional dependence and endogeneity problems. Compared with fixed-effects regressions, the difference and system GMM estimation, the Cup-FM estimator has good finite sample properties and provides more robust results.

We use industrial wastewater and SO₂ as indicators of environmental pollution and find that the EKC hypothesis holds for both the PRC and
its different regions. However, the EKC hypothesis is found to hold only for the high-openness cities and is invalid for the low-openness ones. Cities with higher openness tend to have lower industrial wastewater emission but higher $\text{SO}_2$ emissions. Specifically, a 1% increase in total trade is associated with a 0.04% reduction of industrial wastewater emissions.

The growing income gap between the three regions in the PRC and the pollution haven hypothesis have attracted a lot of attention. From the perspective of policy design, identifying the impact of trade and factors that dominate the downturn in emissions will enlighten the pollution–income paths of less-developed cities. An upgrade of the economic structure, promotion of open economies, and implementation of strict environmental regulations may be proposed in less developed and less open cities to achieve sustainable and green economic growth. Additionally, two issues need to be addressed for future research. First, the mechanism behind the fact that the EKC only holds for high-openness cities needs further exploration, and the impact of trade openness on $\text{SO}_2$ emissions calls for more careful modeling of the income–pollution relationship. Second, it is possible that downward sloping may arise because of the “race to the bottom” scenario. As more and more “dirty” industries are relocated across and within regions in the PRC and environmental regulations are increasingly emphasized in various cities, rich cities might not be able to find poorer cities to serve as pollution havens in the future and pollution may increase again. Hence, more fitting models with a longer time horizon are required.
References


### Table 9.A1: Cross-Sectional Dependence Test Results by Region

<table>
<thead>
<tr>
<th>Pollutant: Industrial wastewater</th>
<th>Test</th>
<th>lnTrade</th>
<th>Trade Share</th>
<th>lnExport, lnImport</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Pesaran (2004)</td>
<td>0.138</td>
<td>0.143</td>
<td>0.184</td>
</tr>
<tr>
<td>Western PRC</td>
<td>Frees (1995) Q</td>
<td>5.441***</td>
<td>5.858***</td>
<td>5.502***</td>
</tr>
<tr>
<td></td>
<td>Pesaran (2004)</td>
<td>0.425</td>
<td>0.533</td>
<td>0.393</td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>Pollutant: SO₂</th>
<th>Test</th>
<th>lnTrade</th>
<th>Trade Share</th>
<th>lnExport, lnImport</th>
</tr>
</thead>
</table>

SO₂ = sulfur dioxide.

Notes:
1. There are 1,010 observations in eastern PRC, 970 observations in central PRC, and 630 observations in western PRC for specifications using lnTrade, Trade share, and lnExport and lnImport as the openness measure.
2. The null hypothesis of both tests is cross-sectional independence.
3. *** denotes a significance level of 1%.

Source: Authors.
Table 9.A2: Panel Unit Root Test Results with Cross-Sectional Dependence by Region

<table>
<thead>
<tr>
<th>Samples</th>
<th>Level Intercept</th>
<th>First Difference Intercept and Trend</th>
<th>Level Intercept</th>
<th>First Difference Intercept and Trend</th>
</tr>
</thead>
<tbody>
<tr>
<td>Eastern PRC</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>lnWater</td>
<td>−1.336</td>
<td>−1.969</td>
<td>−2.636***</td>
<td>−3.403***</td>
</tr>
<tr>
<td>lnGas</td>
<td>−1.655</td>
<td>−2.866**</td>
<td>−2.698***</td>
<td>−2.606</td>
</tr>
<tr>
<td>lnGDP</td>
<td>−3.022***</td>
<td>−3.102***</td>
<td>−3.504***</td>
<td>−3.406***</td>
</tr>
<tr>
<td>lnGDP squared</td>
<td>−2.950***</td>
<td>−2.961**</td>
<td>−3.405***</td>
<td>−3.387***</td>
</tr>
<tr>
<td>lnTrade</td>
<td>−1.871</td>
<td>−1.930</td>
<td>−2.272*</td>
<td>−2.239</td>
</tr>
<tr>
<td>Trade share</td>
<td>−1.861</td>
<td>−2.073</td>
<td>−2.408**</td>
<td>−2.603</td>
</tr>
<tr>
<td>lnExport</td>
<td>−1.651</td>
<td>−1.656</td>
<td>−2.154*</td>
<td>−2.367</td>
</tr>
<tr>
<td>lnImport</td>
<td>−1.939</td>
<td>−1.834</td>
<td>−2.366**</td>
<td>−2.327</td>
</tr>
<tr>
<td>lnElectricity</td>
<td>−2.220*</td>
<td>−2.672</td>
<td>−2.922***</td>
<td>−2.846**</td>
</tr>
<tr>
<td>lnDensity</td>
<td>−1.536</td>
<td>−2.226</td>
<td>−2.640***</td>
<td>−2.811*</td>
</tr>
<tr>
<td>lnManufacturing</td>
<td>−1.321</td>
<td>−3.146***</td>
<td>−3.345***</td>
<td>−3.286***</td>
</tr>
<tr>
<td>lnService</td>
<td>−2.618***</td>
<td>−3.262***</td>
<td>−3.626***</td>
<td>−3.718***</td>
</tr>
<tr>
<td>Central PRC</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>lnWater</td>
<td>−1.984</td>
<td>−2.264</td>
<td>−2.700***</td>
<td>−3.107***</td>
</tr>
<tr>
<td>lnGDP</td>
<td>−1.714</td>
<td>−2.320</td>
<td>−2.322**</td>
<td>−2.123</td>
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<tr>
<td>lnGDP squared</td>
<td>−2.711***</td>
<td>−2.637</td>
<td>−2.828***</td>
<td>−2.824*</td>
</tr>
<tr>
<td>lnTrade</td>
<td>−1.775</td>
<td>−1.954</td>
<td>−2.580***</td>
<td>−2.861**</td>
</tr>
<tr>
<td>Trade share</td>
<td>−1.726</td>
<td>−2.052</td>
<td>−2.602***</td>
<td>−2.713</td>
</tr>
<tr>
<td>lnExport</td>
<td>−1.661</td>
<td>−1.789</td>
<td>−2.483**</td>
<td>−2.731*</td>
</tr>
<tr>
<td>lnImport</td>
<td>−2.495**</td>
<td>−2.445</td>
<td>−3.246***</td>
<td>−3.342***</td>
</tr>
<tr>
<td>lnElectricity</td>
<td>−2.062</td>
<td>−2.069</td>
<td>−2.935***</td>
<td>−3.328***</td>
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<tr>
<td>lnDensity</td>
<td>−1.495</td>
<td>−2.360</td>
<td>−2.609***</td>
<td>−2.900**</td>
</tr>
<tr>
<td>lnManufacturing</td>
<td>−2.480**</td>
<td>−2.446</td>
<td>−2.915***</td>
<td>−2.701</td>
</tr>
<tr>
<td>lnService</td>
<td>−2.387**</td>
<td>−2.394</td>
<td>−2.897***</td>
<td>−2.796*</td>
</tr>
<tr>
<td>Western PRC</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>lnWater</td>
<td>−2.045</td>
<td>−2.353</td>
<td>−2.719***</td>
<td>−2.001</td>
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<tr>
<td>lnGDP</td>
<td>−2.287*</td>
<td>−2.561</td>
<td>−3.014***</td>
<td>−2.750*</td>
</tr>
<tr>
<td>lnGDP squared</td>
<td>−2.624***</td>
<td>−2.540</td>
<td>−3.091***</td>
<td>−3.600***</td>
</tr>
<tr>
<td>lnTrade</td>
<td>−2.099</td>
<td>−1.879</td>
<td>−2.564***</td>
<td>−2.665</td>
</tr>
</tbody>
</table>

continued on next page
### Table 9.A2 continued

<table>
<thead>
<tr>
<th>Samples</th>
<th>Level</th>
<th>First Difference</th>
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</thead>
<tbody>
<tr>
<td></td>
<td>Intercept</td>
<td>Intercept and Trend</td>
</tr>
<tr>
<td>Western PRC</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Trade share</td>
<td>-1.614, -2.215</td>
<td>-2.694***, -3.014**</td>
</tr>
<tr>
<td>lnExport</td>
<td>-2.106, -2.189</td>
<td>-2.881***, -3.063***</td>
</tr>
<tr>
<td>lnImport</td>
<td>-1.929, -1.731</td>
<td>-2.467**, -2.678</td>
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<tr>
<td>lnFDI</td>
<td>-2.255*, -2.423</td>
<td>-2.893***, -3.139***</td>
</tr>
<tr>
<td>lnDensity</td>
<td>-1.643, -1.970</td>
<td>-2.326**, -2.439</td>
</tr>
<tr>
<td>Manufacturing</td>
<td>-1.954, -2.203</td>
<td>-2.633***, -2.552</td>
</tr>
<tr>
<td>Service</td>
<td>-2.084, -2.322</td>
<td>-2.828***, -2.710</td>
</tr>
</tbody>
</table>

FDI = foreign direct investment, GDP = gross domestic product.

Notes:
1. Pesaran’s (2007) CIPS test is applied;
2. ***, **, and * denote significance levels at 1%, 5%, and 10%, respectively.

Source: Authors.
Table 9.A3: Pedroni’s Panel Cointegration Test by Region

<table>
<thead>
<tr>
<th>Pollutant: Industrial Wastewater</th>
<th>Pollutant: SO₂</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>lnTrade</strong></td>
<td><strong>InTrade</strong></td>
</tr>
<tr>
<td><strong>Openness</strong></td>
<td><strong>lnTrade</strong></td>
</tr>
<tr>
<td><strong>Eastern PRC</strong></td>
<td></td>
</tr>
<tr>
<td><strong>Central PRC</strong></td>
<td></td>
</tr>
<tr>
<td><strong>Western PRC</strong></td>
<td></td>
</tr>
</tbody>
</table>

ADF = Augmented Dickey-Fuller, SO₂ = sulfur dioxide.

Notes:
1. The lag order is determined by Schwarz information criterion.
2. *** denotes a significance level of 1%.
3. Results are not available for the specification with export and import.

Source: Authors.
10
Globalization and the Environment in India

Sugata Marjit and Eden Yu

10.1 Introduction

India embarked on a path of liberal economic reform in the 1990s after years of nurturing an intensively regulated and controlled economic environment that was loosened a bit in the mid-1980s. Now it is well recognized that such a sea change in policy has led to impressive achievements in many sectors of the economy. The most important and critical segment of this reform has been trade and foreign investment, including deregulations in the well-known industrial licensing system. The theme of this chapter is to make readers aware of the relevant work on climate change and the impact of trade and environment on climate change with special reference to India's economy. The impact of globalization on India has been felt in terms of increasing prosperity, partly triggered by increasing volumes of trade, investment, and growth.

A cursory look at the evidence suggests that the conventional openness index, represented by the ratio of volume of trade to gross domestic product (GDP), increased substantially from a little over 10% in the 1990s to almost 50% in the first decade of the 21st century. Average tariff rates came down drastically, leading to more imports and exports. Foreign direct investment (FDI) flows also started recording impressive levels over the next 2 decades. Trade and FDI in the Indian context are depicted in Figures 10.1 and 10.2. Per capita energy consumption and carbon dioxide (CO\textsubscript{2}) emissions have also increased in the post-reform period (Figures 10.3 and 10.4). Figure 10.5 depicts India's GDP growth. However, India's per capita consumption of energy and emissions are both way below that of the People's Republic of China (PRC) and the United States (US) (Figures 10.6 and 10.7).
Figure 10.1: Foreign Direct Investment, Trade, and Carbon Dioxide Emissions in India

CO₂ = carbon dioxide, GDP = gross domestic product.

