Environmental Risks, Stranded Assets, and the Impacts of Regulation on Firm Productivity



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This DPhil thesis is dedicated to my mother.

Abstract

Economic activity can and does harm the environment. But environmental factors and environmental policy can also impact economic activity and productivity. This thesis explores this second connection – the impact of environmental risks and regulation on economic activity – with three novel quantitative research papers.

The first paper investigates the impact of georeferenced environmental risks on the postponement or cancellation of Chinese investments in overseas power projects. Using survival analysis, this paper finds that coal power projects are more likely to be suspended if they are located near densely populated areas, or if they are in countries with more deaths from extreme weather events. This paper also finds that hydropower projects are more likely to be suspended if they are closer to protected areas.

The second paper examines the risk of fossil fuel asset stranding in the power sector, exploring whether technologies such as carbon capture and storage and bioenergy could reduce the risk of stranding. Combining Integrated Assessment Model climate scenarios with global power plant data, this paper shows that even if such technologies are fully deployed and existing power plants are converted to be less carbon-intensive, there is still a high risk of stranding.

The third paper employs a heterogeneous difference-in-difference approach to study the impact of air pollution regulation on Chinese industrial firms' productivity and finds evidence of net positive effects, i.e. stringent environmental policy increases productivity. This is possibly driven by process innovation and firm sorting.

The results highlight the complexity of environmental policymaking. Signalling the direction of future environmental policy, and incorporating it into investment decisions, can avoid unexpected stringent changes, asset stranding and financial losses. In some cases, environmental policies can simultaneously protect the environment and foster economic development.

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1 Introduction

The introductory chapter begins with the background and motivations of the DPhil research, contextualises research topics and objectives, and gives an overview of the methods and data used. It also presents the main results and lays out the structure of the DPhil thesis.

1.1 Background and motivations

1.1.1 A global sustainable transition

The interactions between environment and economy are multi-dimensional and complex. The environmental system plays a crucial role in economic development through the many services it provides. Natural resources are used as energy and material inputs in production activities that enable value creation. Key elements such as air, water and soil support life-essential functions and act as a sink for diluting and filtering the adverse effects generated from the human system. High environmental quality improves human well-being and provides recreation opportunities (MacKerron and Mourato, 2009).

While the ecological system influences economic activities, the economic system has also resulted in a considerable impact on the environment. Overexploitation of natural resources leads to the degradation of ecosystems. Pollutants generated from industrial processes deteriorate environmental quality, causing detrimental impacts to human health (Chay and Greenstone, 2003, Ebenstein et al., 2017). The production-derived pollution may, in turn, impede economic activities through deteriorating worker productivity (Zivin and Neidell, 2012) and decreasing labour supply (Hanna and Oliva, 2015).

Specifically, air pollution is one of today's biggest challenges to sustainable development. A global total of 10.2 million premature deaths annually could be at-

tributed to the fossil-fuel component of particulate matter ($PM_{2.5}$), with the greatest mortality impact in China (3.9 million) and India (2.5 million) (Vohra et al., 2021). For example, total suspended particulates (TSPs) air pollution is causing a loss of more than 2.5 billion years of life expectancy among 500 million residents in Northern China due to the heavily subsidized coal-fired winter indoor heating (Chen et al., 2013).

Additionally, greenhouse gases derived from human activities increase the CO_2 levels in the atmosphere, exposing the world to catastrophic risks such as sea-level rise, extreme heat, and more frequent hurricanes and storms. Over recent years, climate change has not slowed down, and its devastating impacts on the human system are increasingly visible. Extreme weather events have hit the poorest countries as well as established economies, resulting in 475,000 deaths and 2.56 trillion USD (in purchasing power parities) between 2000 and 2019 globally (Eckstein et al., 2021). Limiting global temperature rise is a pressing need and requires coordinated global action at an unprecedented scale and speed. The major economies, including the G7 countries and China, have committed to achieving net-zero around mid-century. This requires a system-wide and radical transition towards economic growth pathways compatible with climate change.

There is a need to promote investment in the right direction to accelerate the transition. UNCTAD estimates that an investment of around 3.9 trillion USD is needed each year from 2015 to 2030 to fully meet the 17 Sustainable Development Goals (SDGs) in the developing world (UNCTAD, 2018). With the enormous investment needs, it is critical to distinguish the direction of public spending and private investment. Continued financing in carbon-intensive sectors may eventually turn these assets into "carbon bubbles" (Weyzig et al., 2014), while investing in technologies that enhance the global environmental system could achieve economic and environmental benefits simultaneously (Hepburn et al., 2018).

Technologies are playing an essential role in leading the global sustainable tran-

sition. We are in the midst of a new era characterized by transformative change and disruptive innovation. Technologies could increase the productivity of production systems while improving energy efficiency and minimizing waste. Advanced materials for solar panels and battery technologies have carved out a low-carbon path to meet the increasing electricity demands. Emerging innovations, such as negative emission technologies, have a considerable potential to back up the net-zero target, though the scale needs to be better understood.

A decisive global green transition is practical and of mutual benefits to all countries around the world. With the COVID-19 pandemic disrupting the economic development and delaying the sustainable transition pace, the world is at a crucial crossroad to shift the business-as-usual approach towards a green growth path.

1.1.2 The role of government intervention

To accelerate the sustainable transition, policies that have high potential on both economic multiplier and environmental metrics are favoured (Hepburn et al., 2020). This motivates the discussion on the rationale for environmental policy. The characteristics of the natural environment bring about several market failures (Everett et al., 2010). Firstly, many environmental goods are usually public goods. Their non-rival and non-excludable characteristics mean that consumers could free ride to benefit from the good. For example, access to clean air is not restricted, and an individual benefiting from clean air does not reduce its availability for others.

Secondly, the consumption and production of certain goods and services create externalities – such as pollution – creating a wedge between their social and private costs and benefits. In the absence of regulation, companies discharging pollutants are not fully charged for the cost of their activities, i.e. recreational benefits foregone, the effects on human health, and the economic cost to remediate the environmental quality degradation. Last but not least, these externalities also dampen the incentive for the private sector to invest in clean innovation activities because

they do not capture the full benefits of the investment to society.

In addition to public goods and externalities, other market failures such as myopia and deep uncertainty also justify intervention through policymaking and regulation (Stern and Stiglitz, 2021). In the absence of government intervention, markets alone do not ensure that prices reflect these social costs and benefits, leading to sub-optimal production and consumption of many goods and services.

There are many policy and regulation instruments available for addressing these market failures. Market-based instruments that address externalities such that prices better reflect the benefit or cost of economic activities are commonly advocated as an efficient approach. For example, carbon taxes can account for the social cost of carbon, which is not fully captured by fossil fuel prices. Command-and-control regulations, such as setting technology and performance standards on polluting activities, are considered less flexible than market-based instruments. Nevertheless, direct regulations are preferred when the pollutant is toxic or the impact of non-compliance far outweighs economic efficiency concerns (Everett et al., 2010).

Apart from the market-based and command-and-control regulations, instrument tools such as public spending and information provision are also used. For example, government grants on environmental R&D could spur clean technology invention, and subsidies could promote the diffusion of low-carbon technologies. Another example is eco-labelling, which fills the information gap between producers and consumers on environmental footprints of products, is aimed to encourage environmental-friendly behaviour change.

1.1.3 Economy versus environment debate

To address pollution and climate concerns, policymakers worldwide are formulating diverse forms of environmental policies. For example, the Clean Air Act enacted in 1963 in the United States is one of the most comprehensive regulations in the world to control air pollution (EPA, 2021). European Commission launched the world's

first international emissions trading market — EU ETS carbon trading scheme — in 2005 to combat climate change (European Commission, 2021). Confronted with severe air pollution issues, China reinforced its Environmental Protection Law in 2015, introducing stricter punishments on polluters. The emerging economies are also taking actions. According to Climate Action Tracker, two developing countries — Morocco and Gambia — are the only countries globally that undertake climate actions compatible with the 1.5 degrees target (Climate Action Tracker, 2021).

With numerous environmental policies in place, their impacts on the economy have been heatedly debated. Concerns arise from the burden of regulatory efforts, worrying that ensuring environmental quality is at the expense of slowing economic growth. For example, expenditures for environmental protection in the United States were estimated to exceed 2% of GDP in the 1990s (Morgenstern et al., 2001) and recent commentary argues that to achieve an almost net-zero in the US (a 95% reduction in carbon emissions compared to 2005 levels) by 2050 would annually cost 11.9% of the nation's GDP (Lomborg, 2021). Some theoretical models indicate that pollution abatement costs could impede economic competitiveness (Carbone and Rivers, 2017). A few empirical studies find that environmental regulations could shift production and labour away from regulated industries (Greenstone, 2002, Kahn, 1997) and lead to falling in export (Hering and Poncet, 2014) and foreign direct investment (Cai et al., 2016b).

On the other side, positive consequences of environmental regulations have been found in many economic sectors, such as spurring the evolution of the recycling industry and initiating the emergence of a circular economy (Gurtoo and Antony, 2007). More stringent environmental standards could improve resource efficiency in the production process (Wang et al., 2011) and induce productivity-improved innovation (Porter and Van der Linde, 1995). From a medium- and long-term perspective, integrating measures to tackle climate change into regular growth policymaking can positively impact the economy (OECD, 2017). The annual infrastructure in-

vestment in line with the Paris Agreement is estimated to be USD 6.9 trillion until 2030, only 10% more expensive than the carbon-intensive alternative. In addition, the extra cost could be offset by the savings from energy efficiency contributed by climate-resilient infrastructure. The OECD report suggests that combining growth and climate agendas, instead of setting climate targets as a separate issue, could add 2.8% to average economic output in G20 countries by 2050.

The centrepiece of this public discourse is often framed as an "economy versus environment" trade-off. There has been discussion over whether it is possible to achieve economic growth and environmental performance hand in hand. Given the detrimental impacts of environmental change and the controversial regulatory costs, it is essential to thoroughly understand the relationships between environment and economy to foster a sustainable economic growth path.

1.2 Research topics and objectives

This DPhil thesis is aimed to contribute to a more thorough understanding of the interactions between environment and economy, providing implications for energy investment, asset stranding, and firm productivity. Accordingly, three DPhil research papers are developed examining:

1. whether and which environmental risks could affect power plant investment project failures; and

2. the extent to which technological solutions such as carbon capture and storage and bioenergy deployment may reduce fossil fuel power plant stranding risk under climate constraints; and

3. the effect of air pollution regulation on firm productivity in a developing country context and which mechanisms may explain any observed productivity changes.

Firstly, the DPhil thesis explores how environmental risks impact the likelihood of energy investment project failure. A significant amount of attention has been given to proposed and under construction power projects' potential impact on the environment (Myllyvirta et al., 2021, Shearer and Buckley, 2019, Tong et al., 2019, Chen et al., 2020), but there is a lack of understanding on whether these projects could be successfully commissioned (Alova et al., 2021). Most of the existing energy investment risk literature uses ex-ante methods such as interviews and scenario analysis to identify potential risks that energy investments could face (Duan et al., 2018, Egli, 2020, McCollum et al., 2018, Khan et al., 2021, Tietjen et al., 2016, Yuan et al., 2019, Gorbacheva and Sovacool, 2015). However, the consequences of these risks on energy investments are poorly understood.

The first DPhil paper addresses one component of this question by examining Chinese overseas investments in power plants. Over recent years, China's overseas investments in the global electric power sector have increased rapidly, with two Chinese policy banks—the China Development Bank and Export-Import Bank of China—providing approximately 245.8 billion USD in energy finance to more than 80 countries since 2000 (Gallagher, 2021). The fate of China's overseas energy investments, however, is not always assured. Chinese-backed coal-fired power plants worth more than 65 billion USD have been shelved, mothballed or cancelled, and many more projects have been considerably delayed (CREA, 2021, Nedopil Wang, 2021). Environmental issues have been a central focus in the discussion on energy investment, triggered by concerns about carbon emission increase, environmental quality degradation, and biodiversity loss (Ascensão et al., 2018, Hale et al., 2020, Narain et al., 2020). Given the non-negligible environmental impact that power plants would have, the first DPhil paper identifies specific environmental issues for different power generation technologies and tests whether these risks are associated with investment failures.

Next, the DPhil thesis assesses the role of technologies in mitigating economic

losses from climate policies. Stringent environmental policies may lead to financial loss, especially when investments were poorly thought through in the first place. Fossil fuel power plants have been deployed extensively over the past years, particularly in China and India, where coal-fired power capacity increased by 369 GW and 129 GW between 2010 and 2019, respectively (Carbon Brief, 2020). The existing fossil-fuel power plants are at high risk of stranding (i.e. underused or decommissioned early) if we are to attain the objectives of the Paris Agreement (Pfeiffer et al., 2016, 2018a, Davis and Socolow, 2014). However, the energy industry has argued that technological solutions, especially carbon capture and storage (CCS), bioenergy, and fuel switching to less carbon-intensive sources, could substantially reduce the carbon intensity of fossil fuel power plants and allow fossil fuel business to continue for longer without jeopardising the Paris Agreement (Glencore, 2017, IEA Coal Advisory Board, 2013, Repsol, 2019, Shell, 2019, Total, 2020). So far, no study has estimated the effect of technology deployment and plant conversions on asset stranding, which is important for understanding the required pace for phasing out fossil fuels.

The second DPhil paper addresses this research gap by estimating global fossil fuel power plant stranding risk under optimistic assumptions of technology development, i.e. CCS and other technologies are fully deployed, and existing plants are converted to be less carbon-intensive. The results have significant implications for investors and policymakers since most countries are currently deciding how fast they should decarbonize their economies. Coal phase-out plans have just started to be adopted in several countries with very different timelines.

Lastly, the DPhil thesis provides new evidence from China to understand the long-debated question of whether environmental regulation dampens economic competitiveness. Despite concerns among policymakers that environmental regulations may dampen productivity, there is little evidence of this being true in the literature so far (Dechezleprêtre and Sato, 2017). On the other hand, there is increasing

evidence pointing towards environmental regulations inducing cleantech innovation (the weak form of "porter hypothesis")—such as by increasing R&D and patenting in environmentally-friendly technologies (Aghion et al., 2016, Brunnermeier and Cohen, 2003, Calel and Dechezleprêtre, 2016, Jaffe and Palmer, 1997)—but there is little work on whether this translates into productivity gains so far, i.e. the strong form of "porter hypothesis" (Dechezleprêtre and Sato, 2017).

Until recently, most papers studying productivity effects of environmental regulations have focused on developed countries. Findings in developed economies cannot be simply transferred to the developing country context given the different incentive structures, bureaucratic norms, and socioeconomic conditions (Greenstone and Jack, 2015, King and Mori, 2007). Understanding the economic effects of environmental regulations is particularly salient for the developing world. These regions disproportionately bear the burden of pollution while also facing widespread poverty. There is a growing literature examining environmental regulations' economic outcomes in China (Almond et al., 2009, Liu et al., 2021), but the implications for firm productivity are not well understood.

The third DPhil paper narrows this knowledge gap by providing new evidence on the impact of an air pollution regulation in China on industrial firm productivity. This paper studies the net impact of the environmental regulation on productivity for firms in both pollution-intensive and less polluting industries. It also explores firms' strategies to comply with environmental regulation and investigates whether these strategies translate into productivity change.

1.3 Overview of data and methods

This DPhil thesis is data-driven. It applies quantitative approaches such as survival analysis, scenario modelling and econometrics to answer the empirical research questions. This thesis combines micro-level data with diverse forms of novel datasets

such as satellite data, mapping data and lexical analysis data that have not been widely employed in economics research.

Specifically, the first DPhil paper develops georeferenced proxy variables to reflect the environmental claims expressed in global resistance movements to technology-specific power projects, as systematically mapped in the Global Atlas of Environmental Justice (EJAtlas). It draws high-resolution population density around coal-fired power projects from satellite data as a measurement of the number of people exposed to pollution—the most prominent local environmental risk associated with coal-fired projects. It also collects country-level climate-related fatalities as a proxy for the salience of perceived climate vulnerability. This paper further uses projects' distance to officially designated ecologically protected areas as a proxy for the potential risks posed by projects to ecosystems of high ecological importance and biodiversity value. In addition, this paper includes the presence of place-based environmental protests, recorded in Google's Global Database for Events, Language and Tone (GDELT), as a proxy of the intensity of environmental movements in the local area.

In terms of methodological approach, the first DPhil paper applies survival analysis to the right-censoring power plant data. In the analysis, the dependent variable is the hazard rate, which is the conditional probability that a failure event (i.e. an energy project is cancelled or delayed) occurs at a particular time interval. The survival functions can help identify the determinants of project suspensions associated with different power generation technologies. The main variables of interest are power generation technology types and environmental risks. To control for confounding factors, the analysis controls for project-, firm-, and country-level characteristics.

The second DPhil paper uses two categories of data inputs: global power plant data and Integrated Assessment Models (IAMs) climate scenario data. The global power generator-level data are compiled by merging several databases: Global Coal

Plant Tracker, S&P Global Platt's World Electric Power Plants database and WRI's global database of power plants. They are used to estimate future electricity generation from currently operating, planned and under construction plants. Climate scenario data are gained from climate scenarios in the AMPERE database, which is a modelling comparison project that integrates outputs from different IAMs to improve understanding of possible scenarios toward climate targets. Specifically, AMPERE database provides information on the evolution of electricity generation throughout the end of the century compatible with achieving climate targets, given different technology availabilities (Riahi et al., 2015).

This paper combines the bottom-up global power plant data with climate scenario data. It firstly computes the power sector's asset stranding under optimistic assumptions that all carbon abatement technologies (CCS, bioenergy, solar and wind, and nuclear energy) will be available and fully deployed. Then it assesses the impact of plant conversions on asset stranding by allowing power plants to be converted into less polluting assets, considering three conversion options: 1) coal to gas; 2) adoption of CCS in operating and in-the-pipeline plants; and 3) coal to biomass. For each conversion option, this paper develops assumptions on conversion feasibility and potential on a global scale. It further compares estimation results across pairwise technology scenarios (i.e. the scenarios where all technologies are fully available and the scenarios where one single technology, for example CCS, is not available) to evaluate the impact of technology availability on asset stranding.

To identify the causal effect of environmental regulation on firms' productivity, the third DPhil paper employs a heterogeneous difference-in-difference approach leveraging three sources of variation: geographic variation (whether the prefecture is designated as regulated), timing variation (pre- and post-regulation implementation), and pollution intensity variation (whether the firm is in a more or less pollution-intensive industry). Importantly, this approach of also examining firms in less pollution-intensive industries allows for the entire industrial sector to be stud-

ied rather than just firms that are the "dirtiest" . Less pollution-intensive firms are still regulated in the sense that they are encouraged and required to reduce their emissions—it is just that they face lower compliance costs than those that are more pollution-intensive. Therefore, it is important to consider the impact on these less pollution-intensive firms as well in order to capture the net effect on all industrial activities.

To measure firm productivity, this paper gathers firm-level data from the China Industrial Enterprise Database (CIED), which includes key measures on Chinese manufacturing firms' economic performance such as output, labour, profit and capital. This paper enhances these firm-level data with industry-specific pollutant emissions intensity information and prefecture-level sociodemographic data from government statistical yearbooks and satellite-based pollution concentration levels.

1.4 Preview of main results

The first DPhil paper, *Environmental risk and power plant suspension*, identifies approximately 5% of the 1393 power projects with Chinese overseas investments over the past two decades have been cancelled or postponed at some point in their development, with coal and hydro power projects having a higher chance of being suspended. By applying survival analysis, this paper further finds hydro and coal project suspensions are associated with technology-specific georeferenced environmental risks. Coal projects are more likely to be suspended when they are: located in more densely populated areas where more people are exposed to air pollutants; in countries with more fatalities from extreme weather events; and in places with environmental protests. Additionally, hydro projects closer to protected areas have a higher suspension rate.

Results in this paper suggest that environmental risks may increase the likelihood that power investments will fail. If environmental concerns are left unchecked,

they could translate investments into financial losses by delaying or cancelling the project. The findings send a clear message to investors that they need to question funding environment damaging projects not only on their detrimental environmental impacts but also on financial grounds.

The second DPhil paper, *Plant conversions and abatement technologies cannot prevent stranding of power plant assets in 2°C scenarios*, finds that even if CCS and bioenergy are deployed quickly and extensively in the 21st century, an average 267 PWh of electricity generation (ten times of global electricity production in 2018) from fossil fuel power plants could be at risk of stranding under a 2°C climate target. If operating and in-the-pipeline coal power plants were sufficiently converted to gas, stranding could be reduced by up to 30 PWh. The adoption of CCS and the co-firing of biomass may reduce stranding by 33 to 68 PWh, depending on the share of CCS-suitable plants finally converted and the assumed co-firing ratio. If either CCS or bioenergy is not deployed to full potential, the amount of electricity generation at risk of stranding would rise by 69% or 45%, respectively.

Building upon the recent stranded assets literature, the second DPhil paper advances our understanding of the scale of asset stranding in different technology scenarios. Our findings suggest that it is unwise to rely on optimistic forecasts of technology development in the future, especially when these solutions are not viable to mitigate the risk. To reduce the risk of investments being lost, investors need to fully account for stranding risks now.

The third DPhil paper, *Environmental regulation and productivity are not always at odds: evidence from firms in China*, finds the environmental regulation in China increases productivity for firms in less pollution-intensive industries by 4% while there is no effect on those in the more pollution-intensive industries. The net effect is therefore positive, highlighting the importance of identifying the effects on not just pollution-intensive firms but also those that are less pollution-intensive (but still regulated) to capture the effect of environmental regulation on all industrial

activities.

This paper further finds that the positive net effects are likely driven by firm sorting and efficiency-enhancing processes and practices. Firms that were less productive in the pre-regulation tended to exit more, increasing the average productivity. Besides, firms have invested in high-skilled workers through increasing average wages and expenditures on management, while there is no change in new products related output, suggesting firms' efforts may enhance the efficiency of production processes and practices. This paper also finds the mechanisms differ based on firm ownership: the effects on firm exit and process innovation are mainly led by private firms, while state-owned enterprises (SOE) may have benefited from political protectionism given the close relationships between SOEs and local governments.

While it is aimed to narrow the knowledge gap on the long-standing environmental regulation and firm-level productivity literature from a developing country perspective, the third DPhil paper adds new evidence that environmental regulation has enhanced firm productivity in the Chinese context. The results suggest that intervention aiming to reduce pollution may be an effective tool for fostering synergies between economic competitiveness and environmental quality. The findings on the firm ownership heterogeneity also imply that political favouritism may hinder the potential productivity increase that could be gained through enforcing environmental regulations.

1.5 Thesis structure

This DPhil thesis consists of five chapters. Chapter 1 introduces the background and motivations of this DPhil research, pinpoints research topics and methods, and presents the main results. Chapter 2 reviews the literature systematically in three specific areas, i.e. environmental issues and economic development, the evaluation of environmental policies, and environmental policies in the Chinese context. Chapters

3 to 5 present the three research papers. Chapter 6 discusses the contributions and limitations of this DPhil thesis, presents real-world implications and future research priorities. Finally, Chapter 7 concludes.

2 Literature Review

The literature chapter gives an overview of the major themes underlying the context discussed throughout the thesis. More detailed literature review concerning each research question can be found in each of the substantive papers (Chapters 3-5). This chapter aims to zoom out from the focus of each study and present the wider concepts that tie the papers together.

The literature chapter begins with an introduction of the interactions between environmental issues and economic development, with a particular focus on pollution and infrastructure. It also introduces briefly the environmental movements triggered by public concerns on environmental issues. This chapter proceeds with a review of the state-of-art evaluation of environmental policies, from the empirical estimation of policy effects on economic outcomes to scenario analysis for understanding climate change. Then this chapter introduces environmental policy literature in China's context, highlighting the limited understanding of environmental regulations in the developing world. Finally, it concludes with a summary of the literature review and points out the research gaps.

2.1 Environmental issues and economic development

2.1.1 The relationship between economic growth and environmental quality

The natural environment plays an important role in shaping our economy, however, over-exploitation of natural resources may result in the degradation of ecosystems. Understanding the relationship between environmental quality and economic growth is essential to achieving a green growth path. The Environmental Kuznets

Curve (EKC) is frequently used to describe the inverted-U relationship between pollution level and economic development (Kuznets, 1955). As illustrated in Figure 2.1, the environmental quality degrades as GDP per capita rises within the turning point. In this stage, the society concentrates on expanding production to meet consumption needs, regardless of how the growth is realised. After the turning point, increases in GDP per capita lead to decreases in environmental damage. This could be explained by technological progress and behaviour change, as the production process could become more efficient and individuals pay more attention to the quality of life.

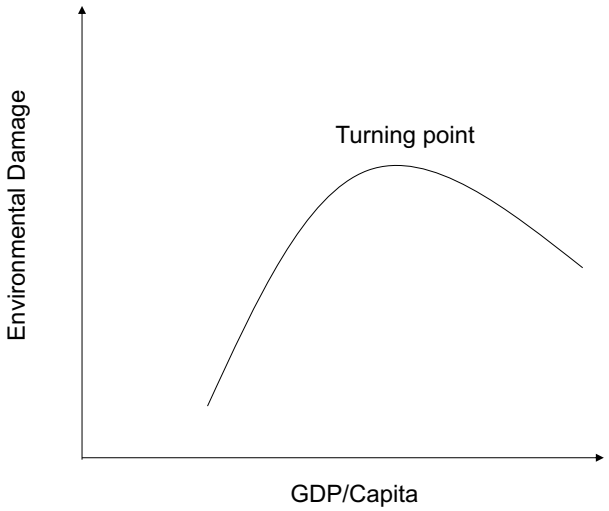


Figure 2.1: Environmental Kuznets Curve

Various studies tend to test the development-pollution dynamic theories empirically. By examining urban growth and the environment, Kahn (2007) shows how the EKC could explain the dynamics of some pollutants such as air and noise whilst not apply to other forms of environmental damage such as deforestation and land deterioration. Gürlük (2009) undertakes an EKC fit analysis for 15 Mediterranean countries and find that all countries apart from France follow a logarithmic increas-

ing or inverse-logarithmic function between the environmental damage, measured by biological oxygen demand (BOD) and per capita income. By investigating the relationship between per capita CO_2 emissions and GDP for a panel of 25 OECD countries between 1980 and 2010, Jebli et al. (2016) finds evidence supporting the inverted U-shaped EKC hypothesis.