Figure 10.2: India’s Trade and Trade Openness

(a) Trade Openness After Liberalization


continued on next page
Figure 10.2 continued

(b) Imports and Exports of Goods and Services

GDP = gross domestic product.

Figure 10.3: Energy Utilization in India

kg = kilogram.
Figure 10.4: Carbon Dioxide Emissions in India after Liberalization

\[ \text{CO}_2 \text{ emissions - India (metric tons per capita)} \]

CO\(_2\) = carbon dioxide.

Figure 10.5: Gross Domestic Product Growth of India’s Economy, 1990–2016

The effects of trade and FDI, though mixed and different, are felt on GDP growth. India has jumped ahead of nations, breaking the taboo of the so-called historic “Hindu” growth rate of 3% per annum, averaging around 8% per annum (India’s GDP growth rate in the post-reform period is shown in Figure 10.5).
In spite of the major financial crises and crashes of 1997 and 2008, India has remained a close second to the PRC and is currently growing at a commendable rate. After the historic demonetization episode in November 2016 when about 86% of currency was withdrawn from circulation to control black market transactions and illegal liquidity, the growth rate faltered a little. The recent work on global fiscal policy and inequality by the International Monetary Fund (Clements et al. 2015) confirms the claim that millions of people have been lifted above the poverty line, thanks to the historic switch to a regime of liberal economic policies. Interestingly, while the degree of inequality in the 2000s shows a moderately increased level relative to the 1980s, the increment is far lower than in the PRC and the increase itself is definitely on the lower side, if compared with the world average during the relevant period.

Against this backdrop, it is hard to be too concerned about the direct environmental consequences of more open trade and investment regimes. It is also extremely difficult to isolate effects that are exclusively due to liberal trade policies and quite independent of the growth effect. The real need is to analyze the problems at various levels by focusing on the effects of a significant change in the growth regime, reflecting the increasing level of prosperity, which has definitely been impacted by liberal trade policies.

The pattern of trade should itself have some effect on environmental elements such as fossil fuels, renewable energy, carbon emissions, etc. Further to this effect, one should worry about overall climate change, food supply, and food security. It is also important to understand the pattern of India's trade and investment that has characterized its growth path over the last few decades. On the one hand, trade and investment policies may directly regulate environmental damage and affect optimal utilization of natural resources. On the other hand, any kind of regulatory policy or its emergence will be guided by several critical factors involving awareness, political lobbies, strategic reactions, and the massive size of the informal sector. Coupled with this, India's participation in global policy making to control transboundary pollution, climate change, and emission-related factors will also be important. We reflect on all these as much as possible, given the limited length of the chapter. The main point of this chapter is to argue that, by global standards, India's performance has not been particularly worrisome. Liberal trade policies and market integration have contributed to growth, resulting in pressure on the use of natural resources. However, poor regulatory control has created sporadic natural and national disasters in tandem with factors affected by global warming. It is this problem of implementing regulations that requires special attention.

Certain global issues involve transboundary concerns and India cannot be insulated from those concerns. Globalization and the environment in
India are dependent partly on global climatic conditions and the policies of other countries, as one cannot ignore global negative externalities. These will be related to the policies with which the Government of India has been engaged. In this introductory section, we highlight research on climate change at the global level and emerging problems that require attention. We reflect on some of these issues in the policy section.

Section 2 of this chapter discusses the theoretical implications, drawing from various contributions in the sphere of environmental regulations. Section 3 provides a panoramic view of the available academic literature on the impact of trade on the environment in general and on India specifically mainly from an empirical perspective. Section 4 highlights the global research on climate change and the Indian counterpart. Section 5 briefly addresses the policy strategies of the Indian government. The concluding section highlights issues that require further attention and link India’s concerns with the greater Asian perspective, before it provides some concluding remarks.

10.2 Theoretical Perspective

Trade and FDI-impacted prosperity may have severe, but sporadic, environmental consequences, along with silent erosion of ecological surroundings. However, a look at the destination of trade and FDI in various sectors shows a relatively benign picture (Kar and Majumdar 2016). It is very difficult to suggest that trade and FDI were singularly important in terms of environmental damage.

The expansion of tourism, an appetite for real estate investments, and prosperity-led demand for the construction sector can lead to excessive use of land-based resources, giving rise to catastrophic events. Agriculture, with increasing use of chemical fertilizers and pesticides, has long affected soil conditions. However, it is very difficult to identify and isolate the role of trade in such a malady. India has suffered, and will continue to suffer, since the regulatory framework may not function properly for institutional reasons. In general, if we give up industries that generate pollution during production and import the underlying products as trade opens up, we reduce the extent of environmental damage. On the other hand, if our export good has significant pollution content, the result is exactly reversed; given that India’s trade pattern is not manufacturing intensive, it may not have suffered from this aspect of the problem. However, the pressure of growth and prosperity on the informal sector, weak and corrupt institutional structure, and social neglect may have severely impacted the ability to regulate or control environmental damage. Theoretical and empirical research on the problem of such regulatory control in the context of the informal sector...
has been amply demonstrated in Biswas, Farzanegan, and Thum (2012) and Biswas and Thum (2017). The problem of regulation with a huge informal sector has been discussed in detail in Marjit, Ghosh, and Biswas (2007), Marjit and Kar (2011, 2012), etc. The basic idea is as follows.

Consider $Y(T)$ as the level of GDP. It is favorably affected by the volume of trade $T$, with

$$Y'(T) > 0 \quad (1)$$

However, with environmental damage, the social value is $eY(T)$ with $0 < e < 1$, as $(1 - e)$ fraction of $Y(T)$ is lost in the process as a cost. If one could do effective green accounting, the true national income would be $eY(T)$. The regulator can regulate by forcing the producers to invest in abatement technology $C(e)$

$$C = C(e), C' > 0, C'' > 0 \quad (2)$$

A better—i.e., higher—environment requires a higher cost of abatement. The socially optimal abatement level is determined by maximizing:

$$V(e) = eY(T) - C(e) \quad (3)$$

F.O.C. $V'(e) = 0$, [with $V''(e) < 0$ and $C'' > 0$]

$$C'(e) = Y(T) \quad (4)$$

Thus, the optimal level can be shown by

$$e^* = f[Y(T)], f' > 0 \quad (5)$$

Note that as $T$ and $Y$ increase, the social marginal benefit from abatement increases. With every increase in $e$, the savings increase; therefore, optimal environmental quality actually increases.

Producers do not internalize. Hence, to them, the cost is $C(e^*)$ and profit is given by:

$$\pi(e^*) = Y(T) - C(e^*)$$

$$= Y(T) - C[e^*(T)] \quad (6)$$

Producers may try to bribe the agent of the regulator. Also, enforcing $e'$ for the informal segment can be very difficult because of unrecorded and unregistered activities. Producers will try to maximize the following:
\[ \pi(\bar{e}) - \pi(e^*) = [C(e^*(T)) - C(\bar{e})] - B(e^* - \bar{e}) \]  

(7)

where \( \bar{e} \) is the maximum abatement that will be engaged in after paying a bribe determined by the bribe function \( B \). In the simplest application of a Nash bargaining problem, the bribe amount will be:

\[ B = \frac{1}{2} [C(e^*) - C(\bar{e})] \]  

(8)

Typically, if there is no other cost, then \( \bar{e} = 0 \), but there may be other costs depending on further monitoring, auditing, etc. This may happen when eventually, due to a national level disaster or calamity and media attention, punishment cannot be avoided. In that case, let that cost be \( Z(e^* - \bar{e}) \), with \( Z' > 0, Z'' > 0 \). The maximum abatement level is then finally guided by:

\[ \max_{[\bar{e}]} \frac{1}{2} [C(e^*) - C(\bar{e})] - qZ(e^* - \bar{e}) \]  

(9)

\[ \Rightarrow qZ'(e^* - \bar{e}) = \frac{1}{2} C'(\bar{e}) \]  

(10)

where \( q \) is the probability that the evader is further audited after paying the bribe. For very low \( q \) or very low \( Z' \), \( e \) will be close to zero. With rising \( T \), \( e^* \) will increase and that will increase \( \bar{e} \). One must appreciate the fact that an environmental disorder is a natural consequence of prosperity, however big or small, and for a country like India, implementing regulatory policy is a huge task due to corruption in the governance process.

One point that has been avoided in the above framework is a direct impact of \( e \) on \( T \). It is possible that the process of abatement involves a cutback in the production of the export good or certain specific imports, adversely affecting \( T \). In general, such a problem has a serious political impact and thus tends to restrict regulatory control. The problem then looks like:

\[ V(e) = eY(T(e)) - C(e) \]  

(11)

\[ V'(e) = 0 \Rightarrow Y(T(e)) + eY'T'(e) = C'(e) \]  

(12)

with \( T'(e) < 0 \). The left-hand side in (12) is less than the right-hand side in (4) as the social marginal benefit for a higher \( e \) is lower, pushing \( e^* \) down.
Another interesting feature of environmental regulation concerns its impact on employment and wages. This constitutes a major problem for the government as a majority of the workforce is engaged in the informal sector, with no exception across the entire developing world. As has been elaborated in Marjit and Kar (2011), this is the stark reality for India. Critical policies have been discussed in the context of the labor market (Marjit 2003; Marjit et al. 2007; Marjit and Kar 2012; Acharyya and Kar 2014; etc.), focusing on interaction between the formal and the informal sectors. However, such a general equilibrium exercise is rare in the context of environmental regulations.

Typically, the political concern seems to be that environmental regulations tend to depress employment and wages and so are difficult to implement in democracies. However, movement of capital between the formal and the informal sectors, coupled with the inability of the government to push through regulations in the informal sector, may actually prove to be helpful to informal workers through wage increases. We provide a theoretical example drawn primarily from Marjit (2003) and Marjit and Kar (2011).

Figure 10.8 describes a situation where \( L_F = O_1A_1 \) is hired in the formal sector with a fixed unionized wage \( \bar{W} \). The rest, \( A_1O_2 = L_t \), are hired in the informal sector at wage \( W \), where \( \bar{W} > W \). (\( D_F, D_I \)) are demand for labor in the formal and informal sectors, respectively. Regulatory tax will shift \( D_F \) to the left, reducing \( L_F \) to \( O_1A_2 \) and informal wage to \( W_1 < W \), the standard contractionary effect. However, if we allow capital (hidden in the level and the slope of \( D_F \)) to move from formal to informal, shifting \( D_F \) down further but shifting \( D_I \) up, \( L_F \) still falls to \( O_1A_3 \) but \( W \) rises to \( W_2 > W > W_1 \). Environmental regulation in this case has actually helped the informal workers. The intuition is that lack of regulation leads to excessive allocation of capital in the formal sector. Also, an upward shift of \( D_I \) can occur even when regulations affect that sector. It is the relative shift that will determine the result.

This has serious political economic implications. If there is a decline in \( W \), votebank politics will not allow such a regulatory move. The above example is based on the assumption that the informal sector produces a final good: the result will change when the informal sector produces an intermediate good for the formal sector. A pollution tax on the formal sector will affect the informal sector directly, even if the government cannot directly enforce an environmental stand on the informal sector. As long as the price of the intermediate good is pegged by trade by a standard small economy assumption, the result will not vary much, as the informal sector can find other buyers in the international market. Return to capital will fall and \( w \) will rise. If the price of the intermediate good is endogenously determined and/or there is a direct environmental
tax on the informal sector, $w$ may fall. The environmental regulations imposed on the formal and informal sectors may exhibit complex policy reactions.