Nevertheless, the EKC hypothesis has been criticised for its relevance to policy-making. EKC analyses are frequently based on certain countries and for a limited set of pollutants, making the generalisation of the conclusions challenging. Several alternative theories also examine the relationship between economic development and environmental performance, such as the limits theory, race to the bottom and new toxics (Stern, 2004, Dasgupta et al., 2002). These various theories underscore the complexity and multi-dimension of the link between economic growth and the environment.

2.1.2 The effects of pollution on the economy

Among the literature studying environmental issues involved in economic development, pollution may have attracted the most attention. Pollutants are emitted from various artificial and natural sources, including industrial processes, electricity generation, agriculture, consumption activities, and have detrimental effects on natural, human and physical capital.

Pollution could impact natural capital, which in turn influences economic growth (Ali and Puppim De Oliveira, 2018). For example, Beier et al. (2017) and Caputo et al. (2017) find the potential economic value of recreational fisheries in the Adirondack lakes in the United States decline measurably with increased acid deposition. Miao et al. (2017) find ozone exposure in China could lead to approximately 15 tons crop yield loss of wheat, rice, maize and soybean in a single year, resulting in economic losses of 3.4 billion USD in 2006. These adverse effects are more pronounced in small regional economies relying on agricultural products.

In addition to the impacts on natural capital, pollution may exert important influences on human capital. Hanna and Oliva (2015) examine plant closures in Mexico and find that a one unit increase in SO_2 leads to 1.3 hours decrease in working hours, and the effect could be attributed to the pollution-health channel. Air pollutants have acute and chronic effects on human health, resulting in respiratory diseases and organ disorders such as asthmatic attacks and lung cancer (Kampa and Castanas, 2008). In the long term, health effects may translate into a reduction in the labour supply. Pollution could increase the infant mortality rate (Chay and Greenstone, 2003), lead to shorter life expectancy (Chen et al., 2013, Ebenstein et al., 2017) and cause premature deaths (Clancy et al., 2002).

Pollution could also result in workers' productivity loss. Agricultural workers are heavily exposed to outdoor pollution and suffer from productivity loss in both industrialised countries (Zivin and Neidell, 2012) and emerging economies (Aragón and Rud, 2016). He et al. (2019) find ambient $PM_{2.5}$ pollution has significant adverse effects on the productivity of industrial manufacturing workers in China. Apart from physical demanding occupations, high skilled workers in the service and public sectors can also experience productivity loss from pollution (Chang et al., 2019, Khanna et al., 2019). Concerns on productivity are not confined only to the outdoor environment. Palacios et al. (2021) conduct a systematic review of 42 studies in 10 countries across 3 continents and find that individuals exposed to indoor air quality inferior to the minimum standards have a higher chance of experiencing poorer levels of health, performance and productivity. Experimental economics shows that poor air quality in buildings can decrease the productivity of office workers by 6 to 9 percent (Wyon, 2004).

Pollution may ignite behaviour change as well. Early work has made it clear that there are strong associations between environmental concerns and participation in outdoor recreation activities (Dunlap and Heffernan, 1975, Geisler et al., 1977). Recent work has shown how pollution has increased migration away from polluted

cities, primarily for well-educated workers (Chen et al., 2017b, Khanna et al., 2019). Fan et al. (2021) examine the effect of perceived air pollution on commuting choice through a survey experiment and find individuals decrease active commuting and increase the use of automobile when receiving tailored air pollution exposure risk information.

Apart from natural and human capital, pollution's effects on physical capital are not to be neglected. Ou et al. (2021) find urban housing prices are adversely associated with $PM_{2.5}$ levels in China, particularly in densely populated megacities. Davis (2011) shows US neighbourhoods within 2 miles of fossil fuel power plants experience a 3% -7% decrease in housing values and rents compared to districts with similar housing and demographic characteristics. The effects are more pronounced for large-capacity power plants and communities located closer to the electricity site.

With regard to the types of pollution empirical studies have focused on, air pollution has received the most attention. This may be due to the sensibly perceptible characteristics of air pollutants and approachable access to data. Thanks to the development of portable and low-cost air pollution sensors, air quality monitoring stations could now provide data in near-real time at a high resolution (Snyder et al., 2013). Over the recent years, satellite data have been increasingly used in empirical studies as a proxy for air pollutant concentration (Fu et al., 2021, Wei et al., 2019). The officially reported monitoring data might be subject to manipulation (Ghanem and Zhang, 2014, Karplus et al., 2018). Therefore, using satellite-based data could provide a more accurate and subjective measurement of pollution levels. Commonly used air pollutant measurements include ozone (Miao et al., 2017), sulphur dioxide (Hanna and Oliva, 2015), total suspended particulates (Chay and Greenstone, 2003, Chen et al., 2013), and particulate matters (Chen et al., 2018b, He et al., 2019, Ou et al., 2021).

In addition to air pollution, water pollution (Chakraborti, 2016, He et al., 2020,

Zhang et al., 2018) and toxic pollution (e.g. mercury) (Outridge et al., 2018, Trasande et al., 2016) have received remarkable attention.

2.1.3 The impacts of infrastructure on the environment

Large infrastructure projects, such as electric power plants and transport routes, are frequently linked to biodiversity loss and environmental degradation. A large part of the scholarly research is conducted by biologists and conservation scientists, who can identify infrastructure projects' potential risks on fragile ecosystems (Ng et al., 2020, Yang et al., 2021) and examine specific species loss in high biodiversity forests (Caro et al., 2014). For example, Ng et al. (2020) conduct a Geographical Information System (GIS) analysis to assess the potential biodiversity effects of the China-backed Belt and Road Initiative in Southeast Asia. They find the road projects are intersected with 32 protected areas, 40 key biodiversity areas and 29 terrestrial ecoregions within 1km from the route, and rail projects may influence 142 threatened species in areas within 5km from the rail.

Economists could contribute to revealing the causal impacts of infrastructures. For example, Asher et al. (2020) use multiple causal identification strategies (i.e. regression discontinuity approach and difference-in-differences specification) to study the new road construction and highway upgrade in India and find that rural roads have precisely zero effect on local deforestation whilst highway upgrades lead to significant forest loss. Barrows et al. (2019) find a one GW increase in coal-fired capacity in India corresponds to a 14% increase in infant mortality rates in communities near the power plant site. Using an instrument based on historical subway planning and granular air quality data, Li et al. (2019) quantify the causal impact of Beijing's subway expansion on air quality. They find an increase in subway density by one standard deviation enhances air quality by two percent, and the discounted health benefit during 20 years could amount to 1.0-3.1 billion USD.

Over the recent years, along with increasingly pronounced climate change effects

and the perceived climate urgency, studies on carbon emissions released from fossil fuel infrastructures have been an active research field. Estimating carbon emissions from different assets has been a priority in climate studies, covering the electricity sector (Tang et al., 2019), carbon-intensive industries (Liu et al., 2015), transport (Tian et al., 2018), buildings (Chau et al., 2015) and agriculture (Tubiello et al., 2013). Tong et al. (2019) estimate that, if operated as historically, existing and proposed fossil fuel infrastructure will cumulatively emit about 846 gigatonnes of CO_2, more than the entire carbon budget that remains to limit global temperature rise to 1.5 degrees or two-thirds of the remaining 2 degrees carbon budget. More than half of these emissions would come from the electricity sector.

2.1.4 Public concerns and environmental movements

Since extracting and production activities put land, air, water, forests and livelihoods under increasingly visible threat, affected communities and civil society organisations worldwide have mounted several campaigns to express their concerns on the detrimental environmental impacts (Inter-American Development Bank, 2017, Temper et al., 2020). Environmental conflicts are defined as *"mobilisations by local communities, which might also include support of national or international networks against particular economic activities whereby environmental impacts are a key element of their grievances"* (Temper et al., 2015).

Until 2020, Global Atlas of Environmental Justice (EJAtlas), the most extensive publicly available database of environmental conflicts, has collected information on over 3200 global cases, ranging from peaceful demonstrations, open letters, petitions, protests to fierce resistance such as violent conflicts. Besides, if there are violations of local and international laws, environmental groups and individuals could take legal actions by filing lawsuits against government authorities, corporations and individuals. By 2020, hundreds of climate change lawsuits have been recorded in the Climate Change Litigation Database, one of the most comprehensive

databases collecting climate change lawsuits in and outside of the United States.

Environmental movements' main actors include non-governmental organisations (NGOs) such as Greenpeace and 350.org, loose coalitions such as the Fridays for Future student strikes and Extinction Rebellion, and local mobilisation groups (Temper et al., 2020). NGOs play an important role in disseminating environmental opinions and formulating environmental policies, particularly in emerging countries (King and Mori, 2007). When international NGOs launched their branches in the developing world in the early 1980s, local grass-root environmental organisations appeared and became active. These NGOs propagate the latest environmental practices in developed countries to emerging economies and organise mobilisations against local ecological issues.

The information and communication revolution has helped amplify the effect of environmental movements as the voice could be easily spread through the Internet (King and Mori, 2007). This allows environmental protests that happened in New Delhi today to be known in London tomorrow. Media news facilitates the cross-boundary diffusion of environmental protests' ideas and tactics (Lester and Hutchins, 2009). A prominent example is the Fridays for Future climate strike, a youth-led movement to protest against the lack of action on climate change. The strike action started by Greta Thunberg in Sweden in 2018 soon became a movement that swept the world. Another example is France's yellow-jacket protest, which was initially ignited by President Macron's fuel tax, and further developed into nationwide movements appealing for energy justice.

Environmental movements have gained success both in established countries and emerging economies, but to varying degrees. Using data of environmental protests and public opinions in the US from 1960 to 1998, Agnone (2007) finds that public opinion is more likely to impact environmental policy outcomes with protest. That is, more environmental laws are passed when there is the presence of protests. Among all the environmental conflict cases that the EJAtlas database

has mapped, approximately 18% are qualified as "success", which means the court cases were won, laws or legislation were changed, citizen participation was increased in decision-making processes, or institutional changes were made for more equitable outcomes (Temper et al., 2015). Nevertheless, it is worth mentioning that most of the claims in climate change lawsuits have been unsuccessful (Stuart-Smith et al., 2021). These shreds provide mixed evidence in understanding the effectiveness of environmental movements.

2.2 Evaluating the effects of environmental policies

2.2.1 The need for policy evaluation

Environmental policies are used to address market failures associated with natural environment's characteristics. They aim to be cost-effective, i.e. achieving a given pollution reduction target at the least cost (Aldy et al., 2003). Nevertheless, ever since the first sets of environmental regulations were enacted in the 1970s, concerns about their economic impacts have never faded. Indeed, some computable general equilibrium (CGE) models indicate that in response to unilateral climate policy, the output, exports, and employment of emission-intensive trade-exposed sectors will decrease, gross welfare levels will decline, and there will be carbon emission leakage from regulated countries to other countries (Carbone and Rivers, 2017). Firms in regulated industries have to invest in end-of-pipe technologies to control pollution, and these investments were often seen as an extra cost (Cohen et al., 1995). Upstream industries of the value chain, such as resource extraction and primary manufacturing, face a competitive disadvantage as the environmental impacts of their production activities are disproportionate to the value-added to their products (Clift and Wright, 2000).

Yet environmental regulations could stimulate investments that enhance efficiency and induce clean innovation. When firms face the additional costs imposed by stringent environmental standards, they have an incentive to invest in new technologies and production processes that could improve resource productivity and ultimately reduce production costs. This is often referred to as the "Porter hypothesis" (Porter and Van der Linde, 1995). There are two forms of "Porter hypothesis". The weak form argues that environmental regulations could induce cleantech innovation, such as firms increasing R&D and patenting in environmentally-friendly technologies (Aghion et al., 2016, Brunnermeier and Cohen, 2003, Calel and Dechezleprêtre, 2016, Jaffe and Palmer, 1997). The strong form emphasises that induced innovation could be further translated into productivity gains. For example, innovative firms could gain "first-mover advantages" by selling their innovative products to other firms. From a dynamic long-term perspective, such innovation abilities could improve the competitiveness of a country as a whole (Esty and Porter, 1998).

With these two contrasting views, policy evaluation is needed to fully understand the impacts and effectiveness of the enforced regulations and provide feedback to improve future policy design and implementation (Holmes and Clark, 2008). Policy evaluation is essential for improving public governance through promoting public accountability, increasing public sector effectiveness and enhancing decision-making (OECD, 2020). Using rigorously established objective evidence to inform policymakers is frequently referred to as evidence-based policymaking (Sanderson, 2002). As opposed to opinion-based decision making (i.e. the opinions and judgements of experts), evidence-based policymaking is a more rigorous and rational approach that gathers the best available research evidence to inform policymakers (Sutcliffe, 2005). Evidence used to aid decision-making requires three attributes: credibility (the proof is assessed in a rigorous approach and went through a robust peer-review process), salience (the information provided is addressing the asked policy questions) and legitimacy (the knowledge is produced in a fair manner and considered

the values and needs of different parties) (Cash et al., 2002).

2.2.2 The enforcement of regulations

Before understanding whether the environmental policies have led to economic impacts, it is vital to ensure the regulation was enforced and polluting agents have complied. A central focus of environmental regulation's effectiveness is their effects on reducing pollutant levels. For example, as one of the most comprehensive national regulations to address air pollutants, the US Clean Air Act has been found to significantly reduce total suspended particulates, sulphur dioxide, ozone and other pollutants levels, though to varying degrees (Chay and Greenstone, 2003, Greenstone, 2004, 2003, McKitrick, 2007). It is worth noting that not all regulations are effective in reaching environmental targets. Betz and Sato (2006) find that the effectiveness and efficiency of carbon trading markets (i.e. EU Emission Trading Scheme) depend on government decisions such as allowance plans, auction design and free allocation.

Environmental regulations may even lead to unintended consequences. There could be a spatial spillover of pollutants from regulated areas to surrounding regions when stringent policies are in place. Feng et al. (2020) find that environmental regulations in surrounding cities could increase the $PM_{2.5}$ concentration in urban agglomerations in China by 0.11-0.76 percent. Meanwhile, reductions in air pollutants may be due to the use of abatement devices such as scrubbers and electrostatic precipitators, and these devices could release residuals into water and land. To test such an effect, Greenstone (2003) studies the cross-media release of the US Clean Air Act and find little evidence that the regulation has increased releases into waterways and the ground.

Heterogeneity exists among facilities with different scales and locations when the regulation was enforced. Larger firms would be more likely to comply and face more significant costs in the US context (Earnhart, 2009, Gray and Shadbegian,

2005, Hanna and Oliva, 2010). Comparing the upstream and downstream agents in waterways in China, Cai et al. (2016a) find that most downstream counties have more water-polluting activities while the enforcement of pollution control regulations is more lenient than upstream counties, suggesting local governments respond to the pollution reduction mandates by shifting enforcement efforts away from the most downstream counties. Similarly, He et al. (2020) find that local governments enforce stricter regulations on firms upstream rather than downstream monitoring stations.

2.2.3 Policy effects on economic outcomes

The impacts of regulations on affected entities' economic performance have become an important issue in environmental policy debates. When facing stringent standards, firms' investment in pollution control may divert resources away from production activities and thus hamper productivity growth. On the other hand, compliance with regulations may lower firms' marginal production costs and enhance firms' productivity and competitiveness.

Early empirical studies did find evidence that environmental regulations hamper productivity (Gollop and Roberts, 1983, Gray and Shadbegian, 2003), though these studies either use small samples or explore only the short-term impact. More recent studies suggest the effects vary across pollutants and industries and sometimes could be positive. For example, Greenstone et al. (2012) find that while regulations governing ozone levels have led to productivity decrease, carbon monoxide regulations could positively impact productivity. Fleishman et al. (2009) explore the changes in the efficiency of US power plants over the period 1994-2004 and find more stringent SO_2 regulations negatively affect plant efficiency whilst NO_x regulations appear to have no significant impact on coal plants but a negative effect on gas plants. More recently, Bailey (2019) finds coal power plants in the US Regional Greenhouse Gas Initiative have become less efficient due to operating at low output rates. This also

leads to a 4.56% carbon emission rebound effect, i.e. regulated coal power plants achieved 4.56% less CO_2 reduction compared to the counterfactual condition (if the efficiency level did not change).

The potential impact of environmental regulation providing incentives for technological innovation has seen increasing popularity. Many studies have shown that environmental regulations could induce the development of clean technologies, frequently measured by R&D expenditure and environment-related patents (Aghion et al., 2016, Calel and Dechezleprêtre, 2016, Jaffe and Palmer, 1997, Popp, D., 2002). New technologies may substantially reduce the cost of pollution abatement (Jaffe et al., 2003). This is the weaker form of the "Porter hypothesis" , compared to a strong version that argues regulation-induced innovation could offset the cost of compliance and increase firms' competitiveness (Ambec et al., 2013). A recent review paper, Dechezleprêtre and Sato (2017), concludes that there is missing evidence supporting the strong version of "Porter hypothesis", i.e. environmental regulations can enhance firms' competitiveness through their effects on innovation.

The effects on employment are also of great interest when scholars explore the economic impacts of environmental regulations. Most empirical studies find no evidence of job loss or statistically significant but relatively small negative employment effect (Ferris et al., 2014, Gray et al., 2014, Greenstone, 2002, Liu et al., 2017). Some studies even find a slight net increase in employment caused by environmental regulations (Berman and Bui, 2001, Morgenstern et al., 2002). For example, Cai et al. (2011) find that pollution mitigation policies in China's power generation sector from 2006 to 2009 have caused 44 thousand net job losses whilst the renewable energy policies from 2006 to 2010 have created 472 thousand net job gains. On average, they find for every 1% increase in the share of solar PV generation, there is a 0.68% increase in total employment in China's power generation sector.

Environmental regulations could also affect trade. Hering and Poncet (2014) find a fall in sectoral exports in targeted cities after the implementing regulation on

sulphur dioxide emissions in China. Similarly, Cherniwchan and Najjar (2020) find a Canadian air quality standard regulation reduced manufacturing plants' export volumes by 30% and increased the likelihood plants stop exporting by 5%. Branger et al. (2017) test whether carbon prices of EU ETS increased EU imports of cement and steel for the period 2004-2012 and find very limited evidence.

A more concrete example of how environmental policies may affect economic activities is the issue of stranded assets, which are defined as *"fossil fuel supply and generation resources which, at some time prior to the end of their economic life (as assumed at the investment decision point), are no longer able to earn an economic return (i.e. meet the company's internal rate of return), as a result of changes associated with the transition to a low-carbon economy"* (Carbon Tracker Initiative, 2017). If we are to have a reasonable chance of limiting global warming to 2 degrees, about 80% of existing coal reserves, half of the gas reserves and a third of oil reserves should remain unused (McGlade and Ekins, 2015). Global CO_2-emitting energy infrastructure amounts to 22 trillion USD (Tong et al., 2019). Stranding of these fossil fuel assets will have far-reaching impacts on the financial markets. Mercure et al. (2018) estimate that the magnitude of deflated global wealth due to stranded assets loss could be between 1–4 trillion USD, comparable to the sum that triggered the 2008 financial crisis (250 billion USD). When the world moves towards net-zero, fossil fuel companies that own coal mines, oilfields and gas reserves will find themselves in a difficult situation. So will the bankers, insurers and investors that are lending to, underwriting and investing in fossil fuel assets. The stranding risk is system-wide: automobile, aviation, steel, cement and other industries that use fossil fuels as inputs or are carbon-intensive are all impacted.

2.2.4 The Pollution Haven hypothesis

There is an overarching international dimension of environmental regulation impact on the global economy, in the context of relocation and foreign direct investment.

Much of this work is motivated by the "pollution haven hypothesis", which asserts that, following the increasing stringency of environmental regulations, pollution-intensive industries seek to avoid the pollution abatement cost by expanding or shifting production to countries where environmental norms are laxer (McGuire, 1982). This may lead to a pollution leakage effect, particularly for carbon emissions (Babiker, 2005). A necessary condition for the hypothesis to hold is that the regulation has translated into significant differences in production costs (Cherniwchan and Najjar, 2020).

A number of studies have empirically tested the hypothesis. List et al. (2003) use US county-level panel data from 1980 to 1990 and find a strictly regulated "nonattainment" county of the Clean Air Act Amendments decreased the inflow of plants relocation by 63 percent versus an "attainment" county. By using the same data, List et al. (2004) further find that the "nonattainment" status deceased the expected inflow of new pollution-intensive plants by 41-61 percent. In contrast, Raspiller and Riedinger (2008) and Ben Kheder and Zugravu (2012) do not find much significant evidence that French firms' foreign location choices are impacted by environmental regulation stringency.

Lax policies may attract more inward manufacturing investments, and stringent home policies may drive more outward investment flows. Dean et al. (2009) find that more equity joint ventures in highly polluting industries funded through Hong Kong, Macao and Taiwan were attracted to mainland China by weak environmental standards. Eskeland and Harrison (2003) test whether US multinationals flock to developing countries where environmental governance is relaxed (i.e. Mexico, Venezuela, Morocco and Cote d'Ivoire). They find that US investors tend to locate in sectors with high levels of air pollution in host countries, though US-owned plants are significantly more energy-efficient and less polluting than their domestic counterparts. Using a panel dataset of US-based multinationals over 1966-1999, Hanna (2010) finds the US Clean Air Act amendments lead to a 5.3 and 9 percent

increase in multinationals' foreign assets and output, respectively.

2.2.5 State-of-art empirical evaluation

Overall, there is no consensus in empirical studies on the economic effects of environmental regulations. Most of the studies are context-specific, impeding the generalisation of conclusions. Early empirical review papers concluded that little evidence could support argument that environmental policies have considerable adverse effects on economic competitiveness (Jaffe et al., 1995). More recent empirical reviews find statistically significant negative impacts on trade, employment, location and productivity in the short run, particularly in high-polluting and energy-intensive sectors. Nevertheless, the scale of these impacts is small relative to other determinants of trade and investment choices. Small and statistically significant positive impacts are found in some sectors, but very little evidence so far has found environmental policy has strong positive impacts on firm performance (Dechezleprêtre and Sato, 2017).

Over the recent years, thanks to more robust research design and advanced causality inference methods, there is a more granular understanding of the effects of environmental regulations in various aspects. In addition to answering the "yes-or-no" question, most recent studies take a step further to examine details of the impact, from investigating the duration and heterogeneity to unpacking the mechanism of effects. For example, studies frequently find heterogeneity in firm ownership (i.e. state-owned firms versus private firms) and size (Dechezleprêtre and Sato, 2017). There are cross-sectoral differences as well, as pollution-intensive sectors are affected differently relative to less pollution-intensive sectors (Barwick et al., 2018, Hering and Poncet, 2014, Liu et al., 2017). Regulation effects may also differ in the short-term and long run (Kahn and Mansur, 2013, Yamazaki, 2017).

In terms of policy institutions investigated, some studies use environmental spending or energy price to measure environmental policy stringency (Deschenes,

2011, Kahn and Mansur, 2013, Morgenstern et al., 2002). Other studies investigate the impact of particular institutional shocks. The environmental policies that have attracted the most attention are United States Clean Air Act (Currie and Walker, 2019) and European Union's carbon trading market, i.e. EU ETS (Dechezleprêtre and Kruse, 2018). Researchers also explore a range of policies in other parts of the world, such as China's air pollution control regulations (Liu et al., 2021) and Canada's carbon tax (Yamazaki, 2017).

2.2.6 Scenario analysis for understanding climate change

While econometric methods are used to empirically evaluate environmental poclies' impacts on economic outcomes, scenario analysis is frequently used to guide climate policymaking. Integrated Assessment Models (IAMs) are the most widely used approach to inform policymakers about questions related to climate change. These complex models are a suite of tools developed by scientists and economists to inform decision-makers on the best ways to tackle anthropogenic climate change in the most cost-effective approach. IPCC reports, which are considered the most authoritative scientific publications on climate change, draw substantially on IAMs to understand the climate change causes, potential impacts and response options (Howarth and Painter, 2016).

IAM models use simplified and stylised numerical approaches to represent extremely complex physical and social systems (IPCC, 2014). They combine different strands of knowledge from physical laws to societal choices to model human development alongside the earth system. As illustrated in Figure 2.2, IAMs use linked modules representing different sectors of the global economy, such as energy, land use, agricultural and climate systems, to capture energy demand and socioeconomic behaviour (Engström et al., 2017; van Vuuren et al., 2011). They produce scenarios that indicate the required development of different sectors, such as how the world's energy system needs to change to achieve climate targets.