Let us move from production-related environmental concern to the problem of regulation in the domain of public goods. Economic growth fueled by trade or FDI-related prosperity often leads to excessive use of natural resources. Examples in India are abundant where the expanding real estate and construction business often encroaches upon water bodies on the one hand and lead to excessive local demand for water on the other, pushing the water level further down. Lack of cooperative effort on the part of citizen users for the renovation of water bodies, cleaning, dredging, etc. leads to suboptimal public investment in critical areas of consumption, as lobbying is inadequate. A few years ago, in Northern India, a massive landslide caused by torrential rain feeding a mountain river led to a huge loss of prosperity and human lives (Kala 2014; Singh et al. 2016). Everyone could see the problem of construction of cheap hotels across riverbanks that crippled the soil base and made it vulnerable to natural disaster. Excessive use of groundwater in India has led to
serious arsenic-related diseases. This has been documented time and again (Das et al. 1994). These point to an inevitable fallout of excessive demand on natural resources, a hallmark of a fast-growing economy. It is well known that energy-intensive consumption activities during winter perennially affect the PRC and its overuse of coal-based resources. The National Development and Reform Commission of the PRC said in its annual report that it would implement policies aimed at reducing coal consumption and controlling the number of energy-intensive projects in polluted regions (Aizhu et al. 2015).

The problem arises at three levels.

First, the regulation priorities of the government can be lopsided, being dependent on the electoral policies and economic performance of the political regime. For example, if trade and FDI stimulate growth and growth relates to ecological problems at the local, regional, or national level, governments may not stir unless natural and national disasters occur through floods, droughts, landslides, etc.

Second, people may not consider environmental degradation as a quality of life issue in a poor country. While growth and affluence are slowly making an impact, public consciousness in this regard may turn out to be too shallow and non-existent. The recent policy drive by the Indian government, known as the Clear India (Swatch Bharat) mission, is an endeavor that has created millions of toilets. This speaks loudly of a chronic problem of defecation-induced health hazards and a lack of awareness among people. International trade, FDI, and other growth-augmenting avenues have very little to deliver in that respect, as prosperity and social awareness may not go hand in hand.

Third, a lack of cooperation in public good-related initiatives among affluent citizens to resolve local problems of excessive exploitation of natural resources makes the problem even more complex.

The latter problem can be related to a simple theoretical structure. Cooperation may not be sustained with increasing prosperity as the rich are less likely to cooperate than the poor.

Consider a community of $n$ persons with expenditures $x_i, i = 1, 2, \ldots, n$. While higher $x_i$ delivers higher utility, $\sum_{i}^{n} x_i$ may affect a natural resource commonly used by all but without any property right assigned. Symmetry assumption implies $x_i = x \forall i$.

Let $U_0 = ax_0 - \beta(nx_0), nx_0$ (with $x_0$ as initial level of expenditure) \hspace{1cm} (13)$

We assume $U_0 > 0$, with $U_0(0) = 0, \beta' > 0, \alpha > 0 \cdot \beta$ represents cost to the environment, for example, a decline in the water level due to excessive use.
With enough curvature restrictions on $\beta(\cdot)$, we can show that with excessive expenditure $U_0$ goes down. However, people usually do not care about the negative component or $\beta$. If a cooperative arrangement could be enforced, we could decide on some $\tilde{x}$ that is optimal and some corresponding $\tilde{U}$. This can be derived by treating $U_0$ as a social welfare function. However, people can cheat within a cooperative agreement and if one of them does it, cooperation breaks down. The cheating payoff is $\tilde{U} > U > U_0$ since the deviant will presume that, given that all others stick to $\tilde{x}$, he or she can increase his or her own $x$. Once cooperation breaks down, they get $U_0$, which is the punishment payoff. Therefore, such an agreement will break down if:

\[
\frac{\bar{u}}{1-\delta} < \bar{U} + \frac{\delta U_0}{1-\delta}
\]

Or, \[
\bar{U} < (1-\delta)\bar{U} + \delta U_0
\]

where $\delta$ is the rate of discount $0 < \delta < 1$. Since $\bar{U} > U_0$, higher $\delta$ will reduce the right-hand side in (15). Therefore, the higher $\delta$ is, the better the chance of cooperation is. As greater prosperity sets in, if $U$ is not renegotiated appropriately, perceived payoff from cheating $\bar{U}$ is likely to increase more than $U_0$ even if $d\bar{x} = dx_0$, thwarting cooperation. Hence, cooperation may be more difficult to sustain under growth and prosperity. The result will change if people become more environment-conscious, with appropriate changes in the $\beta(\cdot)$ function, or if one could design property rights in this context through proper pricing of the natural damage. Related issues have been discussed rigorously by Chander and Tulkens (2006), Chander and Muthukrishnan (2015), and Quah (2015).

### 10.2.1 Overall Research on Trade and Environment

Several papers have discussed the environmental implications of international trade or a more open trade regime. A representative introductory sample is Copeland and Taylor (2004; 2013) and Chao and Yu (2004). While these works have provided rich insights toward an understanding of the problem, there have been other recent papers. Later, we will try to focus on the aspects that are particularly relevant for India. We now briefly summarize some works that have examined the relationship between trade and the environment from a general perspective. This is very selective, but the papers themselves have a plethora of references to draw from that readers can use. Later, we reflect on the Indian scenario.
Neumayer (2000) critically assesses three ways by which trade might harm the environment. First, trade liberalization might exacerbate existing levels of resource depletion and environmental pollution. Second, open borders might allow companies to migrate to “pollution havens,” thus undermining high environmental standards in host countries. Third, the dispute settlement system of the World Trade Organization (WTO) might favor trade over environmental interests in case of conflict. It is shown that while trade liberalization can lead to an increase in environmental degradation, pollution havens are not a statistically significant phenomenon.

Copeland and Taylor (2001) draw quite heavily from trade theory but develop a simple pollution demand and supply system featuring marginal abatement cost and marginal damage schedules familiar to environmental economists. They use a simple model to facilitate extensions examining the environmental consequences of growth, the impact of trade liberalization, and strategic interaction between countries. One could also refer to Antweiler et al. (2001) in this context.

Chen and Woodland (2013) analyze noncooperative environmental policies and investigate whether trade undermines the effectiveness of unilateral environmental policies, in which carbon leakage and international competitiveness are particularly important. They review the interactions between trade and environmental policies, border tax adjustment policies, and the role of the WTO in combating climate change arising from economic activities.

Dellink et al. (2017), by building on the analysis in the 2015 report of the Organisation for Economic Co-operation and Development, *The Economic Consequences of Climate Change*, present a plausible scenario of future socioeconomic developments and climate damage to shed light on the mechanisms at work in explaining how climate change will affect trade.

Sauvage and Timiliotis (2017) discuss international trade in environment-related services. By lowering the costs of these services and improving access to suppliers, trade policy can contribute, alongside energy and environmental policy, to the prevention and abatement of greenhouse gas (GHG) emissions and pollution in all its forms.

Readers can also consult Beghin et al. (1994), Sturm (2003), and Dupuy (2012) for additional references on this area.

### 10.2.2 Research on Trade and Environment for India

India’s trade is typically dominated by the services sector, agriculture, oil, industrial components, gems and jewelry, etc. The fact that wage inequality has been on the rise in the country for quite some time indicates an ever-growing premium for human capital, particularly in
the information technology–led sectors (Marjit and Acharyya 2003; 2009). India’s weak link seems to be manufacturing. While the service sector commands about 60% of GDP, manufacturing contributes only around 25%. Historically, over the last few decades since reform, India’s tradition is exactly the opposite to the PRC’s if we look at the composition of GDP. A simple exercise to trace the impact of trade, FDI, and GDP on emissions has been attempted in terms of econometric analysis.

Scholarly work on trade, FDI, and the environment in India that offers rich theoretical insights and solid empirical evidence is scarce, although there is a good body of work on general environmental issues, some of which we summarize in the next section, along with the general literature on climate change from science, technology, and economic perspectives. We first consider an empirical exercise on the relationship between overall emissions, growth, trade, and FDI in India, specifically to show that it is more GDP-led prosperity than trade and FDI that have led to a growth in emissions.

To understand the effect of liberalization on the extent of carbon emissions in India in a very simple rudimentary framework, we regress emissions on three variables: GDP per capita, trade per capita, and FDI per capita. However, our explanatory variables might not have an immediate effect on the level of emissions; therefore, lagged explanatory variables might be appropriate. Here we have taken a two-period lag. Our equation is as follows.

\[
CO_2 Emission_t = \alpha + \beta_1 GDP \text{ Per Capita}_{t-2} + \beta_2 Trade \text{ Per Capita}_{t-2} + \beta_3 FDI \text{ Per Capita}_{t-2}. \tag{16}
\]

Here \( t \) denotes time point. We run the regression for the period 1978 to 2013.

\begin{table}[h]
\centering
\caption{Regression Results}
\begin{tabular}{|l|l|}
\hline
\textbf{Explained Variable: LOG [CO}_2\text{ emissions (tons per capita)]} & \textbf{Estimated Coefficient} \\
\hline
LOG [GDP per capita (–2)] & 1.44* \\
LOG [TRADE per capita (–2)] & –0.33* \\
LOG [FDI per capita (–2)] & 0.04* \\
Constant & –5.40 \\
R-squared & 0.97 \\
\hline
\end{tabular}
\end{table}

\( CO_2 = \text{carbon dioxide, FDI = foreign direct investment, GDP = gross domestic product.} \)

\( \text{Note:}^* \text{ denotes significance at the 1% level;}^{**} \text{ denotes significance at the 5% level.} \)

\( \text{Source: Author’s own calculation based on World Bank Data.} \)
As we see from the result, GDP per capita has a direct and statistically significant impact on the extent of emissions in India. This implies that the Indian economy has grown at the cost of environmental degradation. The relationship between trade and emissions seems to be inversely proportional, implying that, holding other factors constant, emissions decrease with increased trade. As an educated guess, we may propose that Indian imports are mostly manufactured items, which may involve a polluting production process, and are being produced outside India. On the other hand, due to the import of these manufacturing items, the polluting import substitution manufacturing is being closed down, possibly resulting in lower pollution in India due to imports. Furthermore, Indian exports, though they have manufacturing content, also comprise growing information technology service-oriented activities, which are almost without pollution. Hence, trade helps India reduce pollution, as reflected in the negative relationship in our estimated equation. Like GDP, FDI is found to influence emissions in India positively, though insignificantly. The FDI result echoes the work of Acharyya (2009), which we discuss later.

Now, we all know that India’s economy was opened to the rest of world in 1991 through liberalization. This was expected to have had

**Figure 10.9: Carbon Dioxide Consumption in India**

![Graph showing carbon dioxide consumption in India from 1980 to 2020.](data:image/png;base64,iVBORw0KGgoAAAANSUhEUgAAAoAAAAHcCAYAAAAvQHbAAAACXBIWXMAAAsTAAALEwEAmpwYAAAAB1RU5ErkJggg==)

\(\text{CO}_2 = \text{carbon dioxide.}\\
\text{Source: World Bank Data (www.data.worldbank.org).}\)
some impact on pollution. After liberalization, GDP, trade, and FDI all rose in India's economy. We need to ascertain whether pollution in India increased due to liberalization. In Figure 10.9, we plot the level of emissions across the years and try to see if any structural shift occurred.

It seems from the diagram that there is no structural shift but there is a change in slope after liberalization. To capture this aspect, we use an interaction dummy with the break point being 1992. So, our equation becomes:

\[
CO_2 \text{ Emission}_t = \alpha + \beta_1 \text{ GDP Per Capita}_t - 2 + \\
\beta_2 \text{ Trade Per Capita}_t - 2 + \\
\beta_3 \text{ FDI Per Capita}_t - 2 + \\
(d_i \times \text{ GDP Per Capita}_t - 2) + \varepsilon_t
\]

(17)

where d stands for dummy = 0 if year < 1992, 1 if year ≥ 1992.