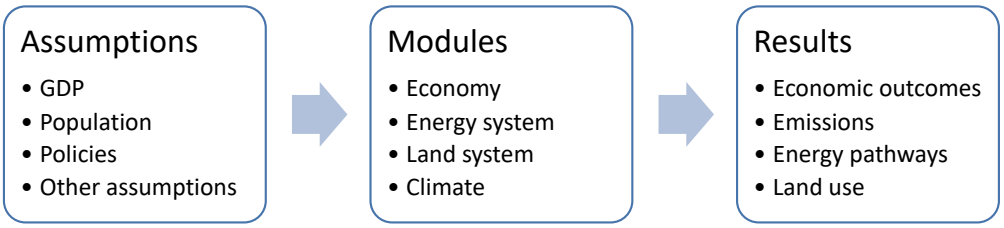


Figure 2.2: Overview of the Integrated Assessment Modelling framework

IAM modellers usually produce two sets of scenarios: the baseline scenarios to show what would happen if the business continues as usual; the technology and policy scenarios to investigate how the world would need to change if we are to limit global warming to 1.5 or 2 degrees. This reflects two broad types of questions frequently asked by policymakers: firstly understanding the scale of the climate risk and then exploring options to avoid catastrophic impacts. Particularly, IAMs model technological changes that reflect the cost of technologies and their potential deployment. By setting diverse levels of technology assumptions, IAMs produce different technology scenarios to show the pathway of meeting the climate goal with and without a particular technology. For example, carbon capture and storage (CCS) and a range of negative emission technologies (e.g. Bioenergy with CCS, i.e. BECCS) are heavily used in decarbonization scenarios. However, modelling the technology deployment does not necessarily mean these technologies are imperative for reaching the climate target. Rather than acknowledging specific technology requirements for achieving climate goals, IAMs could help decision-makers compare the trade-offs between different technology choices.

Despite the benefits for guiding policymaking, IAM models have been long criticised for the way they handle uncertainty, relative prices, behaviour and other factors (Farmer et al., 2015, Pindyck, 2013, Stern, 2013). IAM projections are involved with significant uncertainties as predicting the future is highly uncertain, especially when it comes to modelling socioeconomic norms and behaviours. IAMs tend to provide divergent results due to their different model settings (Kriegler et al., 2015, Riahi et al., 2015) and these differences are useful for illustrating the wide range of uncertainties and sensitivities.

2.3 Environmental policies and the Chinese economy

2.3.1 Limited understanding of developing countries

Much of what we know about the impact of environmental regulation on the economy is derived from developed countries, such as Clean Air Act in the United States and the carbon trading market in European Union. The lack of understanding of the developing world mainly stems from two factors. Firstly, governments in the developing world generally lag behind developed economies in setting environmental standards. In the late 1960s and 1970s, when emerging issues—the discovery of dangerous pesticide use in agriculture (Carson, 1962), mercury poisoning from industries (Takeuchi et al., 1962), the increase of asthma cases due to heavy air pollution (Cohen et al., 1972), and others — prompted governments to set up environmental agencies and introduce the first sets of environmental policies, most of these practices were imposed in developed countries such as the United States, Europe and Japan. In the early years, developing countries' adoption of environmental policies was often due to joining international agreements (King and Mori, 2007). When environmental problems became more widely recognised in the 1980s, de-

veloping countries realised the management of natural resources in the agriculture, forestry and fishery sectors were equally important as containing industrial pollution. Governors in the developing world started to adopt environmental principles that were applied in developed countries (King and Mori, 2007)..

Secondly, the lack of enough and reliable data hinders scholars from evaluating environmental policies in developing countries. Data and statistics provide the essential basis for understanding policy effectiveness and analysing policy impacts. Still, the quality, availability and timeliness of economic and environmental data are insufficient in many parts of the developing world (OECD, 2017). Thanks to the efforts of national and international organisations, macro-level data on growth, population, health, education, and environmental resources are increasingly available in many digital public databases maintained by international organisations such as the UNDP, World Bank, and OECD. Nevertheless, most of these data are aggregated at the country level, allowing for national or cross-country studies. The more granular micro-level data that enable in-depth sub-national studies such as the firms or facilities are still lacking.

It is particularly salient for emerging economies to understand the impact of environmental regulations as these countries face serious pollution problems that threaten human health while achieving strong economic growth. Given the different incentive structures, bureaucratic norms, and policy settings, estimates in developed economies may not be externally valid to the developing country context (Greenstone and Jack, 2015). Simply transferring the understanding of environmental policies from established economies to emerging economies is problematic as people of different cultures and socioeconomic conditions may not have identical behavioural responses to the same set of incentives (King and Mori, 2007). For example, when Arceo et al. (2016) compare the effect of air pollution on infant mortality between developing and developed economies, they find the impact of PM_{10} in Mexico tend to be similar to the US estimates, but the effects for CO are

larger than those based on US context.

While still lacking, the availability and quality of micro-level data from developing countries have seen a vast increase in recent years. Many new data sources are emerging through the global information and communication revolution, such as lexical data extracted from social media and geospatial night light data from satellite images. These big data provide high frequent and granular snapshots from a wide range of angles to fill critical information gaps in understanding emerging economies.

2.3.2 Evaluating environmental policies in China

There has been a particular focus on environmental policies in China over recent years. As one of the world's largest economies, China has been confronted with many environmental challenges. China has issued a considerable number of regulations to improve environmental quality over the recent years, including market-based policies, command-and-control regulations, and other instrument tools. Date back to 1978, China revised the Constitution and placed "environmental protection" as one of the primary national policies. It was followed by a provisional Environmental Protection Law in 1979, signalling the priority to build an environmental protection legal framework in the following years (Child et al., 2007).

Take air pollution control as an example. The earliest air pollution containing measures in China could be traced back to the 1980s when China suffered from heavy black smoke from stacks and severe acid rain. China issued the Atmospheric Pollution Prevention and Control Law in 1987 and amended it in 1995, 2000, 2015 and 2018 with increasing stringency. In the early years (from 1980 to 2010), the regulatory efforts were mainly focused on reducing sulphur dioxide to mitigate acid rain. In recent years, regional air pollution problems dominated by fine particulate matter ($PM_{2.5}$) and ground-level ozone (O_3) have emerged and worsened. After the salient winter-long haze in eastern China in 2013, air pollution has become a top

priority among the government's well-being objectives (Jin et al., 2016). The Chinese government has implemented a suite of policy tools to improve air quality, such as shutting down high polluting firms, imposing driving restrictions, setting technology standards, linking local officials' promotion to pollution emission reduction targets, and direct investments in clean technologies (Zheng and Kahn, 2017).

Meanwhile, compared to other developing economies, China has several micro-level datasets available to researchers. For example, the China Industrial Enterprise Database (CIED), maintained through annual surveys conducted by China's National Bureau of Statistics, provides yearly data of all Chinese state-owned enterprises (SOEs) and private firms with annual sales exceeding 5 million Chinese yuan (Brandt et al., 2014). Another example is the China Stock Market and Accounting Research Database (CSMAR), which keeps annual records of more than 3000 Chinese public-listed companies' financial and employment data. In official statistical yearbooks, Chinese government also incorporates several environmental indicators such as pollutant emissions and concentrations.

Accordingly, investigating the economic effects of Chinese environmental regulations has been of growing interest. Some have studied how pollution impacts labour demand (Gray et al., 2014, Liu et al., 2017, 2018, 2021), others focus on firms' exports (Hering and Poncet, 2014) and inward foreign direct investment (Bu et al., 2019, Cai et al., 2016b). There's been a particular focus on productivity. Fu et al. (2021) use thermal inversion as an instrument and find that one unit decrease in $PM_{2.5}$ raises the productivity of Chinese manufacturing firms by 0.82%. Chang et al. (2019) and Kahn and Li (2020) reveal air pollution impacts not only outdoor workers but also highly skilled workers indoors from the study of Chinese judges and two call centres in China. He et al. (2020) exploit the discontinuity in regulation stringency between firms located immediately upstream and downstream of water monitoring stations and find immediate upstream polluters have higher total factor productivity reduction than their downstream counterparts.

2.3.3 Growing interest in Chinese overseas investments

Most literature studying foreign direct investment under developing country context concentrate on the inward foreign direct investment (FDI) from established economies to emerging markets (Bu et al., 2019, Cai et al., 2016b, Huang and Lanz, 2017, Jiang et al., 2018, Liang, 2017). This is because the central claim in conventional literature on the "pollution haven" hypothesis is that foreign investors from developed countries are attracted by weaker environment regulations in developing countries (Dean et al., 2009, Eskeland and Harrison, 2003). Over recent years, with the economic growth in emerging economies, many developing countries have become significant sources of foreign capital outflows and have gained popularity in scholarly research, such as South Korea (Chung, 2014), India (Hattari and Rajan, 2010) and China (Amighini et al., 2013, Ramasamy et al., 2012). Outflows from Asia represent 40% of global outward FDI in 2018, when China and Japan are the world's largest FDI source economies (UNCTAD, 2020).

It is essential to understand these new trends not only because of their dramatic increase but also due to developing countries' firms' different internationalisation strategies. Overseas investments of multinationals from the developing world seem to differentiate with the conventional theories of internationalisation. For example, the mainstream theory assumes that firms internationalise to exploit competitive advantages, while Chinese firms are found to be aimed at addressing competitive disadvantages when they invest overseas (Child and Rodrigues, 2005).

No other emerging economy has received as much attention as China. With the launch of the Belt Road Initiative, FDI outflows from China have been increasing rapidly. The total outward FDI from China increased from 28.6 billion USD in 2003 to 183 billion USD in 2016, making China the largest FDI source economy in the world in 2017 and 2018 (UNCTAD, 2020). Notably, many Chinese overseas investments are located in middle- and low-income countries spanning around Southeast

Asia, South America and Africa. In the 1990s, Canada, United States and Australia hosted about 40% of Chinese outward FDI, and the proportion had decreased to 10% in 2005. Meanwhile, emerging economies, particularly Asia and South America, accounted for nearly 90% of Chinese outward FDI in 2005 (Ramasamy et al., 2012).

With the rapid increase, Chinese overseas investments have been in the limelight of global environmental discussions. Spotlights have focused on carbon emissions, environmental footprints, and biodiversity impacts (Yang et al., 2021, Ascensão et al., 2018, Narain et al., 2020). While most conventional multilateral development banks gradually reduced financing for carbon-intensive coal-fired power, Chinese together with Korean and Japanese investors have become the leading public financiers of global coal power plants. Concerns on coal financing centred on its climate effects and air pollution (Chen et al., 2020, Radford et al., 2021). China has become one of the world's largest lenders in overseas development finance, and Yang et al. (2021) find 63% of China-financed infrastructure projects between 2008 and 2019 overlap with critical habitats, protected areas or indigenous lands. With Chinese companies becoming the largest dam builders globally, International Rivers (2012) finds some of these projects are being built in areas with critical ecological values, such as the Myitsone hydro power project in Myanmar and the Belinga project in Gabon.

2.4 Summary of literature

The literature chapter highlights some of the overarching themes that bind the three papers of this thesis in a unified purpose: to narrow down our knowledge gap of the interactions between environment and economy. To this end, this chapter briefly reviews several bodies of empirical literature in environment-economy links, including pollution concerns and infrastructure projects, climate constraint

and stranded assets, and environmental policy's effects on key economic measures. This chapter also includes an overview of empirical investigations of environmental issues in China's domestic and international context. These serve as a foundation for spelling out the more salient literature and particular knowledge gaps in the following substantial chapters.

Precisely, previous literature has identified environmental concerns as one of the major risks to economic development, but ex-post research on whether environmental risks could impact a project's successful implementation is limited. Chapter 3 provides new evidence from the cancellation and postponement of Chinese overseas power plants. Additionally, although the burgeoning climate literature has highlighted the stranding risk of fossil fuel assets, there still lacks understanding on solutions to mitigate the risk. Chapter 4 explores this critical question by estimating the role of plant conversions and technological progress (such as carbon capture and storage and bioenergy deployment) in reducing the stranding risk. Lastly, most empirical evidence on environmental regulations and firm competitiveness is drawn from developed countries' contexts. Furthermore, there is limited understanding of how regulations may translate into productivity gains. Chapter 5 adds novel evidence from China and explores the possible mechanisms that may explain the productivity changes.

3 Environmental risk and power plant suspension

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Abstract

Power sector investment is crucial to accelerate a sustainable energy transition, but not all investments are successful. To shed light on investment trends, we examine 1393 Chinese overseas electric power projects over the past two decades. We identify approximately 5% have been cancelled or delayed, with coal and hydro projects having a much higher failure rate than solar and wind projects. We find the suspension is associated with technology-specific environmental risks. Coal projects located in more densely populated areas where there are more people exposed to air pollutants, in countries with more fatalities from extreme weather events, and in places with the presence of environmental protests, are more likely to be suspended. Additionally, hydro projects closer to protected areas have a higher suspension rate. Our results suggest restraining from investing in environmentally risky projects is not just to mitigate environmental losses, but also financial losses incurred due to cancellation and postponement.

Key words: Environmental risks; Chinese overseas investment; Coal power plants; Hydro power plants; Project suspension.

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3.1 Introduction

Greater investment in the energy sector is crucial for building a global energy system capable of delivering several UN Sustainable Development Goals (Griggs et al., 2013). As of 2019, 770 million people (10% of the global population) still do not have access to electricity, and the COVID-19 pandemic may significantly slow progress to achieving global electricity access (IEA, 2020b). Meanwhile, decarbonising the global energy sector is crucial to achieve net zero by 2050 (IEA, 2021). In recent years, China-funded power projects have received considerable attention both for their contribution to energy accessibility in less developed countries (Falchetta et al., 2021) and their impacts on climate change (Shearer and Buckley, 2019, Springer et al., 2021). China has emerged as one of the world's largest financiers in the global electric power sector, with two Chinese policy banks—the China Development Bank and Export-Import Bank of China—providing a total of 245.8 billion USD in energy finance to more than 80 countries since 2000 (Gallagher, 2021).

The fate of China's overseas energy investments, however, is not always secure. In total, about 65 billion USD of Chinese-backed coal-fired power plants have been shelved, mothballed or cancelled, and many more projects have been significantly delayed (CREA, 2021, Nedopil Wang, 2021). For example, in February 2020, the 6.6 gigawatt Hamrawein coal power proposal in Egypt was postponed indefinitely, which would have been the world's second-largest coal-fired plant (Nicholas, 2020). Despite a large number of studies applying ex-ante frameworks to identify potential risks that energy investments could face (Schmidt, 2014, Tietjen et al., 2016, Duan et al., 2018, Islam et al., 2019, Yuan et al., 2019, Egli, 2020), there is a lack of ex-post research on whether risk factors could impact a power project's successful implementation.

Environmental issues have been at the heart of the discussion on Chinese overseas investments, driven by concerns about increased carbon emissions, environmen-

tal degradation, and biodiversity loss (Ascensão et al., 2018, Narain et al., 2020, Yang et al., 2021). The overseas coal power plants with partial or full financing from Chinese development finance institutions, if successfully in operation, will emit 11.8 Gt CO_2 over their lifetime, as much as the annual emissions from global operational coal-fired power plants in 2018 (Chen et al., 2020). Coal projects may also expose local communities to air pollution (Barrows et al., 2019, Radford et al., 2021). Furthermore, Chinese companies have become the largest dam builders in the world, and some of their projects have been located in remote areas that have important ecological values (International Rivers, 2012).

Recent studies have revealed that environmental issues could trigger public resistance against energy projects (Inter-American Development Bank, 2017, Coppens et al., 2018, Temper et al., 2020). There have been cases of electric power projects facing opposition from local communities and environmental groups, and some campaigns have gained success (Renaldi, 2021, Yi, 2021). For example, after a long-lasting demonstration from local groups and non-government organisations (NGO) against Lamu coal power project in Kenya over its environmental impacts, a court ruling suspended the project's permit in 2019, and the project was officially cancelled by the government in 2020 (Yi, 2021). While scholars continue to enrich our understanding of the characteristics of environmental social movements (Gadgil and Guha, 1994, Agnone, 2007, Temper et al., 2020), the potential role of these mobilisations in altering the trajectory of power projects remains unclear.

In this study, we examine overseas power plant investments by Chinese firms that have been suspended (i.e. cancelled or delayed) over the last two decades. We trace the status of Chinese firms' overseas investments in power plants across power generation technologies, ranging from coal, hydro, wind, and solar, compare the differences between the suspended and progressed plants, and examine the association between suspension and factors of environmental risk. Of the 1393 overseas power investments by Chinese firms from 1997 to 2020, we identify 75 (5%) that have

been cancelled or postponed. Using survival analysis, we find that coal and hydro power plants are more likely to be deferred or shelved than other types of plants. Coal power projects have a higher chance of being suspended if they are in densely populated areas, where more people are exposed to pollution, or in countries that are more vulnerable to climate change. Meanwhile, the closer hydro power projects are to protected areas, the more likely they are to be suspended. Additionally, the presence of environmental protests near the power project increases the likelihood of suspension for coal projects, but not for hydro projects.

3.2 Mapping Chinese overseas power projects

By combining several data sources, we construct a dataset of 1393 power units with investments by Chinese firms in 78 host countries from 1997 to 2020 (Figure 3.1 (a)). Chinese firms invest in all types of power units across the globe. Hydro power has received the most investment, followed by wind and solar. One-third of Chinese firms' investments are in fossil fuels. Chinese firms also invest in a small number of newer and less frequently applied technologies overseas, such as biomass (including waste incinerators), geothermal, and nuclear power.

Tracing back project status to the investment year, we identify 75 units, roughly 5% of the sample, that have been suspended at some point. In this paper, suspension refers to a power project being cancelled or delayed at some point in its development. The majority of these suspended units are located in emerging economies, such as Argentina and India, though Chinese firms also experience failures in high-income countries like Australia and Norway (Figure 3.1 (b)).

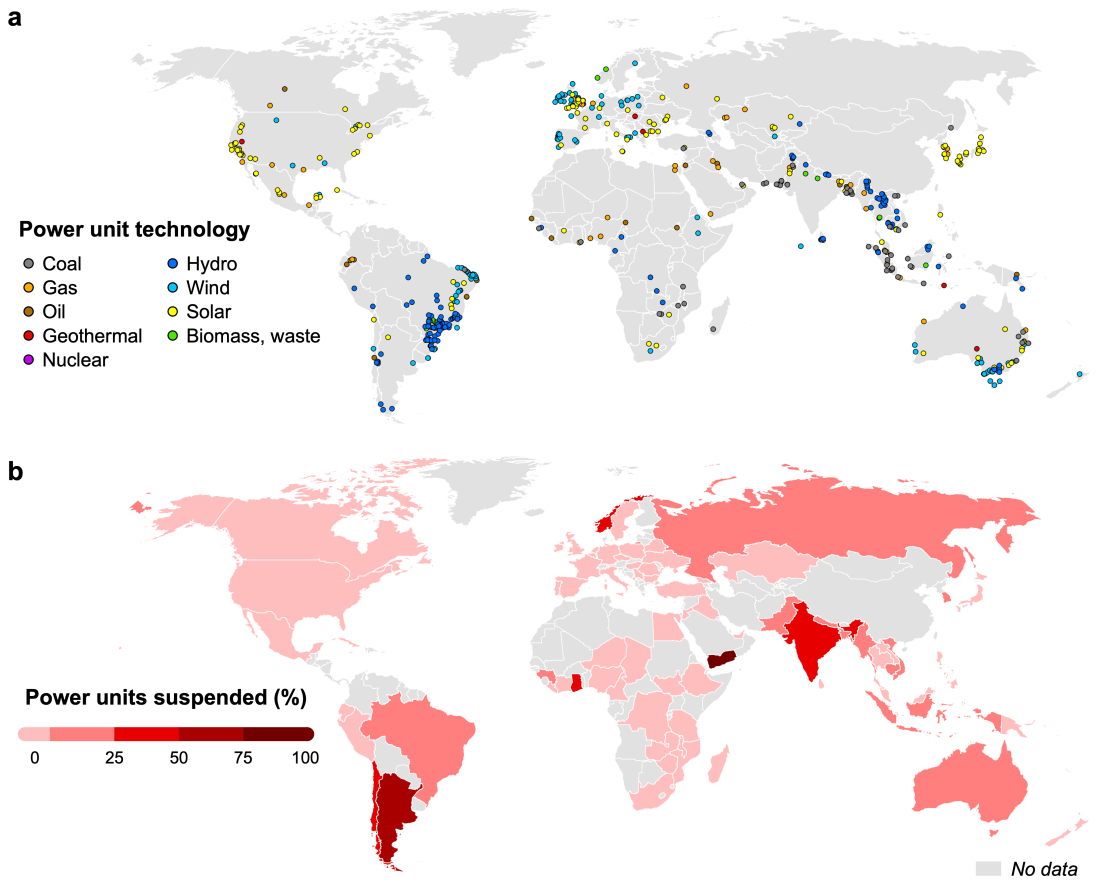


Figure 3.1: Location of Chinese firms' overseas investments in power units (a) and suspension rate by country (b)

Distribution of all 1393 Chinese firms' overseas investments in power units from 1997 to 2020 included in this study.

Out of 75 suspended projects, 60% were paused during the planning phase and 40% during the construction phase (Figure 3.2). Twenty-two projects were directly cancelled at the time of suspension and a quarter of the 53 projects initially classified as "delayed" were eventually cancelled as of 2020. Once suspended, the chance of resuming a project is low. Only 16 suspended power units (21%) have resumed planning or construction as of 2020, and just 7 power units (9%) have since become operational following suspension.

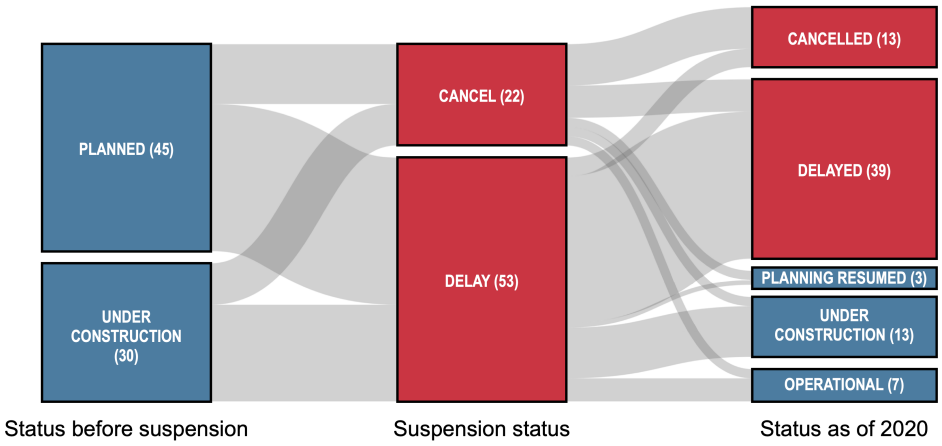


Figure 3.2: Status of suspended power units

Evolution in the status of the 75 power units that have been suspended at some point. Suspension includes project cancellation and delay (in red).

Central state-owned enterprises (SOEs) are the main investors in these energy projects, with the majority of investments in hydro and fossil fuel projects (Figure 3.3 (a)). Provincial SOEs have a similar investing pattern, though the number of investments is much smaller relative to central SOEs. Private firms invest heavily in renewable power projects, particularly solar and geothermal. Across all types of technologies, coal and hydro projects have the highest suspension rates, 17% and 9%, respectively. Geothermal power units also have a high failure rate of 9%, though the sample size is relatively small.

The fate of suspended projects is variable, but some trends are apparent. Chinese firms' investments in power units increased substantially after the launch of the Belt Road and Initiative (BRI) in 2013 (Figure 3.3 (b)). While some early project suspensions occurred in 2009, most suspensions occurred after 2015. Coal projects, in particular, encountered a significant number of suspensions for five continuous years up to 2020. Similarly, there has been a steep decline in the number of new hydro power projects since 2017, and the number of suspensions surpassed investments in 2020.

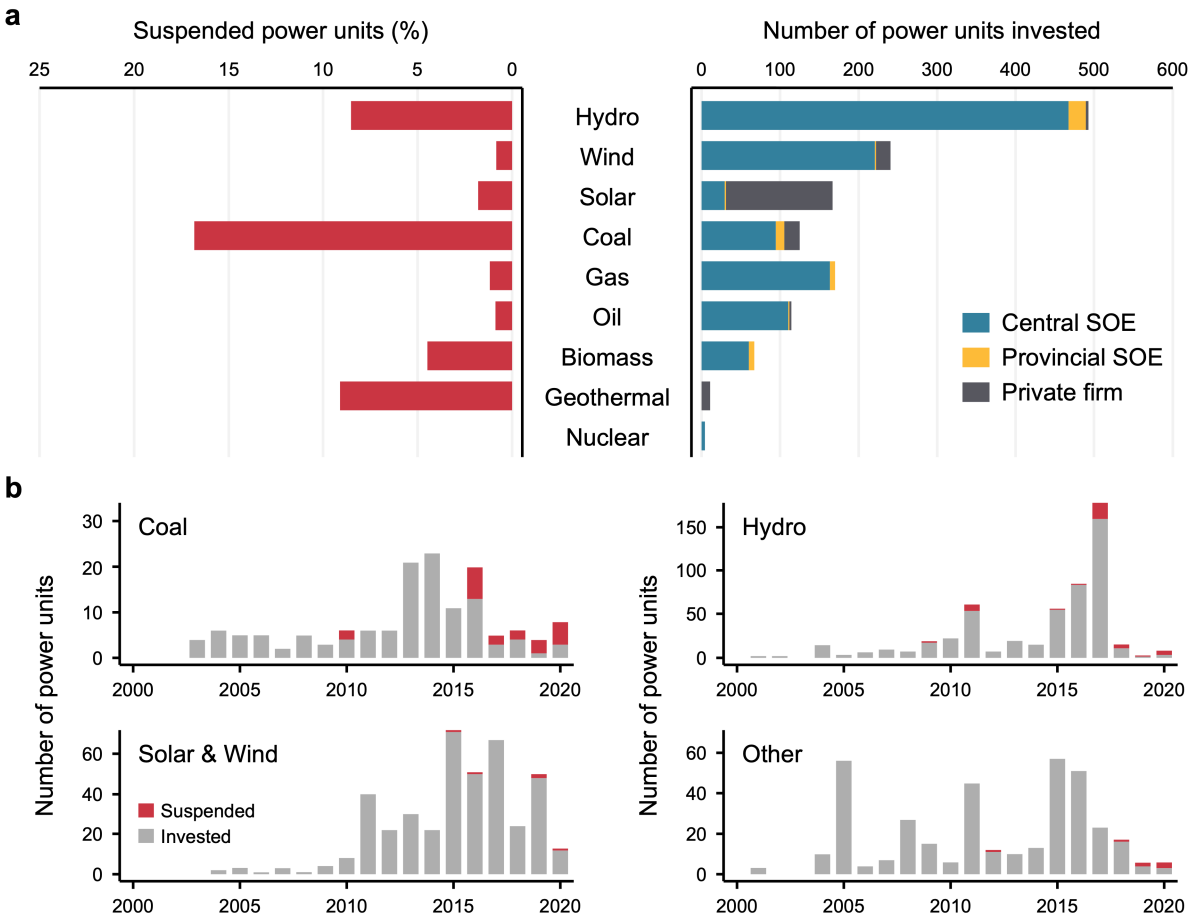


Figure 3.3: Chinese firms' overseas investments in power units by firm ownership (a) and year (b), categorised by technology

The suspension percentage in (a) measures the number of suspended power units relative to the number of total power units with investments by Chinese firms. Grey bars in (b) represent the number of power units invested in a particular year and red bars refer to the number of power units suspended in a particular year.