Table 10.2: Regression Results

<table>
<thead>
<tr>
<th>Explained Variable: LOG [CO₂ emissions (tons per capita)]</th>
<th>Explanatory Variables</th>
<th>Estimated Coefficient</th>
</tr>
</thead>
<tbody>
<tr>
<td>LOG [GDP per capita (-2)]</td>
<td>1.51*</td>
<td></td>
</tr>
<tr>
<td>LOG [Dummy GDP per capita (-2)]</td>
<td>0.10*</td>
<td></td>
</tr>
<tr>
<td>LOG [TRADE per capita (-2)]</td>
<td>-0.37*</td>
<td></td>
</tr>
<tr>
<td>LOG [FDI per capita (-2)]</td>
<td>0.01</td>
<td></td>
</tr>
<tr>
<td>Constant</td>
<td>-5.83*</td>
<td></td>
</tr>
<tr>
<td>R-squared</td>
<td>0.97</td>
<td></td>
</tr>
</tbody>
</table>

CO₂ = carbon dioxide, FDI = foreign direct investment, GDP = gross domestic product.
Note: * denotes significance at the 1% level; ** denotes significance at the 5% level.
Source: Author’s own calculation based on World Bank Data.

Holding other factors constant, our estimated equation (equation 17) seems to show a positive change of slope. This means that the rate of increment of emissions in India has increased due to the rise in GDP after liberalization. Hence, we can infer that the liberalization-backed GDP growth has had an adverse environmental impact on India’s economy.

If we wish to see whether there is any shift due to trade after liberalization, we modify our equation as follows:
CO₂ Emissionₜ = α + β₁ GDP Per Capitaₜ₋₂ +
β₂ Trade Per Capitaₜ₋₂ +
β₃ FDI Per Capitaₜ₋₂ +
(dᵢ × Trade Per Capitaₜ₋₂) + εₜ  \quad (18)

where d stands for dummy = 0 if year < 1992, 1 if year ≥ 1992.

Table 10.3: Regression Results

<table>
<thead>
<tr>
<th>Explained Variable: LOG [CO₂ emissions (tons per capita)]</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Explanatory Variables</strong></td>
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</tr>
<tr>
<td>Constant</td>
</tr>
<tr>
<td>R-squared</td>
</tr>
</tbody>
</table>

CO₂ = carbon dioxide, FDI = foreign direct investment, GDP = gross domestic product.
Note: * denotes significance at the 1% level; ** denotes significance at the 5% level.
Source: Author’s own calculation based on World Bank Data.

Keeping other things constant, trade seems to impact emissions negatively in India. However, there is no statistically significant shift in the slope or rate of increase/decrease of pollution due to trade after the liberalization policy came into force in India.

Acharyya (2009) provides an early analysis of the impact of FDI on the environment, which demonstrated that FDI inflow in the 1990s had a large positive impact on emissions through output growth, and FDI had a positive but marginal impact on growth. The two together imply that growth must have a very high emission elasticity. This, in a way, may contradict the observations of Wei and Smarzynska (1999) and Hassaballa (2013) regarding the developing world and pollution havens, as reported in Kar and Majumdar (2016).

Papers by Kar and Majumdar (2016) have seriously attempted to reflect on trade and technology policies with reference to the developing world and with a special emphasis on India. Before we discuss these papers, it is important to note that the better environmental standards are usually dependent on internal policies rather than on trade and...
FDI-related policies. However, the impact of market integration on the environment may require trade-specific policies. Coupled with such policies, one needs to focus seriously on abatement technology. Continuous upgrading of technology through diffusion and interlinkage across the value chain can have a profound, albeit invisible, impact on emission standards.

Kar and Majumdar (2016) show that for a group of low-middle-income countries sorted on the basis of manufacturing to trade ratios, of which India is a prominent member, a rise in most-favored-nation tariff rates reduces CO$_2$ emissions, and that effect is further reinforced if FDI flows to nonpolluting sectors such as agriculture. A rise in most-favored-nation tariff rates takes care of the trade diversion problem. The paper rigorously demonstrates that further protection may actually help lower emission standards. For agriculture, substitution of imports by FDI has a better effect on emissions.

Majumdar and Kar (2017) study the emission intensity of 15 organized manufacturing industries and agricultural sectors in India and find the relationship between technology adoption and emission intensity at the industry level from 1996 to 2009. They show that when better technologies are adopted to produce export goods as opposed to non-traded goods, emissions fall in a significant way. Typically, international trade and FDI facilitate the adoption of technology. The direct effect of globalization might, therefore, have helped in this regard.

10.3 Global Climate Change: Enduring and Contemporary Policy Issues

Climate change is a complex problem which, though environmental in nature, has consequences for all spheres of existence on our planet. It either impacts, or is impacted by, global issues, including poverty, economic development, population growth, sustainable development, and resource management. Climate change is a global challenge and requires a global solution. GHG emissions have the same impact on the atmosphere, whether they originate in Washington, London, or Beijing. Consequently, action by one country to reduce emissions will do little to reduce global warming unless other countries act as well. Ultimately, an effective strategy will require commitment and action from all the major emitting countries. Climate change poses the serious challenge of CO$_2$ emissions reduction. Emission control by developing countries is becoming key to mitigate climate change effectively, as those countries now account for more than half of global emissions and are still expanding their energy infrastructure.
At the very heart of the response to climate change, however, lies the need to reduce emissions. In 2010, governments agreed that emissions must be reduced so that increases in global temperature are limited to less than 2°C.

In 1992, countries joined an international treaty, the United Nations Framework Convention on Climate Change, to consider what they could do to limit increases in global temperature and the resulting climate change, and to cope with its impacts. By 1995, countries realized that convention provisions on emission reduction were inadequate. As a result, they launched negotiations to strengthen the global response to climate change and, in 1997, adopted the Kyoto Protocol.

In short, the Kyoto Protocol is what “operationalizes” the convention. It commits industrialized countries to stabilizing GHG emissions based on the principles of the convention. The convention itself only encourages countries to do so. The protocol sets binding emission reduction targets for 37 industrialized countries and the European community in its first commitment period. Overall, these targets add up to an average 5% emissions reduction compared to 1990 levels over the period 2008–2012 (the first commitment period). The protocol is structured on the principles of the convention. It only binds developed countries because it recognizes that they are largely responsible for the current high levels of GHG emissions in the atmosphere, which are the result of more than 150 years of industrial activity. The Kyoto Protocol places a heavier burden on developed nations under its central principle, that of common but differentiated responsibility. The second commitment period began on 1 January 2013 and will end in 2020.

10.3.1 Selective Global Research on Climate Change

A change in climatic conditions has a diversified impact on an economy. Good climate helps a country grow by way of good production and thereby helps eradicate inequality and poverty. To begin with, we consider the paper by Blicharska et al. (2017), which gathers scientists from around the world and deals with research on climate change across the globe in recent years. The authors look carefully at the global North–South divide in research on climate change and its negative consequences. They postulate that the northern domination of science in relation to climate change policy and practice, and the limited research led by researchers in southern countries may hinder the further development and implementation of global climate change agreements and nationally appropriate actions. The authors illustrate the extent of the divide, review underlying issues, and analyze the consequences for climate change policy development and implementation. The paper proposes a
set of practical steps that a wide range of actors in both northern and southern countries should take at global, regional, and national levels to span the North–South divide.

A recent article by Burke, Hsiang, and Miguel (2015) analyzed the relationship between historic temperature fluctuations and macroeconomic growth. Their findings can be summarized as follows. First, in contrast to past studies, they argue that 21st century warming could lead to huge global-scale macroeconomic impacts. The best estimate from Burke and colleagues is that business-as-usual emissions throughout the 21st century will decrease per capita GDP by 23% below what it would otherwise be, with the possibility of a much larger impact. Second, they conclude that both the size and the direction of the temperature effect depend on the starting temperature: Countries with an average yearly temperature greater than 13°C (55°F) will see decreased economic growth as temperatures rise. For cooler countries, warming will be an economic boon. This nonlinear response creates a massive redistribution of future growth, away from hot regions and toward cool regions. Based on the analysis, rich and poor countries respond similarly at any temperature but the impact of warming is nonetheless much greater on poor countries because they are mostly in regions that are already warm.

Hsiang et al. (2017) developed a flexible architecture for computing damage that integrates climate science, econometric analyses, and process models. The authors used this approach to construct spatially explicit, probabilistic, and empirically derived estimates of economic damage in the US from climate change. The combined value of market and non-market damage across analyzed sectors—agriculture, crime, coastal storms, energy, human mortality, and labor—increases quadratically with global mean temperature, costing on average roughly 1.2% of GDP per +1°C. Importantly, risk is distributed unequally across locations, generating a large transfer of value northward and westward that increases economic inequality. By the late 21st century, the poorest one-third of countries are projected to experience damage of between 2% and 20% of income (90% chance) under business-as-usual emissions.

Alagidede, Adu, and Frimpong (2016) contribute to the empirics of climate change and its effect on sustainable economic growth in sub-Saharan Africa. Using data on two climate variables, temperature and precipitation, and employing panel cointegration techniques, the authors estimate the short- and long-run effects of climate change on growth. The paper finds that an increase in temperature significantly reduces economic performance in sub-Saharan Africa. Furthermore, it shows that the relationship between real GDP per capita and climate factors is intrinsically nonlinear.
Dell, Jones, and Olken (2008) examine the impact of climatic change on economic activity throughout the world. The authors find three primary results: (i) higher temperatures substantially reduce economic growth in poor countries but have little effect in rich countries; (ii) higher temperatures appear to reduce growth rates in poor countries rather than just the level of output; and (iii) higher temperatures have wide-ranging effects in poor nations—reducing agricultural and industrial outputs and aggregate investment, and increasing political instability. Analysis of decade-long or longer climate shifts also shows substantial negative effects on growth in poor countries.

Costinot, Donaldson, and Smith (2016) seek to quantify the macro-level consequences of some micro-level shocks. Using an extremely rich micro-level data set that contains information about the productivity—both before and after climate change—of 10 crops for 1.7 million fields covering the surface of the earth, the authors find that the impact of climate change on these agricultural markets amounts to a 0.26% reduction in global GDP when trade and production patterns are allowed to adjust. Since the value of output in our 10 crops is equal to 1.8% of world GDP, this corresponds to about one-sixth of the total crop value.

Zhang, Zhang, and Chen (2017) explore the importance of some additional climatic variables other than temperature and precipitation. Using county-level agricultural data from 1980 to 2010 in the PRC, we find that those additional climatic variables, especially humidity and wind speed, are critical for crop growth. Therefore, omitting those variables will likely bias the predicted impacts of climate change on crop yields. In particular, omitting humidity tends to overpredict the cost of climate change on crop yields, while ignoring wind speed is likely to underpredict the effect. The paper finds that climate change will likely decrease the yields of rice, wheat, and corn in the PRC by 36.25%, 18.26%, and 45.10%, respectively, by the end of this century.

Zewdie (2014) reviews literature on the impacts of climate change and food security specifically in sub-Saharan Africa to characterize and synthesize our current understanding of the problem and identify priorities for future research.

In sub-Saharan African countries, fast GDP growth has created a great opportunity to improve developmental indicators, including food security, but has shown only limited improvements. There is scientific consensus on climate change and it is expected to have a substantial impact on food security. Therefore, new advocacy and a public health movement are recommended to reduce the effect of climate change on food security and malnutrition. Zewdie seeks to assess the impacts of climate change on food security in sub-Saharan Africa.
Documents related to the impacts of climate change on food security are reviewed. The literature indicates that climate components like temperature, precipitation, concentration, and extreme climate events affect food security components. Sub-Saharan Africa, where most of the population is dependent on climate-sensitive economic activities, is one of the most severely affected regions in terms of climate change. The most direct effect and well-researched component of climate change in relation to food security is food availability, through reduced net crop production. It is also found that climate change impacts food accessibility and utilization, but this is not well studied due to its complexity. Projections indicate that this problem will be more severe in the future than it is currently unless climate change mitigation and adaptation strategies are undertaken.

The review concludes that climatic conditions are changing in sub-Saharan African countries and this is affecting food availability, food accessibility, and utilization. The problem will be severe in the future unless the current adaptation and mitigation efforts improve. Therefore, to reduce the problem, the region should use its potential to counter climate change.