If we compare coal and hydro power project suspension trends of Chinese firms with other investment entities and regions, we find some comparable patterns and some unique differences (Figure 3.4). In the global market (Figure 3.4(a)), the number of suspended coal power projects surpassed the number of new coal investments in 2015 when the Paris Agreement was announced, and the global suspension number peaked in 2017 with more than 600 projects. Over the next three years, global newly invested coal power projects dropped to double digits. Taking a closer look at the 78 countries that host Chinese firms' investments in the electric power sector, Chinese firms began to push overseas coal projects in 2013, and their invest-

ments plateaued between 2013 and 2016, while non-Chinese entities' investments in host countries' coal power have gradually decreased since 2014. When the Chinese domestic market saw a surge in suspensions (Figure 3.4 (a)), presumably due to coal capacity cut policies imposed in 2016 (Shi et al., 2018, Global Energy Monitor, 2021), the number of suspended Chinese overseas investments also reached a peak.

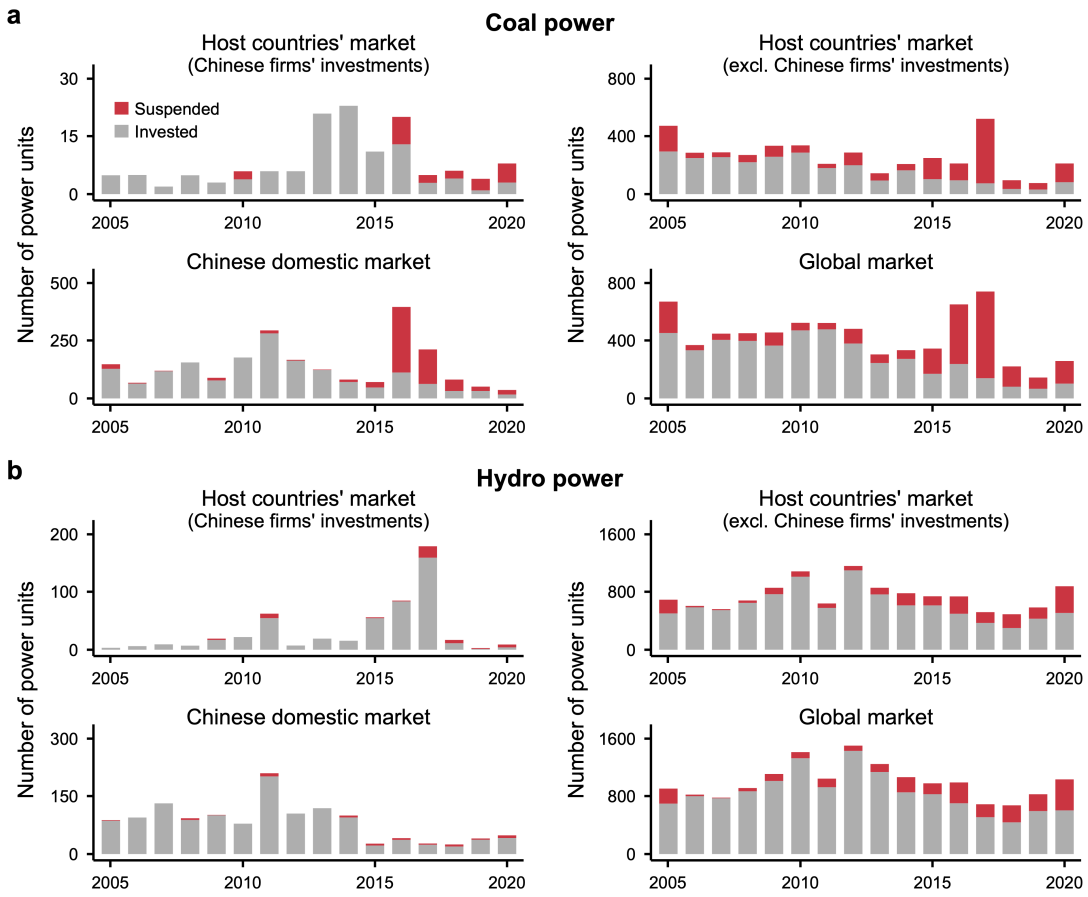


Figure 3.4: The changes in the number of newly invested and suspended coal (a) and hydro (b) projects by investment entity and region

Grey bars represent the number of power units invested in a particular year and red bars refer to the number of power units suspended in a particular year. Host countries refer to the 78 countries where Chinese firms have invested in the electric power sector.

In terms of hydro power (Figure 3.4 (b)): while Chinese overseas investments have increased from 2015 to 2017, their domestic investments have decreased compared to earlier years. On a global scale, the number of hydro power project suspensions have been increasing since 2010, peaking in 2020 with 432 suspensions. A

similar trend is reflected in non-Chinese entities' investments host countries, with a peak of 372 suspensions in 2020.

3.3 Power generation technology and suspension risk

To test whether suspension risks differ among power generation technologies, we perform a survival analysis, which considers not only whether the power unit was suspended but also the length of time it took for the suspension to occur. We collect annual observations for each power project from the year it was invested until the year it was suspended (for suspended projects) or 2020 (the end of sample period, for non-suspended projects) and each observation is recorded as sample year t for project i. The dependent variable is the hazard rate, which is the conditional probability that a failure event (i.e. an energy project is cancelled or delayed) occurs at a particular time interval. We report hazard ratios of independent variables in the results tables. A hazard ratio greater than one suggests an increased risk of failure, while lower than one implies a decreased risk. We use a Weibull parametric proportional hazards model to compare the influence of power technology choice on the hazard rate, and for robustness purposes, we also estimate the semi parametric Cox hazards model (see Section 3.7.2).

We include several controls for project-, firm- and country-level variables that have been shown to impact power generation projects from inception to implementation. At the project level, we control for project size (capacity), as larger projects would have longer lead times and may encounter more uncertainties due to greater complexity (Eybpoosh et al., 2011, Alova et al., 2021). We also control for firms' foreign direct investment (FDI) experience, as firms could learn from their prior international experiences to better manage overseas projects (Shaver et al., 1997, Haleblian et al., 2006, Collins et al., 2009). At the country level, we take into

consideration the gross domestic product (GDP) per capita and the population's electricity access rate. We also control for countries' score on the voice and accountability index, as public opinion could impact the successful implementation of energy projects (Upreti and van der Horst, 2004, Temper et al., 2020). Definitions and sources of all variables are listed in Table 3.4 in the Supplementary Information.

Table 3.1: The effect of technology choice on power project failures (hazard ratios are reported)

Variables	Hazard ratios			
	(1)	(2)	(3)	(4)
Coal	9.499***	3.568***	3.104**	3.787***
	(4.146)	(1.591)	(1.397)	(1.791)
Hydro	4.678***	5.477***	6.128***	8.188***
	(1.910)	(2.254)	(2.527)	(3.672)
Solar and Wind	0.642	1.018	1.018	0.847
	(0.376)	(0.605)	(0.603)	(0.505)
Capacity (log)		2.037***	1.987***	1.975***
		(0.226)	(0.224)	(0.217)
Firm FDI experiences			0.996**	0.994***
			(0.002)	(0.002)
GDP per capita (log)				0.956
				(0.176)
Access to electricity				0.991
				(0.009)
Voice and accountability				1.807**
				(0.425)
Observations	10,195	10,195	10,195	10,195

Notes: Results of the Weibull parametric proportional hazards model are reported as hazard ratios. A hazard ratio greater than one suggests an increased risk of failure, while lower than one implies a decreased risk. Column (1) reports the estimated hazard ratios and (standard errors) for the three interest variables: coal, hydro and solar wind without controlling specific characteristics. Columns (2) to (4) report the estimated hazard ratios and (standard errors) for the three interest variables by controlling for the project-, firm-, and country-specific characteristics, respectively. *P<0.10, **P<0.05, ***P<0.01.

Consistent with our observations in Figure 3.2, coal and hydro power units are more likely to be suspended than solar and wind projects, even after controlling for project-, firm-, and country-specific characteristics (Table 3.1). The hazard rate is 3.8 and 8.2 times as high for coal and hydro projects, respectively, in comparison to other power generation technologies. Among control variables, we also find that the likelihood of suspension is greater for projects with larger capacity, financed

by firms with less FDI experience, and located in countries with a greater degree of voice and accountability. We obtain nearly identical, statistically significant estimates using the semi-parametric Cox proportional hazards model (Table 3.7 in the Supplementary Information).

3.4 Environmental issues associated with coal and hydro power projects

Coal and hydro power plants have the greatest risk of suspension, particularly when they are located in countries where the people have a greater degree of voice and accountability. Given the potential environmental impacts of coal and hydro power projects (Carlson and Adriano, 1993, Rosenberg et al., 1997), it is possible that power plants in higher risk exposure areas may raise the salience of environmental concerns among the public, ultimately influencing the likelihood of suspension. Thus, we apply a survival analysis to test whether environmental risk exposures and protest events are associated with coal and hydro project suspensions. While the environmental risk exposures posed by different types of coal and hydro power projects are often highly variable based on operational or mechanical characteristics (Cui et al., 2012, Kuriqi et al., 2021), we incorporate three variables to reflect the primary environmental claims expressed by mobilising communities in global resistance movements to hydro and coal power projects, as systematically mapped in the Global Atlas of Environmental Justice (EJAtlas). The selected risk exposure variables also relate to the environmental issues that have been reported with suspended Chinese firms' overseas coal and hydro projects (see Table 3.2 for a selection of exemplar case studies).

Firstly, we use local population density surrounding coal power projects as a proxy for the number of people exposed to pollution—the most prominent local environmental risk associated with coal-fired projects. Burning coal emits pollutants

Table 3.2: Examples of environmental issues in suspended coal and hydro projects

Main claims recorded in EJAtlas	Project, Country	Environmental concerns	Conflicts documented	Status in 2020
Coal				
Pollution, Climate justice	Nagan Raya, Indonesia	Air pollution, water pollution, waste disposal	Local residents protest against dust and other air pollutants; local media reported chemicals discharged from the plant killed fish in nearby waterways[1]	Delayed
	Aboano, Ghana	Air pollution, climate change effects	Activists protest as vulnerable population are exposed to air pollutants released by the plant; NGOs expressed opposition in response to heavy flooding events in local area[2]	Cancelled
Hydro				
Impacts on ecosystems larger than expected or stated in official documents	Batang Toru, Indonesia	Biodiversity loss	Organisations warn the project's potential threat to critically endangered species, i.e.Tapanuli orangutan[3]	Delayed
	President NK and JC, Argentina	Biodiversity loss, impacts on glaciers	Suspended by Supreme Court order for envi. impact assessment; Envi. groups express concerns on UNESCO-declared glaciers and critically endangered species[4]	Under construction

[1] Global Energy Monitor's Wiki page; Global Atlas of Environmental Justice
[2] Global Energy Monitor's Wiki page
[3] IUCN(2020). Batang Toru Hydro power Project Factcheck and References on Key Issues
[4] China Dialogue (2021).New Argentina Government Reactivates Controversial Patagonia Dams.

that are detrimental to the health of nearby populations (Chen et al., 2018a, Yang and Chou, 2018, Barrows et al., 2019), and if there is a larger population exposed to these pollutants, the pollution issue may be more salient, ultimately increasing the risk of suspension (Table 3.2). Because coal projects tend to be placed 10-20 km from densely populated areas (Barrows et al., 2019), we use the mean population density for all areas within a radius of 25 km surrounding each coal power project.

Secondly, we hypothesise that people who have experienced catastrophic losses as a result of extreme weather events would be more concerned about the construction of coal power plants nearby, as extreme weather events have been found to play a role in raising public awareness and spurring climate actions, particularly in areas that have suffered from the effects of climate change (McAdam, 2017, Zanocco et al., 2018, Boudet et al., 2020) (cf. Aboano example in Table 3.2). To account for climate concerns from local communities, we include climate-related fatalities as a proxy of the salience of perceived climate vulnerability, which is frequently used in climate risk analyses (Bakkensen and Mendelsohn, 2019, Boudet et al., 2020). Following the method of the Climate Risk Index (Eckstein et al., 2021), we take the country's annual average number of deaths from weather-related loss events (i.e. storms, floods, heatwaves) per 100,000 inhabitants over the past 20 years.

Thirdly, the environmental concern of hydro power projects to date has centred on their impacts on local ecosystems. Hydro power projects can have many negative ecological impacts upstream or downstream of the dam, such as wildlife injury, reductions in connectivity, changes in water temperature and pH values, or declines in species richness (Kuriqi et al., 2021). As documented in the EJAtlas, mobilising communities are mostly concerned about the potential impacts of hydro power projects that far exceed the expected or stated impacts in official documents, particularly when there are threatened species or protected habitats nearby (cf. Indonesia and Argentina case studies in Table 3.2). For this analysis, we include the distance from each hydro power project to the nearest international, national,

or regional protected area (UNEP-WCMC and IUCN, 2021) as a proxy for the project's risks to ecosystems of high ecological importance and biodiversity value. We hypothesise that hydro projects located closer to protected areas may be more likely to adversely impact these important ecosystems (Ng et al., 2020, Yang et al., 2021), ultimately increasing the likelihood of suspension.

In addition to the above technology-specific risk exposure measures, we include the presence of place-based environmental protests as a proxy of the intensity of environmental movements in the local area. Anecdotal evidence suggests that protest is the main environmental movement form used by affected communities and environmental organisations to express public opposition to power projects (Table 3.2). Protests could amplify public opinion and raise the salience of environmental issues to regulators and investors (Gadgil and Guha, 1994, Agnone, 2007). We hypothesise that the presence of environmental protests near the power project is positively related to the suspension risk of a project. We use protest data from Google's Global Database for Events, Language and Tone (GDELT) project — currently the most powerful research tool to analyse the occurrence of protests globally — that monitors the world's news media in print, broadcast, and web formats in over 100 languages from 1979 onwards (Brancati and Lucardi, 2019, Manacorda and Tesei, 2020, Iacoella et al., 2021). We follow Iacoella et al. (2021) to identify environmental protest events that have occurred within a 50 km radius of each power project and construct a dummy variable for each project at sample year t that indicates if there was at least one environmental protest recorded nearby in either year t or year $t-1$. We use protests occurring in year t and $t-1$ to represent the recent environmental movement intensity around each power project. See Table 3.4 in the Supplementary Information for a description of all variables included in the analysis, and Table 3.5 for correlations between variables.

3.5 Could environmental risks be contributing to the failure of power projects?

Our results provide evidence of the association between these environmental risks and the likelihood of power projects being suspended, but to varying degrees. As shown in the summary statistics (Table 3.6 in the Supplementary Information), suspended coal power projects tend to be located in more densely populated areas, and in areas with more climate-related fatalities and the presence of recent environmental protests, than non-suspended projects. Our analysis confirms these environmental risks are associated with coal project suspension (Table 3.3). A one-unit increase in the population density within a radius of 25 km increases the hazard rate by 29% for coal power investments, accounting for all control variables. Coal projects are also more likely to be suspended where climate-related fatalities are greater – one death increase per 100,000 inhabitants in host countries increases the hazard rate nearly ten-fold. We also find that the hazard rate of coal projects in areas that have experienced recent environmental protests are more than four times higher than projects located in areas with no environmental protests.

On average, hydro power projects tend to be located more than 90 km from any protected areas, with the closest distance for a single hydro project being 8 km from the nearest protected area. Suspended hydro power projects, however, tend to be closer to protected areas than non-suspended projects (Table 3.6 in the Supplementary Information). By applying a parametric Weibull survival analysis model, we find that the distance to protected areas is significantly associated with the fate of hydro power projects, with suspension risk decreasing the further a hydro power project is from protected areas (Table 3.3): a one kilometre increase in the distance from a nearby protected area reduces the hydro power project's hazard rate by 3%. However, we find no significant effect of local environmental

protests on suspension. One possible explanation may be that hydro projects tend to be located in remote areas and the place-based protest measure may not fully capture the environmental movement pressure linked to the project's location, as protests often take place in regions where regulators/investors are based rather than at the remote dam site. These results remain robust when we apply an alternative semi-parametric Cox hazards model (Table 3.8 in the Supplementary Information).

Table 3.3: The effect of environmental risks on coal and hydro power project failures (hazard ratios are reported)

Variables	Hazard ratios						
	Coal power projects				Hydro power projects		
	(1)	(2)	(3)	(4)	(5)	(6)	(7)
Population density	1.426**			1.291*			
	(0.205)			(0.195)			
Climate-related fatalities		10.968**		9.503**			
		(10.699)		(9.194)			
Environmental protest			7.852***	4.355**	0.00001		0.00001
			(4.571)	(2.881)	(0.001)		(0.002)
Distance to protected area						0.967***	0.968***
						(0.006)	(0.006)
Capacity (log)	1.345	1.380	1.492*	1.469	2.623***	3.420***	3.385***
	(0.323)	(0.337)	(0.358)	(0.356)	(0.415)	(0.603)	(0.596)
FDI experiences	1.001	1.002	1.002	1.002	0.994***	0.997	0.997
	(0.005)	(0.005)	(0.005)	(0.006)	(0.002)	(0.003)	(0.003)
GDP per capita (log)	0.778	0.494*	0.671	0.792	2.390*	3.359**	3.067**
	(0.256)	(0.208)	(0.228)	(0.293)	(1.156)	(1.702)	(1.611)
Access to electricity	1.010	1.024	1.016	1.011	0.960**	0.947***	0.949***
	(0.017)	(0.018)	(0.017)	(0.020)	(0.015)	(0.015)	(0.015)
Voice and accountability	2.439**	3.908**	2.019	2.865**	1.092	0.815	0.873
	(1.086)	(2.425)	(0.888)	(1.455)	(0.515)	(0.460)	(0.503)
Observations	1044	1044	1044	1044	3278	3278	3278

Notes: Results of the Weibull parametric proportional hazards model are reported as hazard ratios. A hazard ratio greater (lower) than 1 suggests an increased (decreased) risk of failure. Columns (1) to (4) report the estimated hazard ratios and (standard errors) for coal power projects. Columns (5) to (7) report the estimated hazard ratios and (standard errors) for hydro power projects. *P<0.10, **P<0.05, ***P<0.01.

3.6 Discussions and policy implications

This study explores Chinese firms' overseas power projects that have been suspended and whether environmental issues may be contributing to investment failures. Among all technologies of power plants, we find coal and hydro power projects have a higher chance of being suspended at some point in their development. We also identify technology-specific environmental risk exposures associated with hydro and coal project suspensions. Coal projects located in more densely populated areas and in countries with more fatalities from extreme weather events are more likely to be suspended, and hydro power projects located closer to protected areas tend to have a higher probability of being halted. Additionally, coal projects located in areas with the presence of environmental protests have a higher chance of being abandoned, but we do not find such an effect on hydro projects.

Our results suggest that investing in environmentally risky projects is not only potentially detrimental on environmental grounds, but also financial grounds. There is a long history of public scrutiny over the financial support for coal or hydro power plants because of the significant impacts their investments might have on the environment (International Rivers, 2012, Chen et al., 2020). Previous studies have highlighted the stranding risk of investing in fossil fuels (McGlade and Ekins, 2015, Mercure et al., 2018, Tong et al., 2019), yet the implications may not be salient enough to stop fossil fuel financing for investors who view such risks as long-term risks, where the financial impacts will not be felt for decades to come. As our study suggests, environmental risks embedded in coal and hydro projects are likely to translate into cancellation or postponement, resulting in financial losses in a much shorter time scale.

In September 2021, China pledged to stop building new coal-fired power plants abroad (Sullivan, 2021), which is a major step for the global effort against climate change. Meanwhile, there are still a significant number of China-funded overseas

coal projects under construction and in the planning stage (Gallagher et al., 2019). These on-going projects are under high suspension risk, and how stakeholders deal with these projects is crucial for minimising financial losses. Chinese policy banks, state and privately-owned commercial banks, and firms only fund 13% of the total finance (including both public and private) for operational and planned coal capacity outside China between 2013 and mid-2019 (Ma and Gallagher, 2021). Lessons learnt from Chinese overseas investments and the global trend of increasing coal project suspensions in the last five years should send a key message to the remaining international coal financiers that investing in coal is risky. As of November 2021, most international development finance institutions have committed to reducing or ending coal investment (Ray et al., 2021). The private sector, such as institutional investors and commercial banks (Urgewald, 2021), should join public financiers to phase out financing coal and consider directing more capital toward renewable power generation technologies, such as solar and wind. Of equal importance, hydro power projects face a similar level of failure risk as coal power projects due to their potential damage to the ecological systems nearby. This result sends a wake-up call for Chinese investors to proactively review plans for future hydroelectric projects and take further action to end the financing when the environmental costs outweigh the economic benefits.

The surge in the number of cancelled coal and hydro projects and the associated environmental risks should also signal an alarm to host countries. Chinese overseas investments tend to follow the demand of host countries (Voituriez et al., 2019, Tritto, 2021) and these countries are often less developed and lack sufficient environmental policies and incentives to steer investment towards clean energy (Gallagher et al., 2021). Many environmentally risky projects were proposed by host countries, who then approached the Chinese side for funding (Kong and Gallagher, 2021). With China committing to stop building new overseas coal projects, the scarce opportunities to get public coal finance may push host countries to appeal for

hydro finance, which is, nonetheless, also highly risky as shown in our analysis and others (Yang et al., 2021). Seeking foreign investments in developing less risky and greener energy, such as solar and wind power, as well as auxiliary grid connection to integrate renewable energy is promising for host countries (Huang et al., 2019). For example, the Bangladesh government cancelled the China-funded Gazaria 350 MW coal-fired plant in 2020 and replaced it with electricity grid updates to reduce system losses in the rural electricity system (Nicholas, 2020)

Recognisably, our results come with several limitations. This study only focuses on technology-specific environmental risks based on where the power projects are located. Other risks, such as operational, political and financial risks may also affect the implementation of power projects (Yuan et al., 2019). We control for project-, firm- and country-level characteristics to minimise confounding factors, though there could still be omitted variables as each project will be situated in its own unique social, ecological, and political context that ultimately influences the fate of the investment. One needs to be cautious to interpret our findings, as strict causality may not be established from our analysis. While our results suggest a link between coal projects' suspension and the presence of environmental protests in the area, more detailed analyses are necessary to determine whether public resistance to a power project has directly contributed to the failure of that specific project.

3.7 Data and method

3.7.1 Data sources and key variables

Power unit data and suspension status: We obtain unit-level power plant data from the World Electric Power Plants (WEPP) database (S&P Global Market Intelligence, 2020). The WEPP database provides information such as power unit name, technology, capacity, status, commission year, location, investing company, etc. The power unit status is recorded in ten categories: cancelled, under

construction, planned, deferred (halted before construction starts), delayed (halted after construction started), in operation, deactivated, retired, shutdown, and unknown. We treat the status recorded as "cancelled", "delayed", and "deferred" to be suspension statuses. By tracing the historical release of the WEPP database from 2000 to 2020, we construct a panel dataset that records the evolution of power unit's status from the first year it appeared in the database to the most recent year.

Chinese firms' investments: To identify Chinese firms' overseas power investments, we compile investment data from the following sources: 1) Foreign direct investment from China's Global Power Database, maintained by Global Development Policy Center at Boston University; 2) Bloomberg's Merger Acquisition deal database; and 3) China Global Investment Tracker, compiled by the American Enterprise Institute and the Heritage Foundation. We match these investment data with the WEPP power unit and identified 1393 overseas power units that have investments by Chinese firms, among which 75 units have suspension statuses recorded.

Investment year: The investment year of each power unit with investments from Chinese firms is defined in the following way: 1) For investments that we are able to identify the exact year when the investment was made (e.g. from news reports, archives, or investing company's website), we use it as the investment year; 2) For merger and acquisition (M&A) deals, we use the year when the M&A deals were signed as the investment year; 3) For the rest of power units where accurate investment year information is not available, we use the first year when a power unit appeared in the WEPP database as the investment year. For the non-Chinese overseas investments and Chinese domestic investments, we use the first year when a power unit appeared in the WEPP database as the investment year. WEPP database started to have a wide coverage of global power plants from 2004. Therefore, we produce Figure 3.3 from 2005 onwards.