The United Nations System Standing Committee on Nutrition (2010) highlights how climate change further exacerbates the already unacceptable high levels of hunger and undernutrition, and proposes policy directions to address the nutrition impact of climate change for consideration by the 16th Conference of the Parties (COP) to the United Nations Framework Convention on Climate Change. The current negotiation process offers opportunities to identify and address some of the actions needed. However, great efforts will be required beyond COP 16, and nutrition should be part of future negotiations. One could also refer to the publication of the World Health Organization (2008), Protecting Health from Climate Change, that identified major health consequences of climate change and devised a research agenda to create strategies to cope these challenges.

10.3.2 Indian Literature on Climate Change

Panda (2009) discusses the consensus on the definition of vulnerability to climate change and the regionally nuanced mapping of the variable impact of climate change. The author opines that despite considerable advances in the methodologies for assessing vulnerability to climate change, ambiguities and uncertainties nevertheless remain. According to the author, vulnerability research faces challenges in three areas. First, climate change is not the only stress that society faces; multiple stressors operate in all human environment systems. It is, therefore, a
challenging task for researchers to identify and evaluate those stressors most relevant for assessing climate change vulnerability. Second, vulnerability assessment requires characterization of the future in terms of socioeconomic and biophysical variables. However, uncertainties about the future make vulnerability assessment much more difficult and challenging. Third, the apparent lack of consistency in the use and meaning of the various concepts employed in vulnerability research contributes to increasing confusion in this area. The author feels that the research in India on vulnerability to climate change is still underdeveloped. Further research is urgently required in several areas. This research must be based on an understanding of the regional and micro-level aspects of climate change to address the vulnerability of people properly, and more accurately.

In a commentary, Kumar (2007) discusses the existing literature on the effect of climate change on Indian agriculture, covering three strands of assessment: impact, vulnerability, and adaptation. The author finds that the economic impact of climate change on agriculture has been studied extensively the world over and it remains a hotly debated research problem. The author discusses papers based on the two approaches to assessing the economic impact—namely, agronomic-economic and Ricardian. Using these approaches, the GDP of India’s economy is expected to decline due to climate change in the latter half of the 21st century.

Kapur, Khosla, and Mehta (2009) summarize extracts from the papers presented at a conference on climate change held in New Delhi in March 2009 focused on the different bargains India might have to strike, both domestically and internationally. The conference was meant to address the options that India would have to exercise to maintain its growth and emerge as a global superpower. The authors summarize the papers presented at the conference and conclude that climate change poses particularly difficult challenges for India. On the one hand, India does not want any constraints on its development prospects; on the other, it wants to be seen as an emerging global power. While the former may be best served by its current position, the latter will require it to take a leadership role on key global issues, climate change being a critical one. It can either approach climate change as a stand-alone global negotiation or weave these negotiations into a “grand bargain” involving linkages with other international negotiations that also involve key Indian interests, whether reforms of the Security Council, WTO negotiations, the financial architecture, etc.

Hanif et al. (2010) tried quantifying the impacts of changes in normal climate parameters for the variable and sustainable development of the agriculture sector nationally and regionally. The study confirms
the premise that climate change impinges considerably on agricultural production and the price of agricultural land.

The authors find that all climate variables except maximum temperature have a highly significant relationship with land prices. Climate change is imposing a cost at the same time that brings the benefit of an increase in land prices during Rabi season due to the rise in maximum temperature. Benefits show farmers’ adaptation to the changing climate, which leads to an increase in long-run net revenues.

The increase in precipitation during the Kharif season tends to increase land value. The increase in precipitation during the Rabi season results in a loss due to decreased production. The increase in mean minimum Rabi temperature, being negatively significant, imposes a cost on the agriculture sector.

The authors conclude that the aggregate global effects on agricultural productivity are expected to be negative by the latter part of this century, and developing countries are expected to suffer sooner and worse. Kumar, Shyamsundar, and Nambi (2010) summarize the discussions held during a 2010 workshop on the economics of climate change adaptation and draw some conclusions for future policy analyses. Given the possibility of moderate or catastrophic climate change in developing countries and the failure of the climate summit in Copenhagen in December 2009 to achieve any consensus on GHG mitigation plans, adaptation as a policy option requires careful attention. This is a report on the said workshop that examined India’s need to adapt to climate change.

The article by Sharma (2012) reviewed literature on the impacts on human health of climate change and land use transition in the Hindu Kush Himalayan region, specifically dealing with topics such as the relationship between climate change and health; health sensitivity, vulnerability, and adaptation; health determinants related to climate change; temperature extremes and health issues; air pollution, black carbon, and health; food security, nutrition, and health; land use change and infectious diseases; and population migration and livelihood transition. The article outlines an agenda for future research on climate change and human health for the Hindu Kush Himalayan region. The author suggests three main agendas: (i) developing methods to quantify the current impacts of climate and weather on a range of health outcomes for people living both in the mountains and downstream; (ii) improving health impact models for projecting the health impacts of climate and land use change under different ecological and socioeconomic conditions; and (iii) evaluating the costs of the projected health impacts of climate change and the effectiveness of adaptation for policy inputs.

The Joint Global Change Research Institute and Battelle Memorial Institute, Pacific Northwest Division (2009) identifies and summarizes the latest peer-reviewed research related to the impact of
climate change on India, drawing on both the literature summarized in the latest assessment reports of the Intergovernmental Panel on Climate Change and on other peer-reviewed research literature and relevant reporting. It includes such impacts as sea-level rise, water availability, agricultural shifts, ecological disruptions and species extinctions, infrastructure at risk from extreme weather events (severity and frequency), and disease patterns. This chapter addresses the extent to which regions in India are vulnerable to the impact of climate change.

Menon et al. (2016) studied fishermen's perceptions of climate change from two coastal districts of Andhra Pradesh in 2011. Fishermen were interviewed to ascertain their perception of climate change over the last 20 years, the impact of the change in climatic parameters on their lives and on marine fisheries, and the adaptation measures required. All respondent fishermen believed that the climate had changed in the last 2 decades. They ranked wind as the parameter that had changed the most in the last 2 decades, while sea status was the most problematic. Avenues for a safe exit from villages and coastal protection structures in case of natural calamities were the highest scoring adaptation measures. Wind was considered the most critical parameter affecting marine fishery, and overfishing was identified as the biggest problem facing fisheries.

Ruchita and Rohit (2017) estimate the impact of climate change on food grain yields—rice and millet—in India. The authors estimate a crop-specific agricultural production function with exogenous climate variables, namely, precipitation and temperature, and control for key inputs such as irrigation, fertilizer, and labor. The analysis is at the district level using a panel data set for physical yield (output per hectare gross cropped area) for the period 1966–1999. The paper finds significant impacts of climate change (temperature and precipitation) on Indian agriculture. For rice, the evidence is overwhelming that both rainfall and temperature matter, but so do other inputs—labor, fertilizer, and irrigation. For millet, rainfall is the sole determinant.

Kumar, Jawale, and Tandon (2008) look at the impacts of climate change on the financial capital of India, Mumbai. These include the impact of temperature rise on rains and floods, and their consequent effects on health. Other consequences, such as a rise in deaths from vector-borne diseases, dislocation due to floods, and sea-level rise, are shown as projected economic losses for the years 2025 and 2050. The economic costs of sea-level rise in terms of loss of property along the coastline are also projected for a 25- and 50-year time scale. The costs arising due to increases in malaria, diarrhea, and leptospirosis outbreaks are projected up to 2050. The conservative estimate of the total cost of all these impacts, including the impact of climate change on tourism, is found to be enormous.
Sinha and Swaminathan (1991) and Kalra et al. (2008) argue that crop production in India is dependent on temperature. Temperature vs. crop production shows a funnel shape for all seasons. For the lower temperature, the properties are almost linearly correlated. In Rabi, production initially shows a negative trend with temperature, which slowly converts into a positive trend. In Kharif, that negative trend is not visible. At higher temperatures, production increases for both the seasons but with large variations. These findings may be helpful in studying the effect of climate change on crop production.

10.4 Policies of the Government of India

10.4.1 Indian Initiatives for Climate Protection

Various programs have been adopted across the globe to protect the environment. One such program is the Global Environment Facility (GEF), established as a pilot program for environmental protection. The current project cycle is GEF-6, covering the years 2014–2018. In 1992 at Rio de Janeiro, the GEF was introduced to help developing countries meet their financing needs to achieve their climate change goals. As of November 2015, the GEF has directly invested $14.5 billion in 3,946 projects in 167 countries, of which $4.2 billion is in 1,010 projects for climate change mitigation. To date, India has received $516.6 million in GEF grants, of which $324.69 million is for climate change mitigation projects while $10 million is for climate change adaptation projects.

In addition, the Clean Development Mechanism (CDM) was adopted as a mitigation instrument under the Kyoto Protocol. At present, the CDM is facing its most severe crisis, having witnessed a steady decline in the number of projects being registered since 2013 owing to the crash in the price of certified emissions reduction (CER) after 2012. As of January 2016, 1,593 out of 7,685 projects registered by the CDM executive board are from India, the second-highest in the world, with the PRC taking the lead with 3,764 registered projects. Indian projects have been issued with 191 million CERs, 13.27% of the total number of CERs issued. These projects are in energy efficiency, fuel switching, industrial processes, and the municipal solid waste, renewable energy, and forestry sectors, and are spread across the country. Around 90%–95% of CDM projects are being developed by the private sector, facilitating investments of about $87.77 billion in the country, which is more than the total of multilateral grants available for climate change-related activities.

Apart from these international measures, the Government of India has also taken some initiatives domestically. The National Action Plan on Climate Change is known to be the major component of India’s
The National Action Plan on Climate Change has proposed a waste-to-energy mission that will incentivize efforts toward harnessing energy from waste and is aimed at lowering India’s dependence on coal, oil, and gas for power production. The National Mission on Coastal Areas will prepare an integrated coastal resource management plan and map vulnerabilities along the entire (nearly 7,000 kilometer-long) shoreline.

The State Action Plan on Climate Change was also introduced to create institutional capacity and implement sectoral activities to address climate change. These plans are focused on adaptation with mitigation as a co-benefit in sectors such as water, agriculture, tourism, forestry, transport, habitat, and energy.

A National Adaptation Fund for Climate Change was established with a budget provision of $13.5 billion for 2015–2016 and 2016–2017. The fund was meant to assist in meeting the cost of national and state-level adaptation measures in areas that are particularly vulnerable to the adverse effects of climate change.

To reduce the consumption of coal, India introduced a carbon tax in the form of a cess on coal. The National Clean Energy Fund is supported by the cess on coal. The fund was created to finance and promote clean energy initiatives, and finance research in the area of clean energy and any other related activities.

The Perform, Achieve, and Trade scheme under the National Mission on Enhanced Energy Efficiency was introduced by the Government of India as an instrument for reducing specific energy consumption in energy-intensive industries with a market-based mechanism that allows trading of the energy-saving certificate. The first Perform, Achieve, and Trade cycle, which ended on 31 March 2015, included 478 industrial units in eight sectors.

India has also started progressing on the renewable energy front. Renewable energy has become a major focus area of the government, with the ambitious target of achieving 40% cumulative electricity capacity from non-fossil fuel-based energy resources by 2030. India is currently undertaking the largest renewable energy capacity expansion program in the world.

The Prime Minister of India launched the International Solar Alliance (ISA) at COP 21 in Paris on 30 November 2015. The ISA will provide a special platform for mutual cooperation among 121 solar resource-rich countries lying fully or partially between the Tropic of Cancer and the Tropic of Capricorn. The ISA secretariat will be hosted by India.