For the purpose of survival analysis, we convert the data into a panel format.

The observations started from the investment year and ended either the first year when a power unit has a suspension status recorded (for power units that have been suspended) or in 2020 (for power units that do not have suspension statuses recorded). Each observation is recorded as a sample year t for project i. This allows us to construct a panel of 10,195 observations. Eight power units were invested by Chinese firms before 2000 (the earliest is in 1997), but these units were not suspended during 2000 to 2020. For years before 2000 that we are not able to trace power plants' status, we assume they were not suspended from their investment year to 2000.

Firm ownership: To define the ownership of Chinese firms, we identify firms' registered capital and shareholder information recorded in Qichacha. Qichacha is a corporate information query tool that provides information on firms recorded in the National Enterprise Credit Inquiry System. If the main shareholder of a Chinese firm is the State Council's State-Owned Assets Supervision and Administration Commission or provincial governments, we classify the firm as a central state-owned enterprise (SOE) or provincial SOE. Other types of firms are all treated as private firms.

Country-level controls: We obtain annual GDP per capita (2010 constant USD) and access to electricity (percent of population) data from World Bank Development Indicators. The annual voice and accountability data are gained from World Bank's World Governance Indicators.

Power unit's geolocation: We collect power unit's longitude and latitude by matching WEPP data with Global Energy Monitors (GEM) Coal Plant Tracker and WRI's Global Power Plant Database. 80% of the coal power units covered in our dataset are included in the GEM database, therefore we use the geographical coordinates GEM provides for these units. For the rest of coal power units and other types of power units, we match them to the WRI database, which provides latitude and longitude information for a range of publicly available power plants. If

the power unit is not included in the GEM or WRI databases, we geolocate the asset based on the best available location information, such as the city, state, and country information provided in the WEPP and available information from archives, news reports, and other sources.

High resolution population density: The high resolution population density data based on power units' location are extracted from NASA's Socioeconomic Data and Applications Center's gridded population of the world. The data consist of estimates of population density at 30 arc-second grid cells based on counts consistent with national census and population register data, and are available for every 5 years from 2000 to 2020. We use the average population density in buffer zones with a radius of 25 km around each point as the proxy of population exposure to pollution from coal-fired projects. The variable is measured in average number of persons per square kilometre. We use population density data in the year 2000 for observations before 2000, data in the year 2005 for observations between 2001 and 2005, and so forth.

Climate-related fatalities: We extract the number of deaths of climate-related events from Climate Risk Index reports published by Germanwatch (Eckstein et al., 2021). The raw data are retrieved from Munich Re's NatCatSERVICE, one of the world's most comprehensive databases for analysing natural catastrophe losses. The number of deaths incorporate fatalities in weather-related events, i.e., storms, floods, extreme heat and cold waves, etc. Fatalities is a common indicator used in climate risk analysis to measure the loss from climate-related events (Bakkensen and Mendelsohn, 2019, Boudet et al., 2020, Eckstein et al., 2021). The relative form of fatalities (number of deaths per 100,000 inhabitants) is used.

Following Germanwatch's Climate Risk Index methodological framework (Eckstein et al., 2021), we use the country's annual average number of weather-related deaths over the past 20 years to capture the degree to which countries have already been affected by extreme weather events. For example, the 2017 value represents

the annual average of fatalities over the years 1998 and 2017. This could minimise the measurement error associated with the likelihood that a single extreme weather event is recorded. Most years take the average value over the past 20-year period, except for years before 2009, which take the average value dating back to 1990 due to data availability constraints.

Distance to protected areas: We estimate the shortest Euclidean distance in kilometres from each power unit to the nearest protected area. To reduce the large skew in distribution, distances greater than 100 km were assigned a value of 100. We include all international, national, and regional protected areas officially designated or inscribed as of April 2021 (UNEP-WCMC and IUCN, 2021). Protected areas without mapped boundaries are excluded.

Environmental protest: We obtain environmental protest data from Google's Global Database for Events, Language and Tone. The protest data is gathered by Google Jigsaw from the world's digital news sources in over 100 languages from 1979 till present. The big data project identifies action types, location, year and actors involved in protest events by encoding key terms in the news piece and through lexical analysis. Environmental protest data are downloaded by selecting protesters' type being "environment", i.e. entities for whom environmental and ecological issues are their primary focus, such as wildlife preservation and climate change. We follow Iacoella et al. (2021) to match the geocoded information of protest events with power projects' 50 km buffer zones and construct a dummy variable to indicate whether environmental protests were recorded over the past two years within the defined buffer zones. For example, the variable equals one in sample year t if there was at least one environmental protest recorded in 50km buffer zones surrounding the power projects in either year t or year $t-1$.

3.7.2 Methods

The purpose of the study is to investigate factors associated with power plant suspension. Because the power plant project data involve right-censoring, which occurs when they have not yet been suspended and remain operation until the end of the study (December 2020), we use survival analysis for censored data as our research method (Melnyk et al., 1995, Cleves et al., 2008). Compared to (e.g.) logit regression, survival analysis does not only consider information on whether an event occurred, but also the length of time it took for the event to occur. Survival analysis originates from medical research to study the impact of certain treatments on the survival of patients and the model has been widely used in business and management literature, such as Initial Public Offerings (IPO) survival, research project termination, and the survival of religiously motivated financial institutions (Hoang and Rothaermel, 2010, Espenlaub et al., 2012, 2016, Alandejani et al., 2017, Pappas et al., 2017). Survival analysis has also been used in energy literature. For example, Wang et al. (2021) applied a survival analysis to examine factors that contribute to the failure of carbon capture and sequestration projects worldwide.

In our study, the event of interest occurs when a power generation unit is suspended. In survival analysis, the dependent variable is the hazard rate, which is the conditional probability that an event occurs at a particular time interval (Hosmer and Lemeshow, 2008). It is an unobservable variable and controls both the event occurrence and the timing of the event. We first use a parametric Weibull model to address our research question. In general, we specify the parametric hazard rate as a function of time and covariates, as follows:

$$h(t|x) = h_0(t)r(x, \beta) \tag{3.1}$$

The hazard rate, $h(t|x)$ is a product of two functions. The function, $h_0(t)$, is often referred as the baseline hazard function, which characterises how the hazard rate

changes as a function of survival time. It can be casually described as the "time function". The other function, $r(x, \beta)$, describes how the hazard rate changes as a function of our subject covariates. It can be casually described as the "characteristics function". The Weibull parametric model assumes the baseline time function follows Weibull distribution, which is specified as:

$$h_0(t) = p\lambda t^{p-1} \tag{3.2}$$

where the value of p determines the shape of the distribution and the value of λ determines its scale. When p=1, the entire time function collapses to 1, and the overall hazard function turns into an exponential regression that is suitable for modelling data where the hazard (i.e. risk) is constant over time. The hazard rate increases when p>1 and decreases when p<1. Following Cox (1972), we define the characteristics function as:

$$\begin{aligned} r(x, \beta) &= \exp(\beta_1 x_1 + \cdots + \beta_p x_p) \\ &= \exp(\beta_1 Risk + \beta_2 Controls) \end{aligned} \tag{3.3}$$

where, the covariates, $x_1 + \cdots + x_p$, include variables of interest (i.e. power generation technology types or environmental risks) and control variables, and $\beta_1 + + \cdots + \beta_p$ are the model parameters describing the effect of the covariates.

Integrating Equation 3.2 and 3.3 into Equation 3.1, the Weibull parametric regression model is specified as:

$$h(t|x) = p\lambda t^{p-1} \exp(\beta_1 Risk + \beta_2 Controls) \tag{3.4}$$

where $h(t|x)$ is the hazard at time t for a given set of covariates $x_1 + \cdots + x_p$p. The quantities $\exp(\beta_i)$ are called hazard ratios. In our analysis, a hazard ratio greater than one would indicate that the power project is more likely to be suspended, and

a hazard ratio less than one means it is less likely to be suspended.

The semi-parametric Cox hazard model takes the same form as the Weibull parametric model, but it does not specify the baseline hazard function $h_0(t)$ (Hosmer and Lemeshow, 2008). Unlike Weibull model that assumes a parametric form on the baseline hazard (Cleves et al., 2008), Cox model uses a non-parametric Aalen-Breslow estimator to estimate the hazard function. As a robustness test, we use the semi-parametric Cox hazards model to compare with the results of the Weibull parametric model.

3.8 Supplementary information

Table 3.4: Definition of variables

Variables		Definition	Sources
a) Technology types			
Coal		An indicator that equals one if it is a coal power project and zero otherwise	World Electric Power Plants (WEPP)
Hydro		An indicator that equals one if it is a hydro power project and zero otherwise	WEPP
Solar and wind		An indicator that equals one if it is a solar or wind power project and zero otherwise	WEPP
b) Environmental risks			
Coal related	Population density	The average number of persons per square kilometre in buffer zones with a radius of 25km around each power unit. Log form is used.	NASA's Socioeconomic Data and Applications Center's world gridded population
	Climate-related fatalities	The country's annual average number of deaths per 100,000 inhabitants in extreme weather events over the past 20 years.	Climate Risk Index reports, published by Germanwatch
Hydro related	Distance to the protected area	The shortest Euclidean distance in kilometres from each power unit to the nearest protected area.	UNEP-WCMC and IUCN (2021)
	Environmental protest	An indicator that equals one if there was at least one environmental protest recorded in 50km buffer zones of power projects in either year t or year $t-1$ and zero otherwise.	Google's Global Database for Events, Language and Tone
c) Control variables			
Capacity		The amount of electricity a power generator can produce when it is running at full blast. It is measured in megawatts (mw). Log form is used.	WEPP
Firm FDI experiences		The accumulative number of power units invested overseas until year t by firm i.	WEPP
GDP per capita (log)		Gross Domestic Product per capita (in constant 2010 USD).	World Bank Development Indicators
Access to electricity		The percentage of a country's population with access to electricity.	World Bank Development Indicators
Voice and accountability		A country index that captures perceptions of the extent to which a country's citizens are able to participate in selecting their government, as well as freedom of expression, freedom of association, and a free media.	World Bank World Governance Indicators

Table 3.5: Matrix of correlations

Variables	VIF	Coal	Hydro	Solar and Wind	Distance to protected areas	Population density	Climate-related fatalities	Envi. protest	Capacity (log)	Firm FDI experiences	GDP per capita (log)	Access to electricity	Voice and accountability
Coal	1.96	1											
Hydro	2.64	-0.4229	1										
Solar and Wind	1.79	-0.2319	-0.3432	1									
Distance to protected areas	1.41	-0.0544	0.2685	-0.0847	1								
Population density	2.04	0.1319	-0.2292	-0.2457	0.0125	1							
Climate-related fatalities	1.34	0.0628	-0.0646	-0.0807	-0.0335	0.1023	1						
Envi. protest	1.18	-0.0157	-0.1012	-0.0325	-0.1174	0.3041	-0.0208	1					
Capacity (log)	1.45	0.4649	-0.2566	-0.0905	-0.0169	0.0404	0.1244	0.0513	1				
Firm FDI experiences	1.90	-0.4243	0.4034	0.1463	0.2769	-0.0639	-0.2955	-0.1019	-0.3659	1			
GDP per capita (log)	5.84	-0.2449	-0.1168	0.2818	-0.2646	-0.3647	-0.3024	0.0361	-0.1466	0.1995	1		
Access to electricity	2.49	-0.2202	-0.0285	0.188	-0.1208	-0.1704	-0.1989	0.0297	-0.0878	0.309	0.7332	1	
Voice and accountability	4.33	-0.1849	-0.0962	0.2964	-0.0485	-0.4757	-0.4112	-0.0321	-0.2128	0.1984	0.8051	0.5122	1

Table 3.6: Summary statistics of survival analysis sample

Variables			Mean (1)	St.d (2)	Min (3)	Max (4)	Obs (5)
a) Technology types							
Coal			0.102	0.303	0	1	10195
Hydro			0.322	0.467	0	1	10195
Solar and wind			0.235	0.424	0	1	10195
b) Environmental risks							
Coal related	Population density	All Coal	5.003	1.866	0.055	9.30	1044
		Suspended	5.646	1.753	3.57	7.99	121
		Non-suspended	4.894	1.864	0.05	9.30	911
	Climate-related fatalities	All Coal	0.220	0.210	0	2.04	1044
		Suspended	0.300	0.460	0	2.04	121
		Non-suspended	0.210	0.150	0	0.64	911
	Environmental protest	All Coal	0.045	0.207	0	1	1044
		Suspended	0.116	0.321	0	1	121
		Non-suspended	0.032	0.176	0	1	911
Hydro related	Distance to protected area	All Hydro	91.001	22.864	8.28	100	3278
		Suspended	77.598	28.306	32.12	100	247
		Non-suspended	92.093	22.011	8.28	100	3031
	Environmental protest	All Hydro	0.030	0.171	0	1	3278
		Suspended	0.032	0.177	0	1	247
		Non-suspended	0.030	0.171	0	1	3031
c) Control variables							
Capacity (mw)			109.752	185.324	0.003	1770	10195
Firm FDI experiences			77.234	65.702	1	206	10195
GDP per capita(log)			8.830	1.405	5.42	11.44	10195
Access to electricity			88.929	20.513	6	100	10195
Voice and accountability			0.023	0.932	-2.23	1.73	10195

Table 3.7: The effect of technology choice on power project failures failures (semi-parametric Cox model result)

Variables	Hazard ratios			
	(1)	(2)	(3)	(4)
Coal	9.431***	3.572***	3.161**	3.803***
	(4.116)	(1.593)	(1.421)	(1.798)
Hydro	4.634***	5.347***	5.963***	7.852***
	(1.892)	(2.200)	(2.459)	(3.514)
Solar and Wind	0.643	1.012	1.017	0.851
	(0.376)	(0.601)	(0.603)	(0.508)
Capacity (log)		2.016***	1.965***	1.961***
		(0.223)	(0.220)	(0.216)
Firm FDI experiences			0.996**	0.994****
			(0.002)	(0.002)
GDP per capita (log)				0.969
				(0.178)
Access to electricity				0.992
				(0.009)
Voice and accountability				1.733**
				(0.403)
Observations	10,195	10,195	10,195	10,195

Notes: : Results of the semi-parametric Cox model are reported as hazard ratios. A hazard ratio greater than one suggests an increased risk of failure, while lower than one implies a decreased risk. Column (1) reports the estimated hazard ratios and (standard errors) for the three interest variables: coal, hydro and solar wind without controlling specific characteristics. Columns (2) to (4) report the estimated hazard ratios and (standard errors) for the three interest variables by controlling for the project-, firm-, and country-specific characteristics, respectively. *P<0.10, **P<0.05, ***P<0.01.

Table 3.8: The effect of environmental risks on coal and hydro power project failures (semi-parametric Cox model result)

Variables	Hazard ratios						
	Coal power projects				Hydro power projects		
	(1)	(2)	(3)	(4)	(5)	(6)	(7)
Population density	1.405**			1.280*			
	(0.200)			(0.192)			
Climate-related fatalities		7.364**		7.235**			
		(6.931)		(6.831)			
Environmental protest			7.735***	4.534**	0.00001		0.00001
			((4.520)	(2.957)	(0.001)		(0.002)
Distance to protected area						0.970***	0.971***
						(0.006)	(0.006)
Capacity (log)	1.331	1.383	1.495*	1.476	2.575***	3.182***	3.156***
	(0.321)	(0.339)	(0.362)	(0.361)	(0.408)	(0.551)	(0.547)
FDI experiences	1.001	1.002	1.002	1.002	0.994**	0.998	0.998
	(0.005)	(0.005)	(0.005)	(0.005)	(0.002)	(0.003)	(0.003)
GDP per capita (log)	0.795	0.527	0.659	0.780	2.685*	3.502**	3.303**
	(0.255)	(0.216)	(0.223)	(0.282)	(1.365)	(1.758)	(1.720)
Access to electricity	1.011	1.023	1.016	1.011	0.957***	0.947***	0.948***
	(0.018)	(0.018)	(0.017)	(0.020)	(0.016)	(0.015)	(0.016)
Voice and accountability	2.291*	3.435**	2.198*	2.908**	0.977	0.702	0.730
	(1.004)	(2.061)	(0.968)	(1.429)	(0.467)	(0.377)	(0.402)
Observations	1044	1044	1044	1044	3278	3278	3278

Notes: Results of the semi-parametric Cox model are reported as hazard ratios. A hazard ratio greater than one suggests an increased risk of failure, while lower than one implies a decreased risk. Columns (1) to (4) report the estimated hazard ratios and (standard errors) for coal power projects. Columns (5) to (7) report the estimated hazard ratios and (standard errors) for hydro power projects. *P<0.10, **P<0.05, ***P<0.01.

4 Plant conversions and abatement technologies cannot prevent stranding of power plant assets in 2°C scenarios

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Abstract

Continued fossil-fuel development puts existing assets at risk of exceeding the capacity compatible with limiting global warming below 2°C. However, it has been argued that plant conversions and new abatement technologies may allow for a smoother transition. We quantify the impact of future technology availability on the need for fossil fuel power plants to be stranded, i.e. decommissioned or underused. Even with carbon capture and storage (CCS) and bioenergy widely deployed in the future, the average value across all models suggest a total of 267 PWh electricity generation (ten times global electricity production in 2018) may still be stranded. Coal-to-gas conversions could prevent 10-30 PWh of stranded generation. CCS retrofits, combined with biomass co-firing, could prevent 33-68 PWh. In contrast, lack of deployment of CCS or bioenergy could increase stranding by 69 or 45 percent respectively. Stranding risks remain under optimistic technology assumptions and even more so if CCS and bioenergy are not deployed at scale.

Key words: Stranded assets; Carbon Capture and Storage; Bioenergy; Plant conversions; Fuel switching.

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4.1 Introduction

Because fossil fuel capacity has drastically expanded in the last decade (Carbon Brief, 2020), there are now strong concerns that the existing stock of fossil-fuel power plants, if used as planned, could exhaust global carbon budgets consistent with the aims of the Paris Agreement (Davis and Socolow, 2014, Pfeiffer et al., 2016, Tong et al., 2019). However, the energy industry has argued that technological solutions, especially energy efficiency, carbon capture and storage (CCS), bioenergy, natural offsets and switching from coal to gas, could substantially reduce the carbon footprint of energy production from fossil fuels and allow production to continue for longer without jeopardising the Paris Agreement (IEA Coal Advisory Board, 2013, Glencore, 2017, Repsol, 2019, Sanderson, 2019, Shell, 2020, Total, 2020). These arguments come at a critical moment since most countries are currently deciding how fast they should decarbonise their economies. Coal phase-out plans have just started to be adopted in several countries with very different timelines.

This paper assesses the potential for new technologies and plant conversions to reduce the need for assets to be "stranded", i.e. underused or retired early to ensure that global warming remains below 2°C (Caldecott et al., 2016a). We follow Pfeiffer et al. (2018b) and define stranded assets in the power sector in terms of stranded generation: the reduction in electricity generation from fossil fuels arising from the necessary underuse and early decommission of power plants consistent with a 2°C scenario. We firstly estimate the committed electricity generation from operating power plants and those in the pipeline (under construction and planned) and then assess the effect of technologies (CCS, bioenergy, alternative electricity supply and energy efficiency) and plant conversions (from coal to gas or biomass) on the amount of stranded generation.

So far, no study has estimated the effect of technology deployment and plant conversions on the stranding of existing assets, leaving some uncertainty in the required

pace for the energy transition in the power sector. Compared to prior assessments that have focused on calculating committed emissions from existing infrastructure and the impact of climate policy on stranded assets (Davis et al., 2010, Davis and Socolow, 2014, McGlade and Ekins, 2015, Carbon Tracker Initiative, 2017, Mercure et al., 2018, Pfeiffer et al., 2018b, Tong et al., 2019, Binsted et al., 2020), this paper accounts for the possible response of the industry towards developing technological solutions when facing the risk of assets being stranded. Not accounting for this response may overestimate the threat that existing fossil fuel infrastructure could pose to reaching the 2°C target of the Paris Agreement. Furthermore, looking at differences in assumptions regarding technology development allows us to reconcile differences in results across different models of long-term energy generation.

Our results suggest that, even in the presence of a strong industry response to develop low-carbon and negative emission technologies or convert current and planned assets to be less carbon-intensive, the expected amount of required stranding would remain substantial. These results are robust across commonly-used forecasts of energy generation in the 21st century.

4.2 Estimates of electricity generation and stranded generation

We estimate the amount of stranded generation by comparing the amount of electricity forecast to be produced by power plants currently operating, under construction and planned, with energy production pathways consistent with a 2°C target at the end of the century. Firstly, we compile generator-level power plant data globally by merging S&P Global Platt's World Electric Power Plants database; Global Coal Plant Tracker and World Resource Institute's Global Power Plant database. We use the compiled dataset to estimate the future electricity generation from the power plants that are currently operating and in-the-pipeline.

Our estimates for the potential electricity generation from operating and in-the-pipeline fossil fuel power plants are provided in Figure 4.1. Using baseline assumptions for utilisation rates and lifetimes (see baseline assumption details in Section 4.6.2), we estimate that these power plants could produce 540 PWh in total between 2021 and 2100. As shown in Figure 4.1 (a), about two-thirds of this electricity would come from coal-fired power plants, and the other third from gas-fired power plants. Only a negligible amount would come from oil-fired power plants. The currently operating plants would produce slightly more electricity than the plants in the pipeline. Figure 4.1 (b) shows that around 60 percent of this electricity would come from Asia, and 20 percent from the OECD. The remaining 20 percent is shared between the Middle East, Africa, Latin America and Reforming Economies.

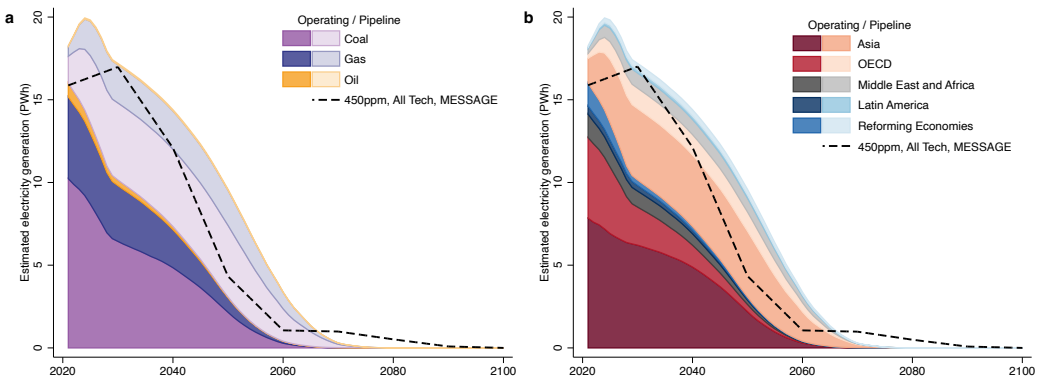


Figure 4.1: Estimated electricity generation, by fuel (a) and by region (b)

We estimate the amount of future electricity that can be generated from currently operating power plants and those in the pipeline. Power units are assumed to be operating at the same utilisation rates as those forecasted in the IEA Stated Policy Scenarios (with a breakdown by category of power units described in Table 4.4) until the end of their lifetime. Darker shading indicates the expected generation from currently operating plants, while lighter shading indicates the expected generation from in-the-pipeline plants. The black dashed line is an example of the electricity that could be produced based on one scenario from one IAM, specifically the 450ppm, "all technologies deployed" scenario obtained from the MESSAGE model. All the electricity generation showed in this figure is from fossil fuel plants without CCS.

Secondly, we compare the forecast of electricity generation of Figure 4.1 with

scenarios consistent with a 2°C target. The scenarios are extracted from the AMPERE database, which gathers model outputs from different Integrated Assessment Models (IAMs) and assume that GHG emissions stabilise at 450ppm by the end of the century. This is widely consistent with achieving the 2°C target in the literature (Meinshausen et al., 2011, Riahi et al., 2015). The general characteristics, socioeconomic drivers and key energy technology assumptions of these IAMs are presented in Table 4.2.

Stranded generation are simply computed as the difference between the electricity generation allowed in the climate scenarios, and the electricity generation that operating and in-the-pipeline power plants could produce at baseline utilisation rates and lifetimes (see baseline assumption details in Section 4.6.2).

An illustration is provided in Figure 4.1. The black dashed lines correspond to one of the scenarios available in the AMPERE Working Package 2 (WP2) database. This scenario assumes that all low-carbon technologies available in the model can be deployed. In this case electricity generation from currently operating plants is roughly within the example's modelled boundary. Nevertheless, adding the electricity generation from in-the-pipeline plants exceeds the scenario's modelled boundary, signalling the stranding risk. Other pathways that do not assume availability of all technologies may lead to higher levels of stranded generation. Furthermore, stranding risk varies by fuel type, with stranding risk for coal-fired power plants much stronger than for other fuels. Separate pathways by fuel type for the same example are provided in Figure 4.6 in the Supplementary Information.

We are particularly interested in the AMPERE scenarios that assume the four technologies are fully deployed: 1) CCS; 2) bioenergy; 3) solar and wind; and 4) nuclear energy, which we refer to as the "all technologies deployed" scenarios. Table 4.3 provides more detail on the definition of the "technologies fully deployed" scenarios compared to other scenarios. In a nutshell, no pre-set constraint is imposed on the function of technology deployment. In other scenarios, limits are usually

imposed, whether the potential is limited (e.g. bioenergy limited to 100 EJ per year or solar and wind are limited to 20 percent of total power generation) or the technology is not allowed (e.g. CCS is not available and no new investments in nuclear power after 2020). The "all technologies deployed" scenarios also assume that energy intensity continues to reduce every year at historical rates. Figure 4.7 in the Supplementary Information displays the global electricity generation by technology in the "all technologies deployed" scenarios of each IAM.

Figure 4.2 provides our estimates of stranded generation for the "all technologies deployed" scenarios included in the AMPERE WP2 database. Results are provided for seven Integrated Assessment Models (GCAM, IMACLIM, IMAGE, MESSAGE, POLES, REMIND, WITCH), and for operating and in-the-pipeline plants separately. We furthermore provide the average amount of stranded generation across these seven models. Estimates of stranded generation are displayed at global level, for five regions (Asia, OECD, Middle East and Africa, Latin America and Reforming Economies) and the top two countries by volume of stranded generation (China and India).

Taking the average across the IAMs, Figure 4.2 shows that about 267 PWh electricity, which is about 50 percent of the future electricity generated from currently operating and in-the-pipeline fossil fuel plants, would have to be stranded to achieve the objective of stabilising GHG concentrations at 450ppm, under the assumption that CCS, bioenergy, solar, wind and nuclear energy are fully deployed. More than 50 percent of the stranded generation globally would be located in Asia, outweighing the total of stranded generation in the other four regions. China and India would face averages of 104 and 37 PWh of stranded generation respectively.

About two thirds of the required stranding is linked to electricity that will be generated in plants that have not been built yet but are in the pipeline. A large amount of stranding could therefore be avoided by stopping the construction of the plants that are currently planned or under construction. In Figure 4.8 in the

Supplementary Information we break down our estimates of stranded generation by fuel type. Coal-fired power plants make most of stranded generation (82 percent across all IAMs), followed by gas-fired power plants contributing 16 percent in the across model mean. Due to their limited capacity, oil-fired plants only constitute a very small share of stranded generation.

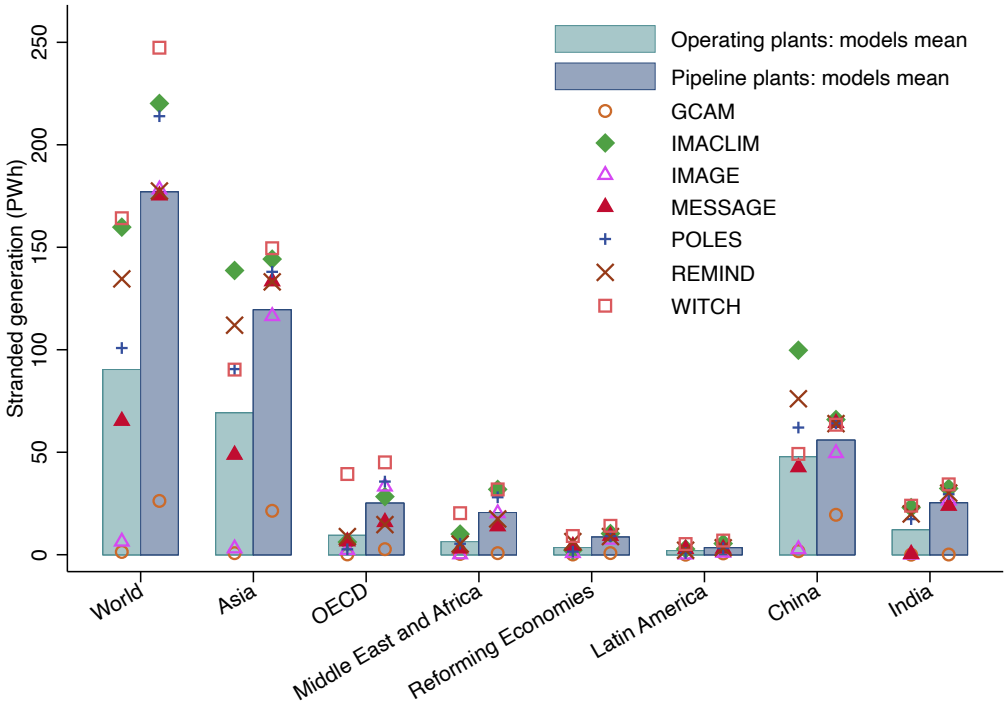


Figure 4.2: Estimated stranded generation in "all technologies deployed" scenarios (2021-2100)

Total stranded generation of fossil fuel power plants (including coal-, gas-, and oil-fired) from 2021 to 2100 in the "all technologies deployed" scenarios of AMPERE consistent with GHG concentrations stabilise at 450ppm. Stranded generation are computed by fuel and then aggregated. We assume no conversion of the currently operating and in-the-pipeline fossil-fuel power plants to CCS, and no conversion of the currently operating and in-the-pipeline coal-fired power plants to bioenergy or gas. Scatter points represent the estimation from individual IAMs and bars show the means across all models.

The results presented in Figure 4.1 and Figure 4.2 are estimated using baseline utilisation rates and lifetimes. We conduct a sensitivity analysis on the utilisation rates and lifetimes assumptions in the Supplementary Information (see Figure 4.9). Higher utilisation rates and longer lifetime assumptions would lead to more future electricity generation and therefore more stranded generation. However, even

though the share of stranded generation increases with longer lifetimes and higher utilisation rates, it remains close to about 50 percent of future electricity generation in most cases.

The amount of stranded generation varies significantly across different IAMs. This variation arises from the substantial differences in structures and assumptions used by IAMs (Farmer et al., 2015, Riahi et al., 2015). The existence of these differences is directly observable in Figure 4.2, where the amount of stranded generation globally is lowest in GCAM (about 28 PWh), and highest in WITCH (nearly 411 PWh). GCAM is the only IAM with very low levels of stranded generation, with all the other models entailing at least 180 PWh of stranded generation. This is due to high levels of bioenergy deployment in GCAM scenarios: electricity produced by biomass with CCS (BECCS) could reach as much as 180 EJ per year in 2100 in GCAM, while other models only forecast production of 6 to 60 EJ per year (see Figure 4.7 in the Supplementary Information). The biomass potential assumed in GCAM is higher than the widely-accepted global sustainable biomass potential of 100 EJ per year (Creutzig et al., 2015). More electricity supplied from biomass with CCS allows more electricity to be generated from fossil fuels, and thus leads to lower stranded generation.

We have compared our AMPERE-derived results to those based on the IPCC SR1.5 database (Huppmann et al., 2018). The IPCC SR1.5 database includes 1.5°C as well as 2°C scenarios. We estimate the amount of stranded generation with this database in Figure 4.10 in the Supplementary Information. The average amount of stranded generation across all models in 2°C scenarios of IPCC SR1.5 is higher by 12 percent compared to the amount in Figure 4.2. This is in part because AMPERE scenarios in Figure 4.2 assume that all technologies capable of reducing stranded generation have been fully deployed, while 2°C scenarios in IPCC SR1.5 have varying assumptions on technology development. The average amount of stranded generation across all models in the 1.5°C scenarios of IPCC SR1.5 is

about 17 percent higher than in the 2°C scenarios of IPCC SR1.5 (see Figure 4.10 in the Supplementary Information).

The rest of this paper is based on the scenarios available in AMPERE, because they allow for the estimation of the effect of technology availability on stranded generation through the pairwise comparison of scenarios with and without a technology (as illustrated in Table 4.3 in Section 4.6.2).

4.3 Impact of plant conversions

Our analysis above suggests high levels of stranded generation under all IAMs apart from GCAM. These estimates may be overly restrictive, however, because they do not consider that power plants could adapt to the risk of stranded generation by converting to use lower-carbon technologies. In Figure 4.3 we allow for power plants to be converted and estimate the impact that this would have on stranded generation (compared to Figure 4.2). We consider three conversion options: 1) conversion from coal to gas; 2) the installation of CCS in existing plants and in those currently in the pipeline; and 3) the use of biomass in coal-fired power plants.

Coal power plants can be modified to operate only on natural gas (conversion); to fire either coal or natural gas (dual fuel); or to fire both coal and natural gas at the same time (co-firing) (Bartnik, 2013, Carapellucci and Giordano, 2015, Bedilion, 2017, Mills, 2018, Stevick, 2019). Converting a coal-fired boiler to gas requires adding new equipment, such as gas igniters, scanners, piping and valves. It also requires a modification of burner management and combustion control systems, an adjustment of pressure-part through the convection pass and a layup of coal and ash handling equipment (Bedilion, 2017, TransAlta, 2020). So far, coal-to-gas projects has been undertaken in more than 80 coal-fired power plants in the US between 2011 and 2019 (EIA, 2020), representing about 5 percent of total US coal-fired power capacity. Other countries such as Canada (TransAlta, 2020) and United

Kingdom (Staff, 2019) also have several coal-to-gas projects at different stages of development.

In Figure 4.3(a), we estimate the change of stranded generation when considering potential conversions from coal to gas. We consider that coal-fired power plants located in countries that also include operating gas-fired power plants are suitable for conversion since they have access to gas infrastructure and supplies. We then assume that between 5 percent and 20 percent of them could be converted. The 5 percent corresponds to the US conversion percentage between 2011 and 2019. 20 percent is an optimistic upper bound based on the possibility that gas is co-fired with coal. In that regard, Domeshek and Burtraw (2021) have suggested that US coal-fired power plants could adopt a 20 percent gas co-firing standard. Because these figures of 5 to 20 percent are derived from the US experience, they are likely to constitute upper bounds for what could happen globally. In the US, natural gas is relatively cheap because of the deployment of shale gas and the significant reduction in gas prices that followed. We furthermore limit possible plant conversions so that the amount of electricity produced from converted plants remains equal or below the electricity produced from gas in the IAMs. Beforehand, we have subtracted the amount already covered by existing gas-fired power plants. In this way, our projections for the impact of plant conversions take into account future forecasts for the use of gas in power generation globally.

Results in Figure 4.3(a) suggest that coal-to-gas conversions may reduce stranded generation by 10 PWh to 30 PWh on average across all models. When we assume a maximum conversion potential of 5 percent of coal-fired power plants, on average 2.8 percent of coal-to-gas conversion occurs in our projections due to the limits imposed by IAMs regarding the use of coal and gas in power generation. When we allow for up to 20 percent of conversions, 10.6 percent of coal-fired generation is converted on average due to the limits imposed by the IAMs.

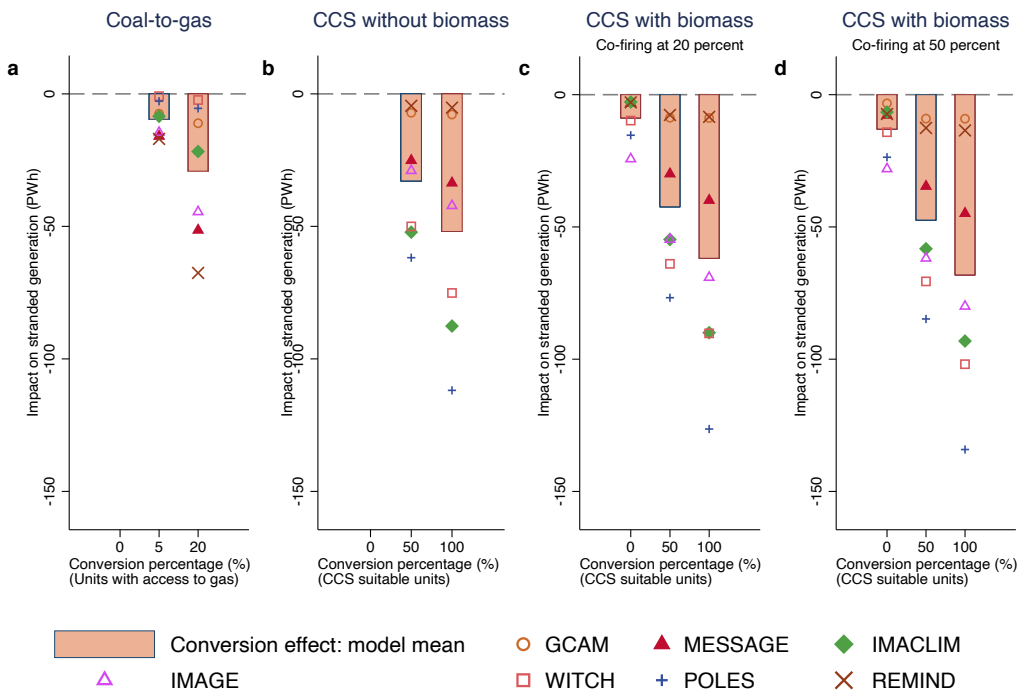


Figure 4.3: Impact of plant conversions on global stranded generation

Scatter points represent the estimation from individual models and bars show the model mean. The conversion percentages of (a) correspond to the share of coal-fired power plants converted to gas. Those of (b) to (d) correspond to the share of CCS suitable plants converted to CCS. Biomass co-firing ratios vary from 0 in (b) to 50 percent in (d).

In Figure 4.3(b), we look at CCS adoption from operating and in-the-pipeline fossil fuel power plants. Caldecott et al. (2016b) assesses that existing generation units are likely to be suitable to CCS installation if they have a capacity above 100MW, are less than 20 years old, emit less than 1000g CO_2/kWh and are located within 40km of geological areas suitable for CCS. Using these criteria, we find that around 24 percent of operating and in-the-pipeline fossil fuel power units would be suitable for CCS conversion. In Figure 4.3(b), we show the impact of CCS conversion on stranded generation if either half of those suitable capacities, or all of them, would be equipped with CCS. We furthermore make sure that CCS installation in existing and in-the-pipeline power plants coincides with the take-up of CCS in the IAMs. Therefore, we bound our estimates of electricity generated from converted plants with CCS to be equal or inferior to the total volume of electricity generated from fossil fuels with CCS in the IAMs. The results displayed in Figure 4.3(b) imply that stranded generation can be reduced by 33 PWh to 52 PWh thanks to the future installation of CCS in operating and in-the-pipeline plants. Reductions are most pronounced in the IAMs in which we found the largest amounts of stranded generation (i.e. POLES, IMACLIM and WITCH).

In Figure 4.3(c) and (d), we consider that coal-fired power plants (with and without CCS) could be adapted to co-fire biomass. So far, more than 150 power plants have fired coal along with biomass, with the majority located in northern Europe and the US (Fernando, 2012). However, in the IAMs, 95 percent of future electricity generation with biomass comes from plants equipped with CCS. Our calculations therefore have to consider the co-firing of biomass in some plants that are not equipped with CCS, and the co-firing of biomass in plants that would be retrofitted with CCS.

Figure 4.3(c) and (d) consider three scenarios of CCS retrofits: no plants are converted to CCS; 50 percent of CCS suitable capacities are converted to CCS; and all CCS suitable capacities are converted to CCS. The last two assumptions

are the same as in Figure 4.3(b). Furthermore, we allow coal plants to co-fire with biomass. Co-firing ratios can vary. The IEA and IRENA (2013) estimate that a 20 percent co-firing ratio is feasible in most cases, while a 50 percent co-firing ratio is technically achievable. Figure 4.3(c) assumes a co-firing ratio of 20 percent and Figure 4.3(d) a co-firing ratio of 50 percent.

In both figures, the coal-to-biomass potential is restricted to the availability of sustainable biomass feedstock within each IAM. Co-firing without CCS cannot exceed the electricity generated from biomass without CCS in the IAMs, after accounting for the electricity that would already be generated from existing biomass plants. Furthermore, biomass co-firing has to be equal or inferior to either 20 percent or 50 percent of the total of energy generated from converted plants: there cannot be more biomass co-fired than the amount of coal allows for.

In the absence of any CCS conversion, Figure 4.3(c) suggests that biomass co-firing could reduce the amount of stranded generation by 9 PWh when assuming a 20 percent co-firing ratio. With the installation of CCS and 20 percent co-firing, stranded generation could be reduced by 42 to 62 PWh, depending on whether half or all CCS suitable plants would be converted to CCS. The contribution of biomass co-firing is relatively small, at between 9 and 10 PWh, since the installation of CCS alone already allows for a reduction in stranded generation by 33 PWh to 52 PWh (as per Figure 4.3(b)). In Figure 4.3(d), with a 50 percent co-firing ratio and no CCS conversion, stranded generation could be reduced by 13 PWh. With a 50 percent co-firing ratio and CCS conversions, stranded generation could be reduced by 47 to 68 PWh in total, of which 14 to 16 PWh would come from biomass co-firing.

Therefore, we find that coal to gas conversions may reduce stranded generation by 10 to 30 PWh, and CCS and biomass together by 33 to 68 PWh. However, the estimates for coal to gas (Figure 4.3(a)) have been calculated separately to the other conversion options (Figure 4.3(b) to (d)) and cannot be directly added (see Section 4.6.2 for more details). Furthermore, there are differences in impact across models

that arise from model settings on energy conversion technologies, energy technology choices, substitutability and deployment (see Table 4.2 for more details).

4.4 Impact of energy demand, alternative electricity sources, CCS and bioenergy availability

Estimates in Figure 4.2 and 4.3 suggest that fast low-carbon technology deployment, enhanced by plant conversions, could mitigate the risk of stranding. Yet, the previous estimates have assumed that four technologies (CCS, bioenergy, solar and wind, and nuclear energy) will be available and fully deployed.

Several scenarios within and outside the AMPERE database make less optimistic assumptions about technology take-up. For example, the IEA's Sustainable Development Scenario (IEA, 2020a) assumes a late take-up of biomass with CCS (see Figure 4.11 in the Supplementary Information). Between 2021 to 2050 (when most asset stranding would happen for existing and in-the-pipeline plants in our database), we can calculate that the IEA scenario corresponds to a total energy production from biomass with CCS of only 4 PWh. In the "all technology deployed" scenarios of AMPERE, this technology generates an average across IAMs of 40 PWh between 2021 and 2050.

Below, we discuss the possible impact of low or late technology diffusion on the amount of stranded generation. In Figure 4.4, we make different assumptions regarding technology availability. We compare the amount of stranded generation in the "all technologies deployed" scenarios with the amount of stranded generation in scenarios that assume one of the following four technologies is unavailable or insufficiently developed: 1) CCS; 2) bioenergy; 3) solar and wind; and 4) nuclear energy. We furthermore look at a scenario in which energy intensity reduces faster,

at 1.5 times of the historical rates. Energy demand is therefore lower, especially at the end of the century. The model-specific estimates (scatter points in Figure 4.4) are directly comparable to those presented in Figure 4.2. However, the averages value across IAMs (the bars in Figure 4.4) are not comparable because not all IAMs have run the scenarios necessary to make technology-specific pairwise comparisons. The impact of the availability of CCS, for example, can only be computed for GCAM, REMIND and MESSAGE.

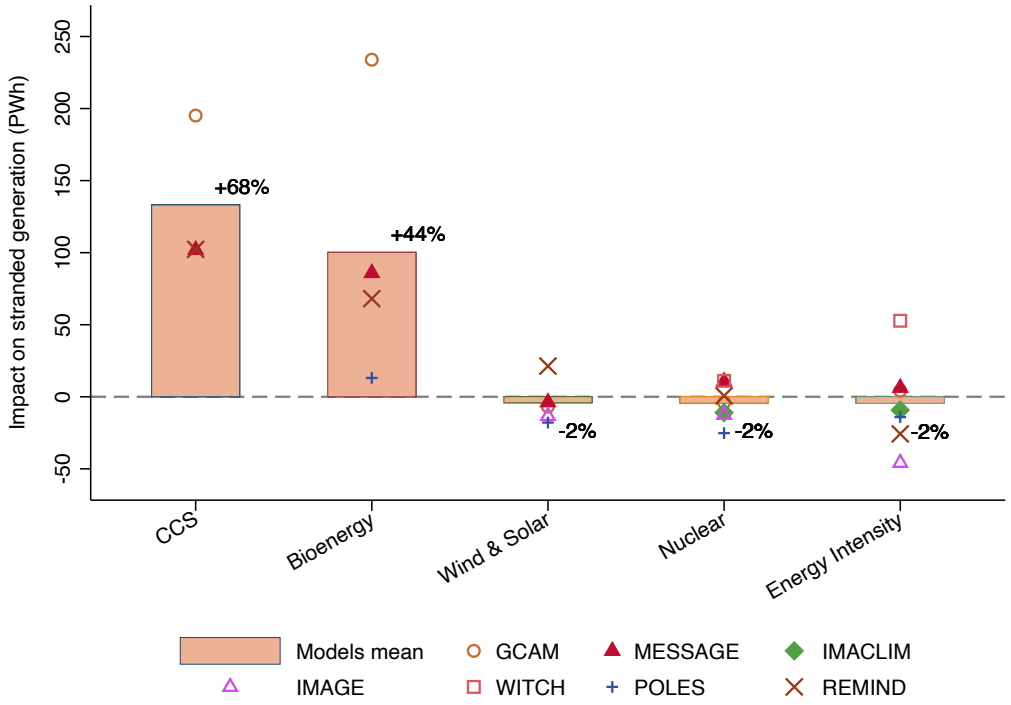


Figure 4.4: Impact of technology availabilities on global stranded generation

Impacts are computed from the difference in stranded generation between the "all technologies deployed" scenarios (results shown in Figure 4.2and the scenarios with one technology insufficiently deployed. Positive effects suggest that the amount of stranded generation increases when a particular technology is unavailable or insufficiently deployed. For energy intensity, the impact represented the difference between the scenarios where the energy intensity reduces at historical rates and 1.5 times historical rates. Scatter points represent the estimation of individual models and bars show the average across all IAMs. The percentage values around each bar represent the change of stranded generation in scenarios with one technology insufficiently developed compared to the "all technologies deployed" scenarios.

Figure 4.4 shows that the unavailability of CCS and bioenergy (mostly combined with CCS) has a significant impact on the amount of stranded generation, increasing

by 69 percent and by 45 percent respectively on average. This is because CCS prevents emissions from going to the atmosphere, while bioenergy coupled to CCS provides a mechanism for negative emissions.

In contrast, the other low-carbon technologies studied (wind, solar and nuclear power), as well as a further reduction in energy intensity, have ambiguous impacts. Some IAMs suggest that they would reduce the amount of stranded generation. This is possible because they reduce the carbon intensity of the electricity supply, allowing for more electricity to be generated from fossil fuels. Other IAMs suggest that the development of these technologies would in fact increase the amount of stranded generation. This could be the case as wind, solar and nuclear energy compete with fossil fuels for meeting electricity demand. Greater reduction in energy intensity could reduce electricity demand and therefore lower the need for fossil fuel power plants.

A corollary finding to Figure 4.4 is that no IAM implies low amounts of stranded generation unless they assume strong deployment of CCS or bioenergy. In the absence of CCS, GCAM, REMIND and MESSAGE – three models with relatively low amounts of stranded generation in Figure 4.2 – would record much higher amounts of stranded generation (by 200 PWh for GCAM and 100 PWh for MESSAGE and REMIND). The amounts of stranded generation obtained with these models are also impacted by limiting bioenergy. Especially, GCAM assumes very large amounts of bioenergy combined with CCS in its "all technologies deployed scenario". If bioenergy is not fully deployed, the amount of stranded generation in GCAM increases by 230 PWh.

4.5 Discussion

Overall, our results underline a clear stranding risk for investors, plant operators and policymakers. Even if CCS and bioenergy are deployed quickly and extensively

in the 21st century, an average 267 PWh of electricity generation (the value ranges from 28 to 411 PWh across models) could be at risk of stranding under a 2°C target. If some operating and in-the-pipeline coal power plants were converted to gas, stranding could be reduced by up to 30 PWh. The adoption of CCS and the co-firing of biomass may reduce stranding by 33 to 68 PWh, depending on the share of CCS-suitable plants finally converted, and the assumed co-firing ratio.

Our analysis comes with limitations. Although we have collected the best available data, our unit-level power plant dataset covers 88 percent of world's installed capacity in 2018. We are likely to underestimate electricity production, and therefore the amount of stranded generation. Furthermore, there is missing data on key variables like the online year and retirement year of power plants. Additionally, our analysis assumes that sufficient plants can be converted to meet an emissions budget. In reality, there may be technical and economic barriers at the individual plant level which mean our estimate of the potential to reduce stranded generation may be overestimated. Also, even though we provide lower- and upper-bound estimates for the reduction in stranded generation from plant conversions, there is much uncertainty regarding the potential and pace of future conversions.

Furthermore, our estimates of stranded generation rely on IAMs, which have long been criticised for the way they handle uncertainty, relative prices, behaviour and other factors (Pindyck, 2013, Stern, 2013, Burke et al., 2015, Farmer et al., 2015, Iyer et al., 2015). A recurrent problem in the literature using IAMs is that models tend to provide divergent results due to their different model settings (Kriegler et al., 2015, Riahi et al., 2015, Gillingham et al., 2008). In our analysis, the results vary significantly across IAMs. Therefore, our analysis reconciles the findings from different models. In that regard, these results provide insight as to the direction and the order of magnitude of the impact of CCS and bioenergy on stranded generation.

The amount of electricity generation at risk of stranding would rise by 69 percent and 45 percent respectively if either CCS or bioenergy are not deployed to their full

potential. These technologies with the greatest potential to reduce stranding are the ones that face the strongest challenges to their large-scale deployment. Despite some pilot and demonstration projects (Global CCS Insitute, 2019), the deployment of CCS is still hampered by the high cost of early demonstration projects and availability of storage sites (Scott et al., 2013, Reiner, 2016). Similarly, the deployment of bioenergy is experiencing challenges in feedstock availability and economical cost (Smith et al., 2016), and facing disputes over sustainability, because of risks of deforestation, concerns over food security and potential biodiversity loss (Creutzig et al., 2015, Ember, 2019). This suggests that asset exposure to stranding is likely to be much higher than estimated in the "all technologies deployed" scenarios.