Another major renewable energy policy initiative is the National Offshore Wind Energy Policy 2015. It aims to help offshore wind energy development, including the setting up of offshore wind power projects.
and research and development activities in waters in or adjacent to the country up to the seaward distance of 200 nautical miles’ exclusive economic zone of the country from the baseline. (Government of India 2017).

10.5 Concluding Remarks

This chapter attempts to describe analytically how forces of globalization—primarily trade and FDI—have impacted the environment in India. Unfortunately, as stated earlier, specific literature on the topic does not contain a significant number of worthwhile research initiatives from analytical and holistic perspectives. We have tried to impress on the readers that the direct impact of trade and FDI on environmental conditions is less of an issue compared to its indirect effect through its positive impact on GDP growth and resultant prosperity. We also emphasize that enforcing regulations is presently a hugely complex task given corruption, informal markets, and the inability of citizens to cooperate and form effective lobbies. We have tried to give simple, readable theoretical inputs and examples. We have provided original time series estimates of the impact of trade and FDI on the environment over the last three and a half decades. We have then elaborated briefly on global climate change research and its Indian counterpart to provide a perspective for our work, followed by a brief summary of recent policy initiatives from the Indian government.

We believe that more specific research is needed to assess the environmental impact of patterns of production and consumption. Recent scientific analysis focuses on better scientific measuring of the damage and the impact of climate change on inequality (Hsiang et al. 2017; Pizer 2017). Clearly, warmer regions in the globe, including India and many developing and Asian countries, are affected by global warming, more than their northern counterparts. Recent US estimates show that climate change has increased inequality between the north and the south of the US; the pattern of production specialization is generally induced by the conditions of global trade and investment, and by the physical infrastructural support. India and the PRC, the two largest countries in Asia, have very different GDP compositions. This poses the question of whether excessive industrialization, coupled with the usual transboundary and climate concerns, makes the PRC more vulnerable than India, which thrives excessively on the growth of the services sector and, in turn, gets the benefit of low pollution growth. This also calls for serious exploration of green accounting and preparation of a database with better environmental indicators, as extensively discussed in Sengupta (2013).
References


11
Trade, Foreign Direct Investment, and Pollution Abatement

Shruti Sharma

11.1 Introduction

The evidence on the effects of international trade and investment on the levels of pollution in developing countries is highly inconclusive. Most studies examine this question by understanding the impact of trade liberalization on emissions, such as levels of nitrogen dioxide and sulfur dioxide in a particular region or country. This chapter uses a more direct method of investigating how international trade and investment might be impacting plant-level behavior toward pollution, by examining plants' spending on pollution abatement equipment. In doing so, this chapter provides a new approach to understanding the relationship between trade, investment, and pollution from an applied microeconomics perspective. To the best of my knowledge, this is the first plant-level study to examine the effects of tariff liberalization on pollution abatement expenditure in a developing-country context. It is also the first study to consider the differential effects of output and input tariff liberalization on plant-level pollution abatement expenditure.

A popular hypothesis regarding the impact on trade and labor is that of Copeland and Taylor (2003). They posit that with increased international trade, developing countries are becoming pollution havens, attracting investment and exporting goods in industries where regulations are typically laxer than in developed countries. Developing and less-developed countries have thus been known to develop a “comparative advantage” in industries known for generating high levels of pollution as lax regulations allow them to produce goods at lower cost than their developed-country competitors. The increase in exports due to their sustained comparative advantage worsens the levels of...
pollution in their respective countries, thereby adversely affecting the environmental conditions. Trade is therefore known to impact the environment of developing countries adversely.

Various studies have empirically tested this hypothesis and have found conflicting results. Cole (2004) finds evidence of pollution haven effects, although these are small compared to the role of other explanatory variables. Levinson (2009), on the other hand, examines this in the context of the manufacturing sector of the United States (US) and finds that pollution levels in the US have declined in the past 30 years. However, the main contributing factor is investment in pollution abatement technology and not the change in the mix of goods manufactured. Further, he shows that increases in net imports of pollution-intensive goods do not play an important role in explaining the reduction in pollution in the US, leading him to conclude that the shifting of pollution-intensive industries overseas is not mainly responsible for the reduced pollution levels in the US. In the context of India, Kathuria (2018) uses data from the manufacturing sector to create an index to compare abatement costs in a particular region (state) at an aggregate level after adjusting for industrial composition, and further measures whether lower-abatement-cost regions experience higher foreign direct investment (FDI) flows. He finds no evidence of the pollution haven hypothesis in the case of India. Grether et al. (2012) find that the magnitude of the polluting effect through delocalization of activities for regional pollutants is small, although the effects might be present.

This analysis, however, considers the differential impact of output and input tariffs on plant-level investment in pollution abatement technology. It draws from the literature that studies the differential effect of output and input tariffs on firm-level productivity and wages. Amiti and Konings (2007) find that input tariff liberalization has a bigger impact on firm-level productivity than output tariffs. This has been confirmed in the case of India by Topalova and Khandelwal (2011) for an analysis of India’s manufacturing sector. In fact, Goldberg et al. (2010) also find that input tariff liberalization allows for increased access to various inputs, which allows firms to produce different kinds of products, thereby increasing their market access.

In a study more relevant to this chapter, Bas and Berthou (2012) study the effect of input tariff liberalization on foreign technology adoption for Indian firms. They show both theoretically and empirically that a reduction in input tariffs causes firms to source foreign technology in the form of capital goods from abroad. Further, Bas and Strauss-Kahn (2015) show that input tariff liberalization in the Chinese context led to imports of higher-quality products. Sharma (2018a) shows that
plants can import out of quality, cost, or variety concerns, and plants
that import inputs of a better quality also experience an increase in
skill composition. The plants that are quality conscious might also seek
access to better production technologies in terms of pollution abatement
equipment.

In this chapter, I investigate whether tariff liberalization or
increased FDI inflows allow plants to access superior technology in
terms of pollution abatement equipment. Do quality-conscious plants
extend their quality consciousness to environmental standards? One can
expect output tariff liberalization to have a negative effect on pollution
abatement expenditure, as increased competition might pressure plants
to cut costs and maintain their comparative advantage. On the other hand,
as input tariffs liberalize, through complementarities between superior
technology and imports of intermediate inputs, one might expect an
increase in investment in pollution abatement technology. Similarly, an
increase in FDI in industries might increase access to better, cheaper
abatement technology or improve the quality consciousness of plants,
thereby causing them to improve plant-level environmental standards
and invest more in pollution control equipment.

I study this in the Indian context using plant-level data from the
Annual Survey of Industries (ASI). This is the most comprehensive
survey of the Indian manufacturing sector. The tariff data have been
obtained from the World Bank World Integrated Trade Solution
database, and the period under consideration is 2002–2008. The data
for FDI are from the National Council for Applied Economic Relations,
and the period of analysis is 2000–2006. I find that a decline in input
tariffs induces plants to spend more on pollution control capital, while a
decline in output tariffs has no significant effect on the same. Second, I
find that with an increase in inward FDI, there is also a positive impact
on plant-level spending on pollution abatement equipment. This is
mainly driven by states that are emerging in terms of attracting FDI,
where the spillover effects are likely the strongest.

The chapter is divided into six sections. Section 2 provides a
summary and statistics from the plant-level, tariff, and FDI data used
for this analysis. Section 3 presents the empirical strategy used for the
analysis. Section 4 discusses the empirical findings using the data and
based on the empirical strategy. Section 5 shows that the results are
robust to alternative specifications. Section 6 presents a concluding
discussion.

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11.2 Data and Summary Statistics

11.2.1 Plant-Level Data

This study focuses on India’s manufacturing using the most comprehensive plant-level data set available from ASI by the Ministry of Statistics and Planning in India. The data contain information on various plant-level characteristics, including that of workers employed by the plant, and on products produced by the plant. Further, it provides information on various capital outlays by the plant, including information on pollution abatement capital, which is important for this study. This data on pollution abatement equipment used by plants is available from the year 2002. For the analysis relating to tariff liberalization, the period under analysis is 2004–2008 and has a total of 2,259 plant-year observations. The summary statistics are presented in Table 11.1. Based on the FDI data, the period of analysis is 2002–2006 and comprises 6,633 observations. The summary statistics used in this panel are presented in Table 11.2.

Table 11.1: Summary Statistics for Tariff Panel

<p>| | | |</p>
<table>
<thead>
<tr>
<th></th>
<th></th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td>Log(Total Workers)</td>
<td>5.764</td>
<td>(1.136)</td>
</tr>
<tr>
<td>Log(Total Pollution Control Capital)</td>
<td>15.08</td>
<td>(2.350)</td>
</tr>
<tr>
<td>Log(Fixed Capital)</td>
<td>19.58</td>
<td>(1.811)</td>
</tr>
<tr>
<td>Log(Working Capital)</td>
<td>18.03</td>
<td>(1.916)</td>
</tr>
<tr>
<td>Log(Total Sales)</td>
<td>20.47</td>
<td>(1.895)</td>
</tr>
<tr>
<td>Input Tariff</td>
<td>2.395</td>
<td>(1.269)</td>
</tr>
<tr>
<td>Output Tariff</td>
<td>16.52</td>
<td>(8.779)</td>
</tr>
<tr>
<td>Observations</td>
<td>2,259</td>
<td></td>
</tr>
</tbody>
</table>

Source: Author’s calculations. Data from the Annual Survey of Industries.
Table 11.2: Summary Statistics for Foreign Direct Investment Panel

<table>
<thead>
<tr>
<th>Variable</th>
<th>Mean</th>
<th>Std. Error</th>
</tr>
</thead>
<tbody>
<tr>
<td>Log(Total Workers)</td>
<td>5.975</td>
<td>(0.956)</td>
</tr>
<tr>
<td>Log(Total Pollution Control Capital)</td>
<td>14.80</td>
<td>(2.208)</td>
</tr>
<tr>
<td>Log(Fixed Capital)</td>
<td>19.54</td>
<td>(1.621)</td>
</tr>
<tr>
<td>Log(Working Capital)</td>
<td>17.94</td>
<td>(1.852)</td>
</tr>
<tr>
<td>Log(Total Sales)</td>
<td>20.19</td>
<td>(1.707)</td>
</tr>
<tr>
<td>Log(FDI)</td>
<td>16.91</td>
<td>(1.792)</td>
</tr>
<tr>
<td>Observations</td>
<td>6,633</td>
<td></td>
</tr>
</tbody>
</table>

FDI = foreign direct investment.

Source: Author’s calculations. Data from the Annual Survey of Industries.

Input Tariffs

Changes in output and input tariffs are used for identification in the empirical exercise of this chapter. While output tariffs are duties on the final product category produced by the plant, input tariffs are the duties on inputs used by the plants in the production process. A reduction of input tariffs would reduce the price of imports of these inputs, and thus induce a plant to import more such inputs. To calculate input tariffs, industry-level shares of inputs used by a particular industry are calculated. The formula used to calculate the same has been given as follows, where an industry (j) uses inputs from various other industries (k):

\[
\text{Input tariff}_{jt} = \sum_k s_{jk} \times \text{output tariff}_{kt}
\]

To produce output j, the share of input used from industry k is denoted by \(s_{jk}\). Input–output tables from India’s Central Statistical Organization have been used to obtain these shares. These shares have been obtained from the tables for the year 2003, which is the year
before the period of analysis (2004–2008), and these do not change for the period under consideration. Output tariffs are collected from the World Bank Database at the three-digit level of National Industrial Classification (NIC). I use a concordance from Ahsan (2013) to map the sector codes from the input–output tables to the three-digit NIC (1998) codes for the input tariff data.

For 2004–2008, the period of analysis, output and input tariffs both declined, as part of India’s ongoing trade reform process. It can be inferred that the rule followed by policy makers was to reduce significantly the tariffs on industries that began with high output and input tariffs. Figure 11.1 reflects this strategy. Also, no strong correlation exists between changes in input tariffs and changes in output tariffs at the three-digit level, with a correlation coefficient of 0.41 (Figure 11.2).

![Figure 11.1: Change in Input Tariffs Relative to Initial Tariff Levels](http://wits.worldbank.org/WITS/WITS/Default-A.aspx?Page=Default)

Note: Based on author’s calculations.
India’s Tariff Liberalization

One may assume that after 1991, tariff liberalization in India was exogenous, especially because it was a response to its balance of payments crisis in August 1991 and due to pressures from the International Monetary Fund (IMF) to liberalize. This change in trade policy as part of India’s economic reform was unanticipated by plants. Since then, India has been reducing tariffs as part of the structural adjustment program and as per the guidelines of the World Trade Organization as a member since 1995. However, there have been some concerns regarding the endogeneity of trade reform in recent years, especially because the pressure from the IMF may have abated, or due to possible government protection of laggard industries, or lobbying from industries to lower tariffs on upstream industries.

I try to address these issues by running robustness checks in this chapter. The tests try to determine whether a significant relationship exists between changes in tariffs and the size of the industry. Previous studies that also consider tariffs for identification in the same period in
India have conducted similar tests. One such study that considers input tariffs during the same period as this chapter to analyze the impact of imports on contract enforcement finds no relationship between total factor productivity at the industry level and input tariffs (Ahsan 2013). I extend this test to consider whether industry size, as measured by various variables, might have any effect on input tariffs. This is to rule out the possibility that big industries, which may not be as productive as others, may also have influenced trade policy. I regress input tariffs at the three-digit NIC level on various measures of industry size (lagged), such as total sales and total employment. The estimation results reveal no significant relationship between input tariffs and industry size (Models 1 and 3, Table 11.3) or between output tariffs and industry size (Models 2 and 4, Table 11.3).

### Table 11.3: Endogeneity Tests for Input and Output Tariffs

<table>
<thead>
<tr>
<th></th>
<th>(1)</th>
<th>(2)</th>
<th>(3)</th>
<th>(4)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Input Tariff</td>
<td>Output Tariff</td>
<td>Input Tariff</td>
<td>Output Tariff</td>
</tr>
<tr>
<td>Lagged Log(Total Workers)</td>
<td>−0.0907 (0.676)</td>
<td>−3.561 (3.631)</td>
<td>−0.147 (0.458)</td>
<td>−2.217 (2.445)</td>
</tr>
<tr>
<td>Lagged Log(Total Sales)</td>
<td></td>
<td>−0.147 (0.458)</td>
<td></td>
<td>−2.217 (2.445)</td>
</tr>
<tr>
<td>Constant</td>
<td>2.860 (6.906)</td>
<td>4.380 (6.846)</td>
<td>5.563 (11.33)</td>
<td>−1.857 (1.422)</td>
</tr>
<tr>
<td>Year Fixed Effects</td>
<td>Yes</td>
<td>Yes</td>
<td>Yes</td>
<td>Yes</td>
</tr>
<tr>
<td>Industry Fixed Effects</td>
<td>Yes</td>
<td>Yes</td>
<td>Yes</td>
<td>Yes</td>
</tr>
<tr>
<td>Observations</td>
<td>72</td>
<td>72</td>
<td>72</td>
<td>72</td>
</tr>
<tr>
<td>Adjusted R²</td>
<td>0.529</td>
<td>0.530</td>
<td>0.530</td>
<td>0.570</td>
</tr>
</tbody>
</table>

*p < 0.10, **p < 0.05, ***p < 0.001.
Note: Standard errors in parentheses.
Source: Author’s calculations. Data from the Annual Survey of Industries.

### 11.2.2 Data on Foreign Direct Investment

This study uses FDI data from a report by the National Council of Applied Economic Research in 2009 using data from the Department of Industrial Policy and Promotion. The report uses statistics from the
Reserve Bank of India. Data are available for the years 2000–2006. I have used inward FDI flows at the three-digit NIC 1998 level. These were obtained using a concordance provided in the report that maps the Department of Industrial Policy and Promotion’s sector-level codes to the three-digit NIC 2004 codes, and further using a concordance between three-digit NIC codes from 1998 to 2004 provided on the Ministry of Commerce and Industry website. For this analysis, a total of 75 industries have been included in the manufacturing sector with significant variation across industries. Because the data on pollution control equipment are only available from 2002 in the ASI, data have been considered for 2002–2006.

Studies on inward FDI in India investigate the determinants of FDI inflows into the country. Banga (2003) examines the role of the investment policies of various states in attracting FDI. She finds that developing economies can attract FDI from developed economies by removing restrictions on FDI. However, to attract FDI, developing states need fiscal incentives and bilateral treaties. Another study (Aggarwal 2005) finds that rigid labor market institutions, while discouraging both domestic market-seeking and export-oriented FDI, have a stronger impact on domestic market-seeking FDI.

Thus, FDI inflows are impacted by state-level policies. This causes FDI in India to be highly regionally concentrated. According to Mukherjee (2011), other state-level factors that have a positive and significant role are the size of the services and manufacturing base in a state, the market size, and the agglomeration effect. Labor costs and taxation policies, on the other hand, negatively impact FDI inflows.

Given these findings, it becomes important to control for these effects in empirical investigations that study the relationship between FDI and plant-level outcomes. My regressions control for these effects; additionally, I also divide all states into three regions—states receiving low FDI inflows, those receiving medium FDI inflows, and those receiving high FDI inflows.

### 11.3 Estimation Strategy

To understand the differential effects of input and output tariffs on expenditure in pollution abatement capital by plants, I estimate the following specification:

\[
\log_{10}\text{pollutioncapital}_{it} = \alpha + \beta_1\text{inputtariff}_{it} + \beta_2\text{outputtariff}_{it} + \beta_3\text{shareimportedinputs}_{it} + \beta_4\text{inputtariff}_{it}\text{shareimportedinputs}_{it} + \beta_5\text{logemployment}_{it} + \beta_6\text{outputtariff}_{it}\text{logemployment}_{it} + \theta_i + \theta_t + \epsilon_{it} \tag{1}
\]
Input and output tariffs vary at the four-digit NIC level. $X_{it}$ controls for the size of the plant; in this case, size is measured as total employment. Fixed effects, $\theta_i$, control for any unobserved time-invariant characteristics of the plant that might affect the coefficients. $\theta_t$ controls for time fixed effects—any year-wise changes that affected all plants equally and could potentially influence the relationship being estimated. Standard errors are robust and have been clustered at the four-digit industry-year level.

Here, I expect $\beta_1$ to be negative, which implies that as tariffs decline, plants spend more on pollution control equipment. One can expect that with an increase in access to imported intermediate inputs, both directly and indirectly through importing suppliers, plants might gain access to superior, cleaner technologies. Further, I expect $\beta_2$ to be positive because with increased competition, a decline in tariffs on the final good might motivate plants to cut costs and reduce expenditure on pollution abatement equipment, especially if environmental regulations are not stringent.

Further, I investigate whether these effects are differential for plants that import more intermediate inputs. From Bas and Berthou (2012), we know that complementarities exist between importing foreign capital and plants that import intermediate inputs. Does this extend to spending on pollution abatement equipment? Below is the specification I estimate:

$$
\text{logpollutioncapital}_{it} = \alpha + \beta_1 \text{inputtariff}_{it} + \beta_2 \text{shareimportedinputs}_{it} + \\
\beta_3 \text{inputtariff}_{it} * \text{shareimportedinputs}_{it} + \beta_4 \text{outputtariff}_{it} + \\
\beta_5 X_{it} + \theta_t + \theta_i + \varepsilon_{it} \quad (2)
$$

Finally, as a control for exporting behavior, following Bernard and Jensen (1997), I use a proxy that measures the size of the plant. In this case, it is the log of total employment. To make it easier to interpret, a model in Table 11.3 (Model 4) also considers log total employment centered around its mean.

$$
\text{logpollutioncapital}_{it} = \alpha + \beta_1 \text{inputtariff}_{it} + \beta_2 \text{shareimportedinputs}_{it} + \\
\beta_3 \text{inputtariff}_{it} * \text{shareimportedinputs}_{it} + \beta_4 \text{outputtariff}_{it} + \\
\beta_5 \text{logtotalemployment}_{it} + \beta_6 \text{outputtariff}_{it} * \text{logtotalemployment}_{it} + \theta_i + \varepsilon_{it} \quad (3)
$$

I move on to analyzing the effects of FDI on logtotalpollutioncapital. Below is the specification I estimate:
\[ \text{logpollutioncapital}_{it} = \alpha + \beta_1 \text{logFDI}_{jt} + \beta_2 \text{logtotalemployment}_{it} + \theta_i + \theta_t + \varepsilon_{it} \] (4)

Here again, FDI varies at the three-digit NIC industry level. With access to better technology in the industry through increased FDI, one can expect an increase in the investment on pollution abatement equipment; for this we expect \( \beta_1 > 0 \).

### 11.4 Estimation Results and Discussion

Table 11.4 considers the impact of input and output tariffs on the opening stock of “logpollutioncapital.” The first model considers the effect of input tariffs only. The coefficient on input tariffs is negative and significant at the 5% level, suggesting that as input tariffs in their respective industries decline, plants spend more on pollution abatement equipment. This corroborates the hypothesis that input tariff liberalization, with access to better inputs and superior technology, actually improves the quality consciousness of a plant and induces it to spend more on pollution abatement. In Model 2, only the impact of a decline in output tariffs is considered. The impact of output tariffs on logpollutioncontrolcapital turns out to be positive, but insignificant. The positive coefficient suggests that an increase in competition might induce plants to cut costs and reduce investment in pollution control capital. This effect, however, is insignificant. The third model considers the impact of both input and output tariffs on pollution control equipment. The inclusion of output tariffs does not affect the sign or significance of the coefficient on input tariffs. The coefficient on output tariffs continues to be positive and insignificant. In the final model, logtotalemployment as a measure of plant size is introduced with both tariffs. I find that the effects remain the same, negative, and significant on input tariffs and positive and insignificant on output tariffs. This exercise corroborates the hypothesis that tariff liberalization might actually motivate plants to adhere to higher environmental standards, especially with access to cheaper and technologically superior inputs, which brings down the cost of production and provides better access to cleaner technologies. Furthermore, there is no strong evidence that increased competition through trade adversely impacts investment in pollution abatement equipment.

In the next set of analyses, I consider another aspect of globalization—FDI. Table 11.5 examines the effect of inward FDI on
“logpollutioncapital.” Inward FDI is considered at the three-digit NIC industry level. Model 1 considers the impact of inward FDI only, and the coefficient on logFDI is positive and significant at the 10% level. This suggests that increased FDI inflows in an industry a plant belongs to are associated with higher levels of spending on pollution abatement capital. This means that as an industry gets more globalized in terms of increased FDI inflows, plants start investing more in pollution abatement equipment. Model 2 controls for plant size by including “logtotalemployment,” and I find that the coefficient on logFDI continues to be positive and significant. Given that FDI inflows are influenced by state legislations as highlighted in Section 3.3, I control for state-fixed effects in Model 3 and state-year fixed effects in Model 4. I find that effects of FDI on spending on pollution control capital continue to be positive and significant.