The most immediate option to reduce the risk of assets being stranded and investments being lost is to fully account for stranding risks now. Because decisions have to be made today, it may be unwise to rely on optimistic forecasts of technology deployment in the future. Most of the risk of stranding assessed in this paper relates to plants that are currently in the pipeline. If companies continue to invest in fossil fuel-based infrastructure, some of these assets risk stranding even before they are built. This puts stakeholders and policymakers in a situation in which precaution would suggest that very little or no new fossil fuel power plants can be commissioned, and existing plants may have to retire early or reduce their utilisation rate substantially.

4.6 Data and method

4.6.1 Data

Our analysis uses two categories of data: (1) global power plant data; (2) climate scenario data.

Power plant data

We use a global generator-level power plant dataset to estimate the future electricity generation from power plants that are currently operating and in the pipeline (i.e. under construction and planned). We compile generator-level data by merging the following databases: (a) Global Coal Plant Tracker (January 2019) (Endcoal, 2019); (b) SP Global Platt's World Electric Power Plants (WEPP) database (Q4 2019) (SP Global Market Intelligence, 2020); (c) WRI's global database of power plants (June 2019) (World Resources Institute, 2019). Merging several sources gives us a better coverage of power plants than using only one database. For example, WEPP is known to have imperfect coverage of microgeneration and Chinese power generators, which is compensated by the data in the Global Coal Plant Tracker and WRI's database. An overview of the generator-level power plant data can be found in Table 4.1. The capacity of currently operating generators in our dataset aggregates to 6349 GW, covering 88 percent of the global electricity capacity in the year 2018 (IEA, 2019). The sources are merged manually by confirming power generator name, location, current status, online year and capacity. Online year refers to the year in which the power generator started operating. We supplemented this effort with internet searches wherever necessary. Where matched generators have conflicting fields (for example different statuses) the most recent data are used. We follow the method developed by Caldecott et al. (2018) to use available information from similar clusters (i.e. power generators that burn the same fuel, use the same type of generator and turbine, belong to the same capacity range) to complete missing data on key variables (e.g. online year and expected lifetime).

Table 4.1: Overview of power generators in our dataset

	Total capacity (GW)	Average online year	Number of generation units
Panel A:	In operation in 2019		
Coal	2217	1994	9924
Gas	1495	1997	28423
Oil	436	1992	29296
Bio gas	19	2000	2477
Biomass	36	2004	4082
Bio oil	10	1994	419
Geothermal	15	2000	642
Nuclear	414	1989	444
Solar	128	2012	15581
Waste	339	2003	6759
Hydro	1229	1981	51633
Unknown	11	1982	1088
All	6349		
Panel B:	In pipeline in 2019		
Coal	959	2022	2094
Gas	682	2022	2840
Oil	42	2022	696
Bio gas	4	2022	213
Biomass	13	2022	582
Bio oil	1	2022	25
Geothermal	11	2022	290
Nuclear	295	2023	268
Solar	159	2022	2711
Waste	87	2022	971
Hydro	688	2022	9066
Unknown	0.4	2023	15
All	2941		

Climate scenario data

We use climate scenarios from the AMPERE database to determine the evolution of electricity generation throughout the end of the century. AMPERE is a modelling comparison project that integrates outputs from different Integrated Assessment Models (IAMs) to improve understanding of possible scenarios toward climate targets. AMPERE stands for "Assessment of Climate Change Mitigation Pathways and Evaluation of the Robustness of Mitigation Cost Estimates".

Particularly, we use the AMPERE Working Package 2 (WP2) database, which provides pathways of electricity generation given different technology availabilities (Riahi et al., 2015). AMPERE WP2 compiles a set of around 400 scenarios to 2100, built upon 7 IAMs (GCAM, IMACLIM, IMAGE, MESSAGE-MACRO, POLES, REMIND, and WITCH) that have been published in peer-reviewed journals. The AMPERE WP2 database also includes scenarios modelled by DNE21+ and MERGE-ETL models. However, these two models have been excluded in our analysis as DNE21+ only models the period through to 2050 and MERGE-ETL only models scenarios of "OECD".

Table 4.2 presents the general characteristics (panel a), socioeconomic drivers (panel b) and key energy technology assumptions (panel c) of these IAMs. The information is compiled from IPCC (2014) and Kriegler et al. (2015). As a modelling comparison project, AMPERE harmonised GDP and population assumptions across models to facilitate the analysis of model differences. Nevertheless, models differ in numerous ways and lead to varying results, as showed in our results. Additional details on the models used in AMPERE project are provided in Kriegler et al. (2015).

Table 4.2: Overview of IAM models used in this study

a) General characteristics

Model	Institution	Reference	Equilibrium type	Modelling approach	Time horizon
GCAM	JGCRI, US	JGCRI (2010)	Partial	Recursive dynamic	2100
IMACLIM	CIRED, France	Sassi et al. (2010)	General	Recursive dynamic	2100
IMAGE	PBL, Netherlands	Bouwman et al. (2006)	Partial	Recursive dynamic	2100
MESSAGE	IIASA, Austria	Messner et al. (2000)	General	Intertemporal optimization	2100
POLES	EC-JRC, Belgium	IPTS (2010)	Partial	Recursive dynamic	2100
REMIND	PIK, Germany	Leimbach et al. (2010)	General	Intertemporal optimization	2100
WITCH	FEEM, Italy	De Cian et al. (2009)	General	Intertemporal optimization	2100

b) Social economic drivers

Model	Exogenous drivers	Endogenous drivers
GCAM	Population, GDP, labour rate, labour productivity	None
IMACLIM	Labour productivity, energy technical progress, population, active population	None
IMAGE	Exogenous GDP, GDP per capita, population	Energy demand, renewable price, fossil fuel prices, carbon prices, technology progress, energy intensity, preferences, learning by doing, agricultural demand, value added
MESSAGE	Labour productivity, energy technical progress, GDP per capita, population	None
POLES	Exogenous GDP, population	Value added, mobility needs, fossil fuel prices, buildings surfaces
REMIND	Labour productivity,energy efficiency parameters of the production function, population	Investments in industrial capital and specific energy technology capital, endogenous learning-by-doing for wind and solar power and electric and fuel cell vehicle
WITCH	Total factor productivity, labour productivity, capital technical progress	None

Table 4.2:(continued)

c)Key energy technology assumptions

Model	Energy Conversion technologies	Deployment and substitutability
GCAM	CHP, hydrogen from coal, oil, gas, and biomass, nuclear and solar thermochemical, fuel to gas, coal to gas, biomass, fuel to liquid, coal to liquids, gas to liquids, biomass to liquids	Logit choice model with usually high substitutability
IMACLIM	Fuel to liquid	Logit choice model, high and low substitutability depending sectors, expansion and decline constraints, system integration constraints
IMAGE	CHP, hydrogen	Logit choice model, mostly high substitutability, expansion and decline constraints,system integration constraints
MESSAGE	CHP, hydrogen, fuel to gas, fuel to liquid	Linear choice, mostly high substitutability, expansion and decline constraints, system integration constraints
POLES	CHP, hydrogen, fuel to liquid	Logit choice, mostly high substitutability, expansion and decline constraints
REMIND	CHP, Heat pumps, hydrogen, fuel to gas, fuel to liquid, heat plants	Discrete technology choices with high to full substitutability, expansion and decline constraints, system integration constraints
WITCH	None	No discrete technology choices, mostly high substitutability, expansion and decline constraints, system integration constraints

Table 4.2, panel c, shows that plant conversions from coal to gas or coal to biomass are not included in the calculations of electricity generation by fuels in the IAMs, except for GCAM. Probably, this implies that GCAM might allow for more coal at the beginning of the century, since generation can be converted into gas or biomass. However, it is difficult to determine the separate impact of this assumption, since other assumptions in the models may also create differences in electricity generation levels by fuels in the IAMs. Due to the heterogeneity of assumptions on technology development, it is essential to rely on a set of different IAMs to obtain robust results.

The AMPERE WP2 database models the availability of four low carbon technology options: carbon capture and storage, solar and wind, bioenergy and nuclear energy. It furthermore assumes that energy intensity reduces at either historical

rates or 1.5 times historical rates. The details of the technology dimensions are presented in Table 4.3. The baseline scenario is an "all technologies deployed" scenario that assumes that all four technologies are fully available and energy intensity reduces at historical rates. Figure 4.7 in the Supplementary Information displays the global electricity generation by technology in the "all technologies deployed" scenarios of each IAM. Alongside, the AMPERE database provides other scenarios that assume each one of the four technology options is either unavailable or of limited availability. Another scenario assumes that all four technologies are available and energy intensity reduces faster, by 1.5 times historical rates. Comparisons between the "all technologies deployed" scenario and these limited technology scenarios allow us to estimate the impact of each technology availability on stranded generation.

Scenarios in AMPERE WP2 are constructed for different levels of climate policy stringency. We use the "450 Optimal" scenarios, which require greenhouse gas levels to stabilise at 450ppm CO_2-equivalent. This level of stabilisation corresponds to GHG emission scenarios in the literature widely consistent with keeping long-term temperature rise below 2°C compared to pre-industrial levels (Meinshausen et al., 2011, Riahi et al., 2015). The 450ppm scenarios in the AMPERE WP2 database correspond to a cumulative CO_2 emission budget of 1500Gt CO_2 from 2000 to 2100. "Optimal" indicates that global emissions follow an optimal pathway in which required policies are introduced immediately to meet the long-term emissions budget and no explicit short-term target for 2030 is assumed (Riahi et al., 2015).

AMPERE WP2 scenarios include data at regional and country levels. The world is divided into five regions: Asia, Latin America (LAM), Middle East and Africa (MAF), OECD countries (OECD) and the Reforming Economies of the former Soviet Union (REF).

The regional data make it possible to estimate stranded generation at the regional level and then aggregate to the global level, which provides a more accurate

Table 4.3: Technology settings in AMPERE scenarios

Technology dimensions	All technologies deployed scenarios	With one technology insufficiently deployed scenarios
Carbon Capture and Storage	CCS is fully available.	CCS never becomes available, including for both fossil fuel and bio-based plants.
Nuclear	Nuclear energy is fully available.	No new investments into nuclear power after 2020; existing plants are phased out over their lifetime.
Solar and wind	Advanced* techno-economic assumptions for solar and wind.	Limited contribution of solar and wind to 20 percent of total power generation, reflecting implementation barriers of renewable energy at high penetration rates.
Bioenergy	Total global bio-energy supply shall top out at the level generated endogenously by each model.	Total global bio-energy supply for all sectors from purpose-grown crops, residues and municipal solid waste shall be limited to 100 EJ/year as primary energy.
Other setting		
Energy intensity	Energy intensity improves at historical rates.	A combination of efficiency measures and behavioural changes leads roughly to a 50 percent increase of the energy intensity improvement rate compared to historical rates.

Source: AMPERE Working Package 2 model comparison study protocol and Riahi et al. (2015).
: It is left to the modeller's choice what is being considered "advanced".

global estimation as this does not make assumptions about exchanges of electricity between regions. If we calculated stranded generation at the global level, this may assume inter-regional exchanges of energy (e.g. between Europe and Asia) and this may mask regional stranding. The global level estimated for stranding might be lower than the sum of all regions. For example, while electricity generation from coal in OECD goes down, it may go up in Asia. This amount of reduced electricity generation in the OECD would not be stranded if it powered the Asian market, as assumed with aggregated values. On a regional or country level analysis, however, this amount would become stranded. Therefore, we calculate stranded generation at regional level first and then aggregate them at global level when using climate scenarios from the AMPERE database.

AMPERE WP2 also includes country-level scenarios (or clusters of countries): China, Brazil, India, USA, Japan, Russia and EU. We use these data to produce complementary country-level analyses.

4.6.2 Methods

We proceed in four steps, combining data-driven method and IAM results. Firstly, we estimate the future electricity generation from currently operating and in-the-pipeline plants (results showed in Figure 4.1). Secondly, we compute stranded generation for each scenario as the difference between the future electricity generation from operating and in-the-pipeline plants, and the electricity generation in the AMPERE 450ppm scenarios (results showed in Figure 4.2). Thirdly, we recalculate stranded generation after accounting for the fact that operating and in-the-pipeline plants could be converted into less polluting assets, for example by converting a coal-fired power plant into a gas-fired power plant (results showed in Figure 4.3). Finally, we compare stranded generation across technology scenarios to evaluate the impact of technology availability on stranded generation (results showed in Figure 4.4).

The advantage of our method is that it allows us to understand stranded generation by using results from different IAMs. IAMs are constructed based on different structures and assumptions, therefore results from different IAMs usually differ substantially and are not directly comparable (Kriegler et al., 2015, Riahi et al., 2015). As we have several models corresponding to each scenario, we have several estimates for assessing stranded generation in each scenario. To address this, we present the mean value across different IAMs as well as individual model results in our figures.

Step 1: Estimate future electricity generation from operating and in-the-pipeline plants

We use our power plant data and follow the approach developed by Pfeiffer et al. (2018b) to estimate how much future electricity demand can be met from current plants. We estimate yearly electricity generation from currently operating plants and those in the pipeline (planned and under construction) at the generator level and then aggregate it to the country level, the regional level and globally. We compute the electricity generation from each generator i in year t as follows:

$$Generation_{it} = Capacity_i * 24 * 365 * utilisation_{it} \tag{4.1}$$

Where $Generation_{it}$ is the annual electricity generation of generator i in year t; $Capacity_i$ is the maximum hourly capacity of generator i (which we multiply by 24 hours times 365 days); and $Utilisation_{it}$ is the utilisation rate of generator i in year t. We do not know the exact starting date or retirement date of a generator within the year. Therefore, in the online years and retirement years, we simply assume that generators are operating for six months (the utilisation rates take half values as in other years).

Utilisation rates. In the absence of any climate constraint, we assume that currently operating plants and those in-the-pipeline would follow the utilisation

rates assumed in IEA scenarios. We extract global fuel-specific load factors (i.e. utilisation rate) for power generators from IEA's World Energy Outlook 2019 (IEA, 2019) and IEA World Energy Outlook in 2005-2019 for historical data and present them in Figure 4.5. The IEA Current Policies Scenario presents what happens if business continues as usual, without any additional policy changes, while the Stated Policies Scenario considers today's policy intentions and targets. No significant differences are found between future utilisation rates in these scenarios. We see a clear decreasing trend in historical utilisation rates of coal- and oil-fired power plants, while gas-fired power plants saw a slight increase from 2004 to 2018.

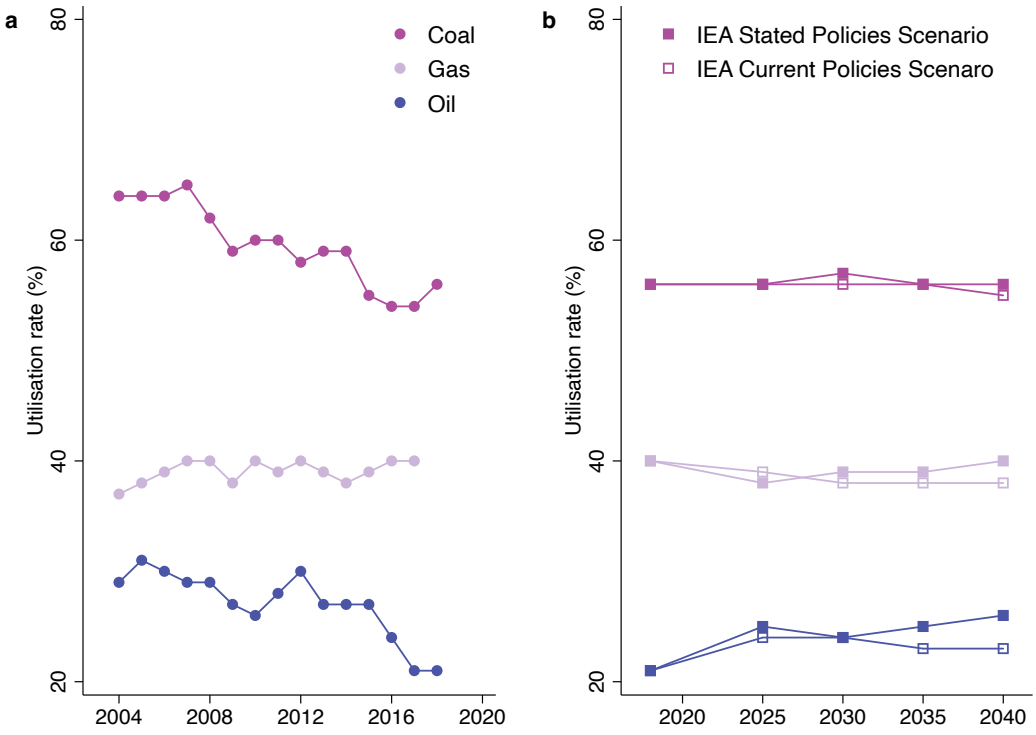


Figure 4.5: Historical and future utilisation rates in IEA World Energy Outlook 2019 for future rates and 2005-2019 for historical rates

Therefore, for our main results, we have assigned the average value of utilisation rates in Stated Policies Scenarios to each individual generator based on its fuel type (as presented in Column (1) in Table 4.4). The assigned utilisation rates for coal-, gas-, oil- and biomass-fired power generators in the baseline are 56%, 39%, 23% and

55%, respectively. In Figure 4.9 in the Supplementary Information, we furthermore show the sensitivity of our results to the choice of utilisation rates by using the maximum and minimum historical utilisation rates (as presented in Columns (2) and (3) in Table 4.4).

Table 4.4: Assumed utilisation rates

	Baseline	Minimum	Maximum
Coal	56%	54%	65%
Gas	39%	37%	40 %
Oil	23%	21%	31%

Note: The baseline represents the average value of utilisation rates in the Stated Policies Scenarios from IEA World Energy Outlook 2019. The maximum and minimum cases use the maximum and minimum historical utilisation rates in the years 2004-2018 respectively, extracted from IEA World Energy Outlook 2005-2019.

Lifetime of generators. Since we are interested in total future electricity generation until the end of the century, we need to make assumptions about when currently operating and in-the-pipeline generators will retire. Only about 5 percent of power generators in our dataset contain information on when the generator is expected to be retired. For the other ones, we simulate their lifetime based on the retirement year of similar generators. We consider that generators are of same type if they use the same fuel, unit technology type and steam type (usually either subcritical, supercritical, and ultra-supercritical steam conditions), are in the same capacity range (under 400 MW, from 400 to 500 MW, from 500 to 700 MW, from 700 to 900 MW and beyond 900 MW), and started to operate in the same year. We use the most disaggregated levels of technology types available. For example, coal-fired power plants include coal, coke, syngas from gasified coal, synthetic gas from petroleum coke, coke-oven gas, coal steam gas, coal-water slurry, corex process offgas, etc. Then, we assume that the lifetime of power generators of a given type to follow a Poisson distribution with mean value equal to the average known lifetime of generators of same type.

For the generators that are already older than their estimated lifetime but are

still operating, we assume that these power generators will progressively retire within the next 10 years. We assign them a retirement date depending on their age. To do so, we rank these generators based on their age, from the oldest (first) to the youngest (last). We then multiply their ranking, e.g. 15th oldest out of 42 (15/42), by 10 years to assign them with a remaining lifetime (in this case 15/42 times 10 is equal 3.57 years).

We compare our assumptions of the lifetime of generators with the assumptions used previously in the literature in Table 4.5. Column (1) shows the average lifetimes of operating and pipeline fossil fuel power generators, which are assumed in our baseline. Columns (2) to (4) list the lifetime assumed in key literature (Tong et al., 2019, IRENA, 2017).

With these baseline assumptions for lifetime, we provide estimates for the global yearly electricity generation levels in Figure 4.1. We further show the sensitivity of our estimates to the assumptions of lifetime in Figure 4.9 in the Supplementary Information. In Figure 4.9, the "extend" case assumes all fossil fuel power generators extend their lifetime by 10 percent compared to what we assumed in our baseline estimations while, in the "reduce" case, the expected lifetime is reduced by 10 percent.

Table 4.5: Comparison of lifetime assumptions (Years)

Fuel	Average lifetime assumed in our baseline	Average lifetime in Tong et al. (2019)	Technical lifetime in IRENA (2017)	Economic lifetime in IRENA (2017)
Coal	40	36	50	25
Gas	39	37	30	15
Oil	39	34	50	25

Step 2: Estimating stranded generation for each climate scenario

We define stranded generation as the difference between the future electricity generation from currently operating and in-the-pipeline plants and the electricity generation levels as given in the different scenarios of the AMPERE database. Therefore,

our estimates of stranded generation are scenario-specific and depend on the assumed climate target (450ppm) as well as the technologies available in the future to produce electricity. Furthermore, because the AMPERE database covers the results of different IAMs for each specific scenario, we have as many estimates as we have IAMs.

For each AMPERE scenario, we estimate the amount of stranded generation by fossil fuel types (coal, gas, oil) in each year separately and then sum up the total fossil fuel stranded generation over the period 2021-2100. The AMPERE database provides us with the yearly amounts of electricity production from fossil fuels with and without carbon capture and storage in each scenario. In our estimates in Figure 4.2, we assume no conversion of any operating or in-the-pipeline fossil fuel power plants to CCS. Therefore, we use the electricity production from fossil fuels without carbon capture and storage in the IAM scenarios to compute the amount of stranded generation.

If the electricity generation estimated from operating and in-the-pipeline plants is larger than the electricity generation assumed in the climate scenario, then the difference corresponds to the amount of stranded generation for this year; otherwise, there is no stranded generation for this year in this scenario. We consider electricity generation from currently operating plants to supply the energy demand first, as these plants are already built.

Precisely, we estimate the stranded generation of fuel r for region/country c in year t as follows:

$$SA_{rct,oper} = \max(Generation_{rct,oper} - Scenario_{rct,oper}, 0) \tag{4.2}$$

$$SA_{rct,pipe} = Generation_{rct,pipe} \text{ if } SA_{rct,oper} > 0 \tag{4.3}$$

$$\begin{aligned} &SA_{rct,pipe} = \max(Generation_{rct,pipe} + Generation_{rct,oper} - \\ &Scenario_{rct,woCCS}, 0) \text{ if } SA_{rct,oper} = 0 \end{aligned} \tag{4.4}$$

$$SA_{rct} = SA_{rct,pipe} + SA_{rct,oper} \tag{4.5}$$

Where $SA_{rct,oper.}$, $SA_{rct,oper}$ and SA_{rct} are the estimated stranded generation of operating, in-the-pipeline and total plants of fuel r for region/country c in year t, respectively. $Generation_{rct,pipe}$ and $Generation_{rct,oper}$ are the electricity generation of operating and pipeline plants of fuel r for region/country c in year t, respectively; $Scenario_{rct,woCCS}$ is the predicted level of electricity production of fuel r without CCS for region/country c in year t in AMPERE climate scenarios.

We define stranding in terms of generation since it captures the overall impact that climate constraints could have on the amount of electricity generation that will have to be cancelled.

Step 3: Re-estimating stranded generation while taking potential plants conversion into account

In the above calculations, we have assumed that operating and in-the-pipeline power plants would not be converted to cleaner options as a way to reduce stranded generation. We relax this assumption below and re-estimate stranded generation in all technologies deployed" scenarios under the assumption that some plants could be converted (the results showed in Figure 4.3). We consider three conversion options: 1) coal to gas; 2) adoption of CCS in existing and in-the-pipeline plants; and 3) coal to biomass.

Coal to gas conversions. Firstly, we consider that conversions are only viable if there is a need for gas-fired power generation that cannot already be covered by existing and in-the-pipeline power plants. This is the case if the electricity generation from gas-fired power plants in the IAM is superior to the electricity that can be produced by existing and in-the-pipeline plants. In this case, conversions can be used to produce the remaining gas-fired power generation. The difference between

the total electricity generated in the IAM and the electricity than can already be provided by existed and in-the-pipeline plants constitutes an upper bound for the conversion potential. This is given by:

$$ExtraGas_{ct} = \max(Scenario_{gas,ct,woCCS} - Generation_{gas,ct}, 0) \tag{4.6}$$

With $ExtraGas_{ct}$ in region/country c and year t is the additional power generation from gas that cannot be already satisfied with existing and in-the-pipeline gas-fired power plants. $Scenario_{gas,ct,woCCS}$ is the electricity generation from gas-fired power plants without CCS, and $Generation_{gas,ct}$ is the electricity generated from existing and in-the-pipeline gas-fired power plants.

Secondly, we consider that the technical feasibility of plant conversions might limit the quantity of electricity generated by converted plants from coal to gas. We consider that coal-to-gas conversions can only take place in countries that also have operating gas-fired power plants since the converted plants must be able to have access to gas infrastructure and supplies. In these countries, we consider that only a fraction of coal-fired power plants would be converted. We calibrate this share based on evidence from EIA (2020) that 5 percent of coal-fired power capacity have already been converted to gas in the US. We provide results in which we assume that either up to 5 percent or 20 percent of coal-fired power capacity would be converted to gas (only in countries where there already are gas-fired power plants).

The electricity generated from converted plants in each year is then assumed to respect both boundaries: the one set by the IAM on gas demand, and the one that we set on technical feasibility. This is given by:

$$\begin{aligned} Generation_{gas,ct} = \min(&Generation_{ct,coal-to-gassuitable} * Percentage, \\ &ExtraGas_{ct}) \end{aligned} \tag{4.7}$$

$Generation_{gas,ct}$ is the amount of electricity generated from coal-to-gas converted

plants. *Percentage* is the conversion percentage of coal-to-gas suitable units. It is equal to 5 or 20 percent in our calculations.