### Table 11.4: Impact of Input and Output Tariff on Pollution Control Expenditure

<table>
<thead>
<tr>
<th></th>
<th>(1)</th>
<th>(2)</th>
<th>(3)</th>
<th>(4)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Log(Total Pollution Capital)</td>
<td>Log(Total Pollution Capital)</td>
<td>Log(Total Pollution Capital)</td>
<td>Log(Total Pollution Capital)</td>
</tr>
<tr>
<td>Input Tariff</td>
<td>–0.0502**</td>
<td>–0.0620**</td>
<td>–0.0586**</td>
<td></td>
</tr>
<tr>
<td></td>
<td>(0.0237)</td>
<td>(0.0223)</td>
<td>(0.0195)</td>
<td></td>
</tr>
<tr>
<td>Output Tariff</td>
<td>0.00207</td>
<td>0.00526</td>
<td>0.00539</td>
<td></td>
</tr>
<tr>
<td></td>
<td>(0.00466)</td>
<td>(0.00408)</td>
<td>(0.00403)</td>
<td></td>
</tr>
<tr>
<td>Log(Total Workers)</td>
<td></td>
<td></td>
<td></td>
<td>0.169**</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td>(0.0660)</td>
</tr>
<tr>
<td>Constant</td>
<td>15.07***</td>
<td>14.86***</td>
<td>14.97***</td>
<td>13.99***</td>
</tr>
<tr>
<td></td>
<td>(0.0816)</td>
<td>(0.126)</td>
<td>(0.136)</td>
<td>(0.370)</td>
</tr>
<tr>
<td>Year Fixed Effects</td>
<td>Yes</td>
<td>Yes</td>
<td>Yes</td>
<td>Yes</td>
</tr>
<tr>
<td>Observations</td>
<td>2,259</td>
<td>2,259</td>
<td>2,259</td>
<td>2,253</td>
</tr>
<tr>
<td>Adjusted $R^2$</td>
<td>0.038</td>
<td>0.036</td>
<td>0.038</td>
<td>0.047</td>
</tr>
</tbody>
</table>

* $p < 0.10$, ** $p < 0.05$, *** $p < 0.001$.

Notes: All regressions include plant fixed effects. Standard errors are robust and clustered at the three-digit National Industrial Classification year level. Standard errors in parentheses.

Source: Author’s calculations. Data from the Annual Survey of Industries.
In this section, I consider robustness checks for the analysis of the impact of tariff liberalization on pollution control equipment, and a region-wise analysis of the impact of FDI. The robustness checks for the tariff liberalization analyses are presented in Table 11.6. Here, I consider the differential effects of input and output tariffs on importing and non-importing plants, as well as potentially exporting and non-exporting plants. While information on the exporting status of plants is not available, following Bernard and Jensen (1997), I use the size of the plant as a proxy for exporting behavior. Model 1 considers the share of imported intermediate inputs in the total inputs purchased by the plant. I find that on average, there is no significant difference between plants that import intermediate inputs and plants that do not in terms of expenditure on pollution control equipment. Model 2 includes the interaction of this share of imported intermediate inputs with input tariffs. I do not find
Table 11.6: Differential Effects for Importing and Potentially Exporting Plants

<table>
<thead>
<tr>
<th></th>
<th>(1)</th>
<th>(2)</th>
<th>(3)</th>
<th>(4)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Log(Total Pollution Control Capital)</td>
<td>Log(Total Pollution Control Capital)</td>
<td>Log(Total Pollution Control Capital)</td>
<td>Log(Total Pollution Control Capital)</td>
</tr>
<tr>
<td>Input Tariff</td>
<td>-0.0582**</td>
<td>-0.0794**</td>
<td>-0.0791**</td>
<td>-0.0689**</td>
</tr>
<tr>
<td></td>
<td>(0.0172)</td>
<td>(0.0236)</td>
<td>(0.0261)</td>
<td>(0.0255)</td>
</tr>
<tr>
<td>Share of Imported Expense (%)</td>
<td>0.00291</td>
<td>-0.000957</td>
<td>-0.000753</td>
<td>-0.000462</td>
</tr>
<tr>
<td></td>
<td>(0.00224)</td>
<td>(0.00347)</td>
<td>(0.00344)</td>
<td>(0.00328)</td>
</tr>
<tr>
<td>Output Tariff</td>
<td>0.00540</td>
<td>0.00519</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>(0.00401)</td>
<td>(0.00411)</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Log(Total Workers)</td>
<td>0.187**</td>
<td>0.188**</td>
<td>0.172**</td>
<td>0.172**</td>
</tr>
<tr>
<td></td>
<td>(0.0834)</td>
<td>(0.0822)</td>
<td>(0.0786)</td>
<td>(0.0786)</td>
</tr>
<tr>
<td>shareInputTariff</td>
<td>0.00138</td>
<td>0.00130</td>
<td>0.00121</td>
<td>0.00121</td>
</tr>
<tr>
<td></td>
<td>(0.00102)</td>
<td>(0.00101)</td>
<td>(0.000949)</td>
<td>(0.000949)</td>
</tr>
<tr>
<td>outputWorkers</td>
<td></td>
<td>0.000999</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>(0.000712)</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Centered Log(Total Workers)</td>
<td></td>
<td>0.00232</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>(0.00215)</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Constant</td>
<td>13.87***</td>
<td>13.93***</td>
<td>14.00***</td>
<td>14.93***</td>
</tr>
<tr>
<td></td>
<td>(0.481)</td>
<td>(0.447)</td>
<td>(0.437)</td>
<td>(0.0968)</td>
</tr>
<tr>
<td>Year Fixed Effects</td>
<td>Yes</td>
<td>Yes</td>
<td>Yes</td>
<td>Yes</td>
</tr>
<tr>
<td>Observations</td>
<td>2,200</td>
<td>2,200</td>
<td>2,200</td>
<td>2,200</td>
</tr>
<tr>
<td>Adjusted R²</td>
<td>0.053</td>
<td>0.054</td>
<td>0.055</td>
<td>0.054</td>
</tr>
</tbody>
</table>

*p < 0.10, **p < 0.05, ***p < 0.001.
Notes: All regressions include plant fixed effects. Standard errors are robust and clustered at the three-digit National Industrial Classification year level. Standard errors in parentheses.
Source: Author’s calculations. Data from the Annual Survey of Industries.

any differential effects of directly importing intermediate inputs. One explanation is that plants could access a large supply of imported inputs through domestic suppliers. The coefficient on input tariffs continues to be negative and significant, suggesting that access to these inputs plays an important role in pollution abatement expenditure. Model 3
considers the interaction term of two continuous variables—outputtariff and logtotalemployment—to show that plants that have more workers (or are bigger in size and thus more likely to export) invest more in pollution equipment as output tariffs decline. This, however, is not very informative, as the differential effect is between plants with nonzero workers and those with zero workers. To make this more informative, Model 4 centers logtotalemployment around the mean, so we can compare the effect of a decline in output tariffs between plants that are mean-sized and higher and plants that are below the mean size. I find no differential effects on spending in pollution control equipment based on potential exporter status. This suggests that exporters and non-exporters are equally unlikely to reduce investment in pollution control equipment as competition increases through a decline in output tariffs.

Based on the extensive literature on FDI inflows in India, which highlights the concentration of FDI in certain regions and the role of state-level policies in determining the same, I conduct a region-wise analysis of the impact of FDI on plant-level pollution abatement. I divide the regions into low FDI recipients, medium FDI recipients, and high FDI recipients and investigate the effects. The results are presented in Table 11.7. I find that most of the effects are driven by states receiving

<p>| Table 11.7: Regional Heterogeneity in Differential Effects of Foreign Direct Investment |
|-----------------------------------------|-----------------------------------------|-----------------------------------------|</p>
<table>
<thead>
<tr>
<th></th>
<th>Low FDI</th>
<th>Medium FDI</th>
<th>High FDI</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Log(FDI)</strong></td>
<td>0.0136</td>
<td>0.0324*</td>
<td>0.0105</td>
</tr>
<tr>
<td></td>
<td>(0.0138)</td>
<td>(0.0190)</td>
<td>(0.00816)</td>
</tr>
<tr>
<td><strong>Log(Total Workers)</strong></td>
<td>-0.116</td>
<td>0.273***</td>
<td>0.0437**</td>
</tr>
<tr>
<td></td>
<td>(0.0738)</td>
<td>(0.0515)</td>
<td>(0.0217)</td>
</tr>
<tr>
<td><strong>Constant</strong></td>
<td>14.25***</td>
<td>12.52***</td>
<td>14.32***</td>
</tr>
<tr>
<td></td>
<td>(0.439)</td>
<td>(0.470)</td>
<td>(0.215)</td>
</tr>
<tr>
<td><strong>Year Fixed Effects</strong></td>
<td>Yes</td>
<td>Yes</td>
<td>Yes</td>
</tr>
<tr>
<td><strong>Observations</strong></td>
<td>470</td>
<td>1,030</td>
<td>4,970</td>
</tr>
<tr>
<td><strong>Adjusted R²</strong></td>
<td>0.026</td>
<td>0.036</td>
<td>0.027</td>
</tr>
</tbody>
</table>

FDI = foreign direct investment.

*p < 0.10, ** p < 0.05, *** p < 0.001.

Notes: All regressions include plant fixed effects. Standard errors are robust and clustered at the three-digit National Industrial Classification year level. Standard errors in parentheses.

Source: Author’s calculations. Data from the Annual Survey of Industries.
medium FDI. This is an interesting finding, and perhaps suggests that for the period under consideration, the effects of access to better technology for pollution abatement were most important for regions that were becoming more prominent in terms of contenders for inward FDI. Sharma (2018b) argues that a critical mass of FDI needs to be achieved in a region before plants experience the spillover effects. Thus, plants in states that receive small FDI inflows have not yet experienced these benefits. On the other hand, it is also possible that those that received high FDI levels had already benefited from access to technology and plant-level spillovers, and therefore no longer experience such benefits from increased investments.

11.6 Conclusion

This chapter considers the impact of globalization, both in terms of tariff liberalization and inward FDI, on plant-level expenditure on pollution abatement. The debate on the impact of trade and investment on pollution adds a new perspective—a microeconomic foundation for understanding how plants behave in a globalized environment. The analyses focused on tariff liberalization consider the differential impact of output and input tariffs on plant-level expenditure in pollution control equipment. I find that input tariff liberalization, through increased access to better technology, induces plants to invest more in pollution control equipment. Further, a decline in output tariffs that increase competition in an industry does not seem to play an important role in discouraging plants from investing in pollution abatement capital, as the pollution haven theories might indicate. The second main analysis focuses on the role of FDI. I find that with increased inflows of FDI in their industry, plants get access to better, cleaner technologies, which is complementary to higher spending on pollution abatement capital. The effect, although less strong, is similar to that of access to imports of intermediate inputs. This is robust to the inclusion of state-year fixed effects and state-fixed effects, which control for the fact that FDI inflows in India tend to be regionally concentrated. I further analyze the relationship between FDI and pollution abatement equipment across regions. I divide the states into low FDI, medium FDI, and high FDI recipients. I find that plants in states that are medium FDI recipients largely drive this effect. These states are relatively new in terms of receiving inflows of FDI compared to other states (high FDI recipients) that have been receiving these flows historically. Thus, they are still enjoying the spillover effects of better technology access as industry-level inflows of FDI increase.
The findings in this chapter suggest that it is important to understand the relationship between trade, investment, and pollution from a microeconomic perspective. From a policy perspective, an interesting finding is that input tariffs can actually induce plants into spending more on pollution abatement equipment. Further, it is also informative that spending on pollution abatement equipment might not be affected by increased competition in the industry. Overall, these findings add a new dimension to the whole debate on globalization and pollution by focusing on an aspect aside from levels of pollutants in a region, and actually understanding plant-level behavior in terms of investment in pollution abatement.
References


Ways to Achieve Green Asia

Escalating environmental degradation and the risk of climate change are attracting growing attention from both policy makers and the public. For Asian countries, decades of remarkable economic growth have had mixed results in terms of environmental implications.

This book provides a comprehensive analysis of various aspects of the environment and climate change in Asia. It first gives an overview of the environmental challenges facing the region and summarizes the economic impacts of climate change. It also offers in-depth discussions on environmental regulations, environmental governance, environmental evaluation, and the growth of carbon markets in Asia. The volume finally explores the relationship between globalization and the environment, particularly through informative case studies on the People’s Republic of China and India. Along this vein, this book aims to advance policy recommendations for more effective environmental and climate change governance in the region.

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