Based on Equation 4.7, we re-estimate the amount of stranded generation for coal plants by subtracting the generation that comes from converted coal-fired power plants:

$$SA_{coal,ct} = \max(Generation_{coal,ct} - Conversion_{gas,ct} - Scenario_{coal,ct,woCCS}, 0) \tag{4.8}$$

$SA_{coal,ct}$ are the stranded generation from coal-fired power plants. $Generation_{coal,ct}$ is the possible generation from coal-fired power plants, and $Scenario_{coal,ct,woCCS}$ is the electricity generated in the IAM from coal-fired power plants without CCS.

Adoption of CCS in existing plants and those in the pipeline.The AMPERE scenarios distinguish between the electricity produced by plants without and with CCS. In Figure 4.2, we assumed that none of the electricity produced with CCS was coming from operating and in-the-pipeline plants. We relax this assumption.

The total energy generated from fossil fuels with CCS in the IAM scenarios sets an upper bound for CCS conversions $Scenario_{rct,wCCS}$. Furthermore, we also look at the potential for conversions among existing plants and those in the pipeline. We use the criteria of Caldecott et al. (2016b): units are deemed CCS-suitable if they have a capacity above 100 MW, are less than 20 years old, emit less than 1000 g CO_2/kWh and are located within 40 km of geological areas suitable for CCS. Using these criteria, we find that around 24 percent of operating and in-the-pipeline fossil fuel power units would be suitable for CCS conversion. We then assume that either half or all CCS-suitable plants could be converted to CCS.

The total share of energy produced from existing or in-the-pipeline coal-fired power plants equipped with CCS is then set to be either equal to the maximum limit set by the IAM on the energy that could be generated from CCS; or equal to the total potential from existing or in-the-pipeline plants, which we assume to be

either the 24 percent of CCS-suitable plants, or half of these.

The corresponding equations for fuel r, country c and year t are:

$$Conversion_{rct,CCS} = \min(Generation_{rct,CCSsuitable} * CCS_{share}, \\ Scenario_{rct,wCCS}) \tag{4.9}$$

$Conversion_{rct,CCS}$ is the amount of fossil fuel generation from plants converted to CCS from fuel r, in region/country c in year t. $Generation_{rct,CCSsuitable}$ is the electricity generated from fossil fuel power plants that are CCS-suitable. CCS_{share} is the share of these plants that are converted to CCS. It is assumed to be equal to 50 percent or 100 percent in our calculations. Then, the amount of stranded generation with CCS conversions is equal to:

$$SA_{rct} = \max(Generation_{rct} - Scenario_{rct,woCCS} \\ - Conversion_{rct,CCS}, 0) \tag{4.10}$$

Where SA_{rct} is the estimated amount of stranded generation from fuel r, in region/country c in year t. $Generation_{rct}$ is the electricity that can be generated by all existing and in-the-pipeline power plants (for fuel r). $Scenario_{rct,woCCS}$ is the electricity generated from fossil fuels without CCS in the IAM scenario.

Coal-to-biomass conversions. The co-firing of biomass in coal power plants could reduce the amount of stranded generation. It is likely to happen in coal-fired power plants that have CCS installed. We recalculate the total amount of CCS conversions while allowing for biomass to be co-fired in coal-fired CCS-suitable plants. We therefore modify Equation 4.9 to include biomass as a potential fuel for CCS-suitable plants. The change does not affect CCS conversions for gas- and oil-fired power plants but it has an impact on CCS conversions for coal-fired power plants. With co-firing, the electricity generated from coal-fired CCS-converted plants has to be inferior to the electricity generated from the sum of coal and biomass with

CCS in the IAMs:

$$Conversion_{coal,ct,CCS} \leq Scenario_{coal,ct,wCCS} + Scenario_{biomass,ct,wCCS} \quad (4.11)$$

Furthermore, the co-firing ratio between coal and biomass has to be respected:

$$Conversion_{coal,ct,CCS} \leq Scenario_{coal,ct,wCCS}/(1 - ratio) \quad (4.12)$$

ratio is the share of biomass used to co-fire with coal. We set this ratio to be either 20 or 50 percent. This is because IEA and IRENA (2013) estimates that a 20 percent co-firing ratio is feasible in most cases, while a 50 percent co-firing ratio is technically achievable.

The two constraints above ensure that the amount of biomass co-fired in coal-fired power plants retrofitted with CCS remain below $Scenario_{coal,ct,wCCS}$ and below the maximum amount possible for a given co-firing ratio, i.e.:$Scenario_{coal,ct,wCCS} * ratio/(1 - ratio)$.

We rewrite Equation 4.9 with these new constraints for the amount of CCS conversions from coal-fired power plants:

$$\begin{aligned} Conversion_{coal,ct,CCS} = \min(&Generation_{coal,ct,CCSsuitable} * CCS_{share}, \\ &Scenario_{coal,ct,wCCS} + Scenario_{biomass,ct,wCCS}, \\ &Scenario_{coal,ct,wCCS}/(1 - ratio)) \end{aligned} \quad (4.13)$$

The difference between Equation 4.9 and Equation 4.13 corresponds to the amount of biomass co-fired in converted plants with CCS.

Furthermore, coal plants without CCS can also cofire biomass. The maximum amount of biomass co-fired in plants without CCS is equal to the electricity generation from coal without CCS times the co-firing ratio. However, because some plants have been converted to CCS, we need to subtract $Conversion_{coal,ct,CCS}$ from

$Generation_{coal,ct}$:

$$MaxGen_{ct,cofired,woCCS} = (Generation_{coal,ct} - Conversion_{coal,ct,CCS} * ratio \tag{4.14}$$

$MaxGen_{ct,cofired,woCCS}$ is the maximum amount of electricity from biomass without CCS that can be generated from coal-fired power plants that have not adopted CCS, given the co-firing ratio.

In addition, coal-to-biomass conversions without CCS are only viable if there is a need for power generation from biomass without CCS that is not already covered by existing and in-the-pipeline biomass power plants. This is the case if the electricity generation from biomass power plants without CCS in the IAM is superior to the electricity than can be produced by existing and in-the-pipeline biomass power plants. This is given by Equation 4.15:

$$ExtraBio_{ct,woCCS} = \max((Generation_{coal,ct} - Conversion_{coal,ct,CCS} * ratio), 0) \tag{4.15}$$

$ExtraBio_{ct,woCCS}$ is the power generation from biomass without CCS that is not already satisfied with existing and in-the-pipeline biomass power plants.
$Scenario_{biomass,ct,woCCS}$ is the electricity generated from biomass without CCS in the IAM scenario. $Generation_{biomass,ct,woCCS}$ is the electricity that can be generated from existing and in-the-pipeline biomass power plants. Biomass-fired power plants include plants that use waste, biogas, biomass and bio-oil as fuel. We apply the utilisation rates in Stated Policies Scenarios from IEA World Energy Outlook 2019 to biomass power generators, which is 55%, to calculate $Generation_{biomass,ct,woCCS}$.

The electricity generated from coal-to-biomass converted plants without CCS is either equal to $ExtraBio_{ct,woCCS}$ or to $MaxGen_{ct,cofired,woCCS}$:

$$Conversion_{ct,biomass,woCCS} = \min(ExtraBio_{ct,woCCS}, \\ MaxGen_{ct,cofired,woCCS} \tag{4.16}$$

The total impact of biomass co-firing, with CCS and without CCS, is equal to $Conversion_{ct,biomasswoCCS}$, plus Equation 4.13, minus Equation 4.9. We furthermore recalculate the amount of stranded generation for coal-fired power plants: $MaxGen_{ct,cofired,woCCS}$:

$$SA_{coal,ct} = \max(Generation_{coal,ct} - Scenario_{coal,ct,woCCS} \\ - Conversion_{coal,ct,CCS} - Conversion_{ct,biomass,woCCS}, 0) \tag{4.17}$$

In Equation 4.17, we use Equation 4.13 and not Equation 4.9 to compute $Conversion_{coal,ct,CCS}$.

Step 4: Estimating the impact of technology availability

We construct pairwise scenario comparisons between a) the scenarios where the four technologies are fully available; and b) the scenarios where one single technology is not available, or its diffusion is limited. Specifically, the matched pairwise scenarios are using the same IAM (e.g. GCAM) but only have one technology assumption different (e.g. scenarios where CCS is fully available are paired with scenarios where CCS is not allowed, while other technology dimensions on nuclear energy, wind solar and bioenergy potential are the same). Then we compute the difference of stranded generation (estimated in Equation 4.2 to Equation 4.5) between matched pairwise scenarios and we define this as the impact of a particular technology on stranded generation.

The AMPERE database also includes scenarios with all four low carbon technologies available, but a higher reduction rates (by 50 percent) in energy intensity.

We compare the scenarios which assume energy intensity reduce at historical rates with these higher reduction rates scenarios to calculate the impact that faster energy intensity reduction could have on the amount of stranded generation.

4.7 Supplementary information

Separate pathways of electricity generation by fuel. Figure 4.6 displays the committed electricity generation between 2021 and 2100 by fuel. We also display the electricity generation forecast in the "all technologies deployed" scenario modelled by MESSAGE model for a climate stabilisation at 450ppm. In this example, we observe that coal and oil assets are at high risk of stranding, while gas assets are at lower risk of stranding. Please note that this is only one example for one IAM and one scenario. Results from different models and scenarios would differ significantly.

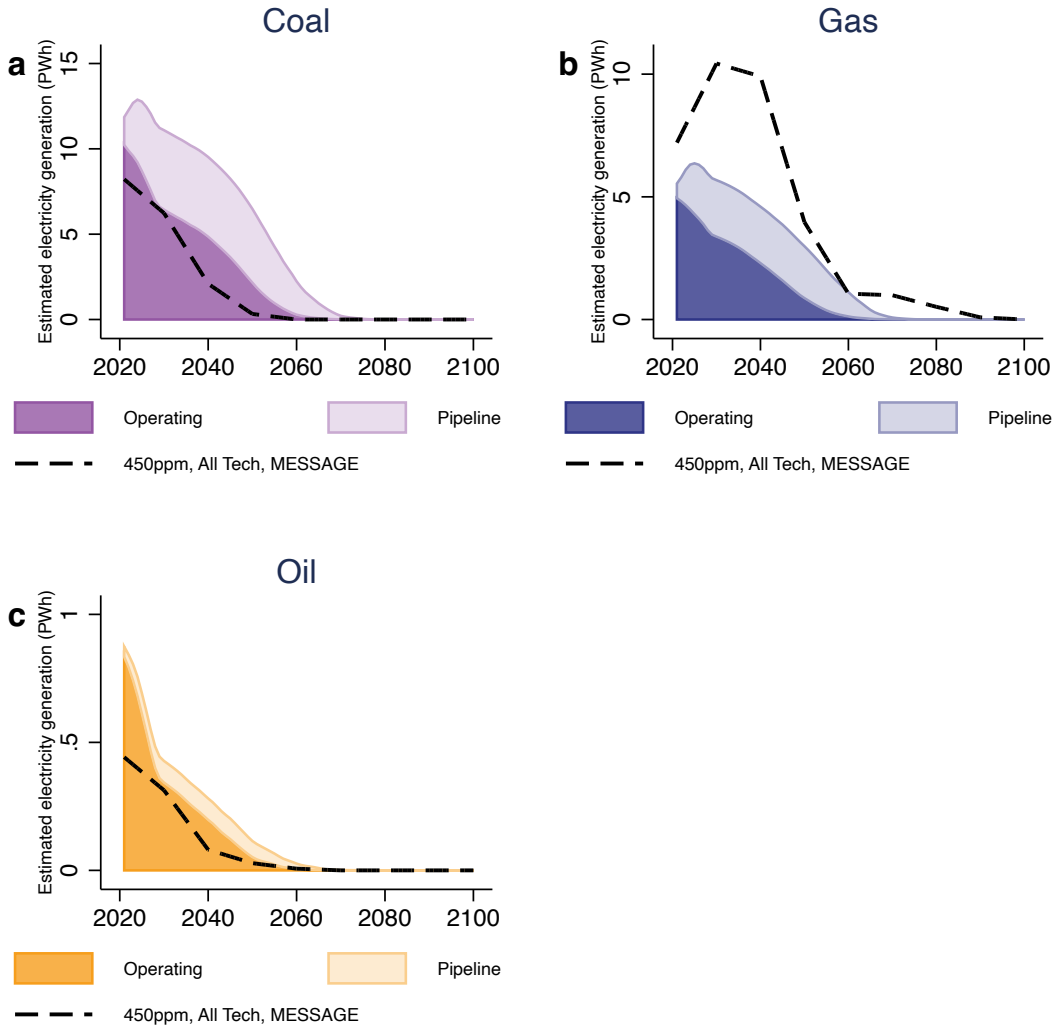


Figure 4.6: Breakdown of committed electricity generation between 2021 and 2100 for coal (a), gas (b) and oil (c) separately.

Estimates of the amount of future electricity that can be generated from currently operating and in-the-pipeline power plants by fuel. Power units are assumed to be operating at the same utilisation rates as those forecasted in the IEA Stated Policy Scenarios until the end of their expected lifetime. Darker shading indicates the expected generation from currently operating plants, while lighter shading indicates the expected generation from in-the-pipeline plants. The black dashed line is an example for the electricity that could be produced based on one scenario from one IAM. The specific example is the 450ppm, "all technologies deployed" scenario obtained from the MESSAGE model.

Global electricity generation by technology in the "all technologies deployed" scenarios. Figure 4.7 shows the global electricity generation by technology in each IAM.

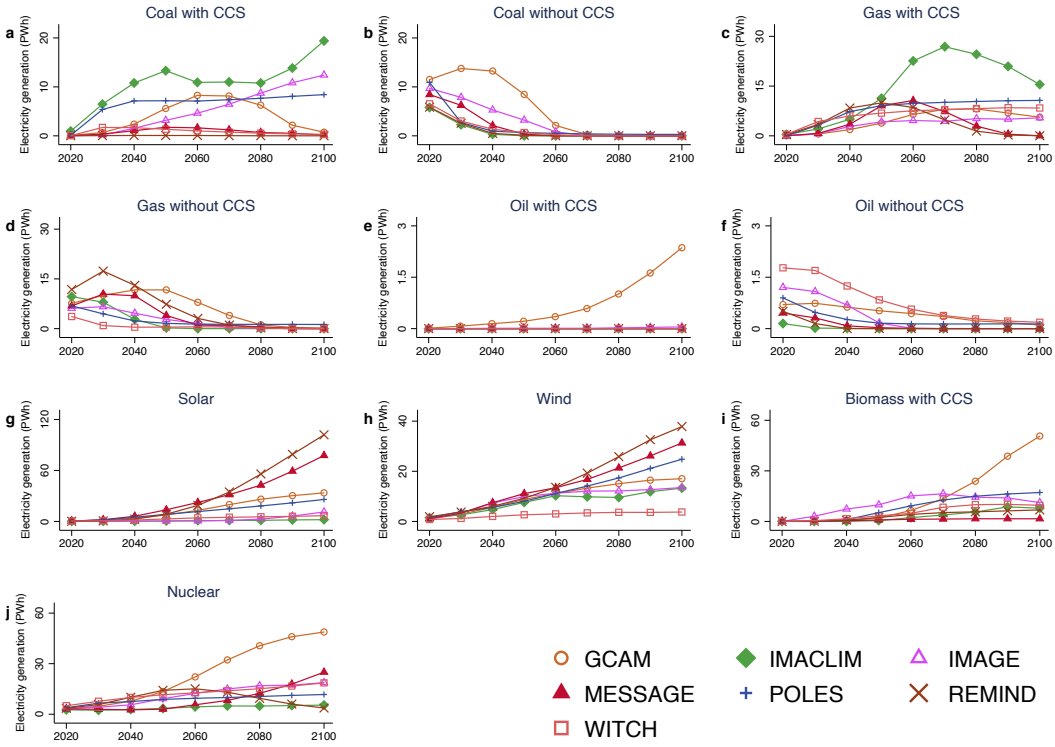


Figure 4.7: Global electricity generation from different technologies between 2021 and 2100 in AMPERE "all technology deployed" scenarios

Estimates of stranded generation by fuel. Figure 4.2 presented the amount of stranded generation for all types of fossil fuel power plants (including coal-, gas-, and oil-fired). In Figure 4.8, we show the breakdown of stranded generation by fuel, for: a) all fuels (same as Figure 4.2); b) coal; c) gas; and d) oil. Coal-fired power plants make most of stranded generation (82 percent in the across all IAMs), followed by gas-fired power plants contributing 16 percent in the across model mean. Due to their limited capacity, oil-fired plants only constitute a very small share of stranded generation.

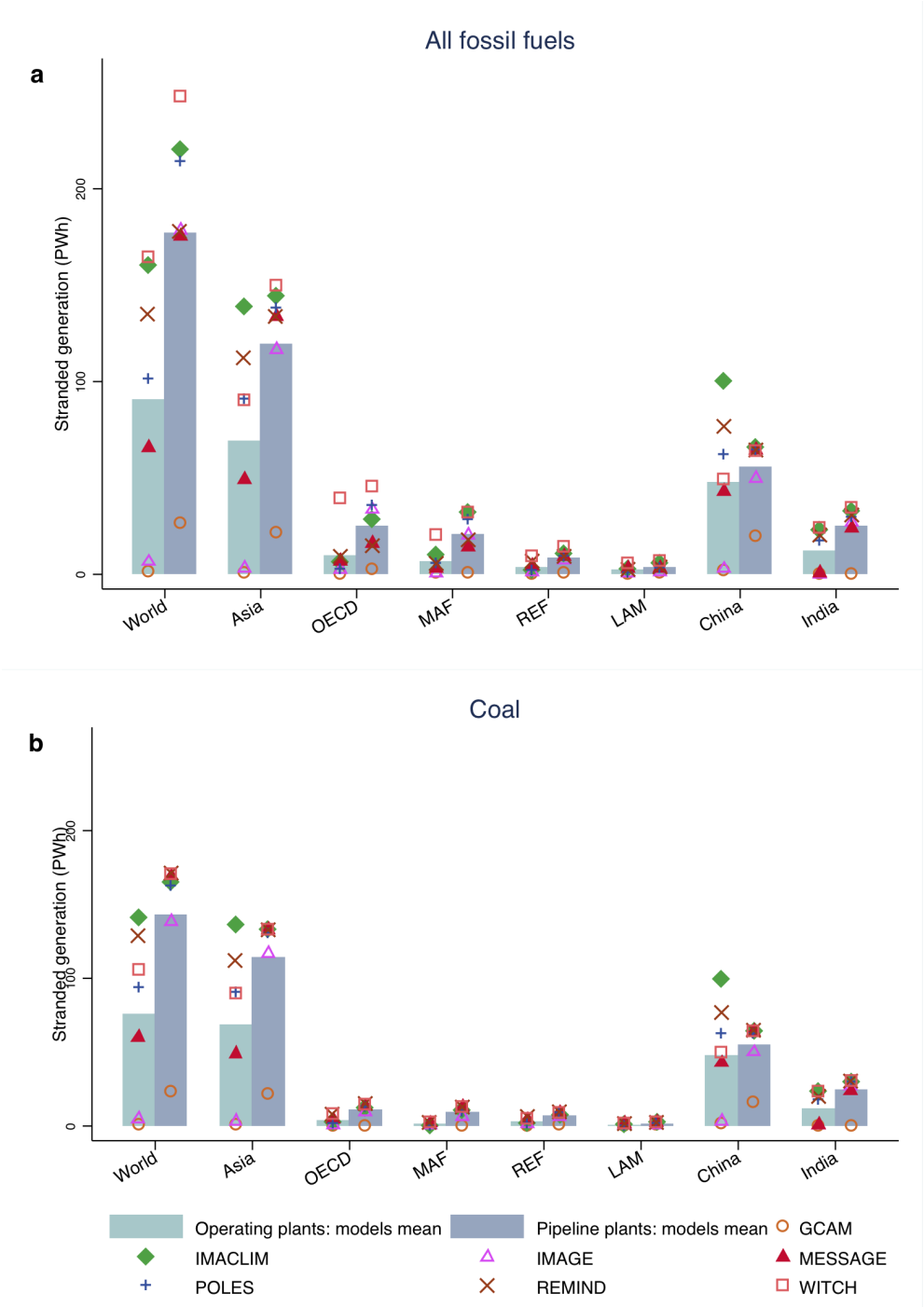


Figure 4.8: Estimated stranded generation in "all technologies deployed" scenarios (2021-2100) by type of fuel

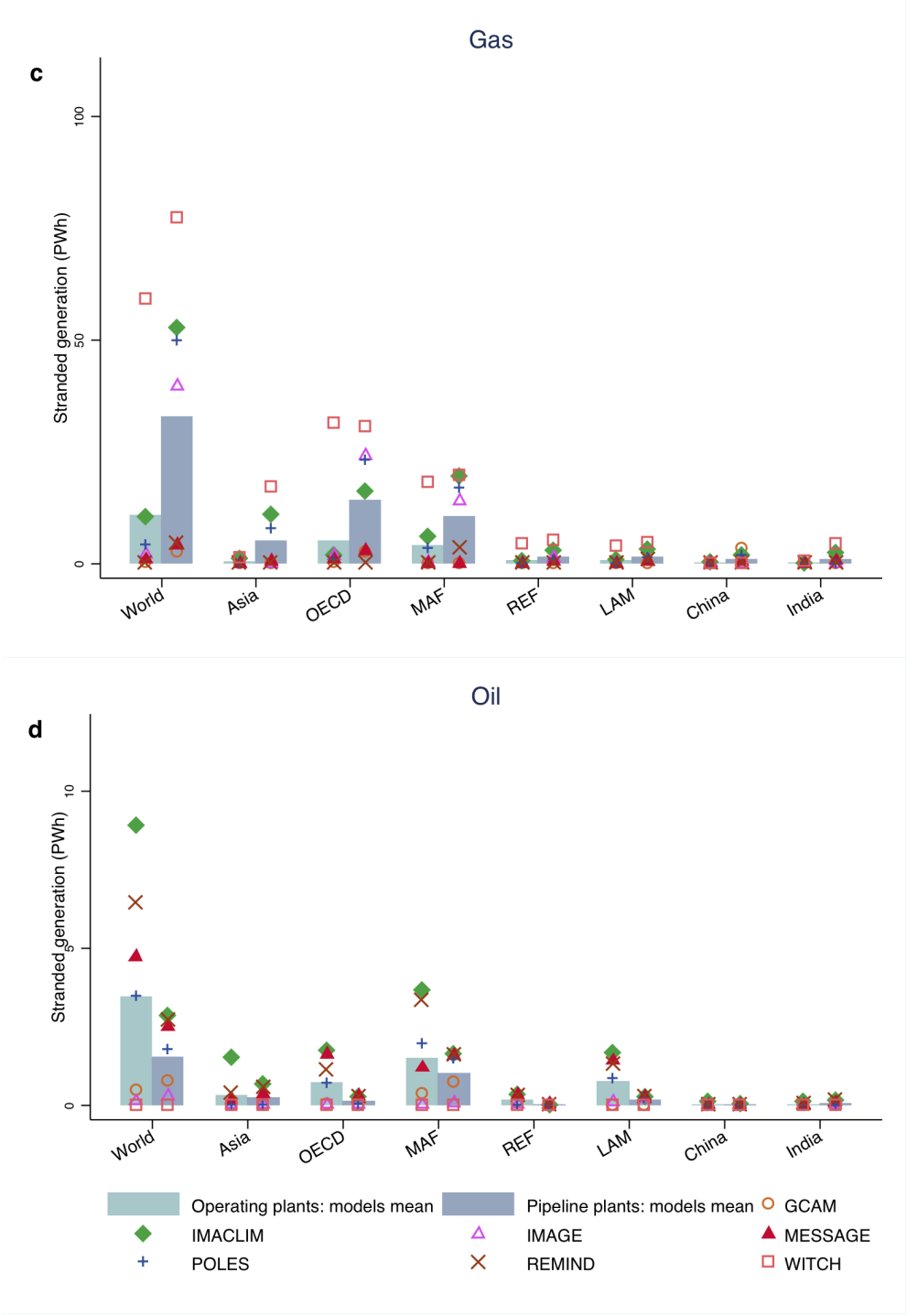


Figure 4.8 (continued)

Estimates of the amount of future electricity that can be generated from currently operating and in-the-pipeline power plants by fuel. Power units are assumed to be operating at the same utilisation rates as those forecasted in the IEA Stated Policy Scenarios until the end of their expected lifetime. Darker shading indicates the expected generation from currently operating plants, while lighter shading indicates the expected generation from in-the-pipeline plants. The black dashed line is an example for the electricity that could be produced based on one scenario from one IAM. The specific example is the 450ppm, "all technologies deployed" scenario obtained from the MESSAGE model.

Sensitivity analysis for lifetimes and utilisation rates. In Figure 4.9, we check the sensitivity of the estimated electricity generation and the amount of stranded generation to the choice of utilisation rates and lifetimes. For utilisation rates, the baseline uses the average fuel-specific utilisation rates of the Stated Policy Scenarios (2018-2040) from IEA World Energy Outlook (2019). The maximum and minimum cases rely on the maximum and minimum fuel-specific utilisation rates observed during years 2004-2018, extracted from IEA World Energy Outlook 2005-2019. For lifetime assumptions, baseline assumes that power generators will operate until the end of their estimated expected lifetime. The "reduce" and "extend" cases assume a decrease / increase in lifetime by 10 percent compared to the baseline.

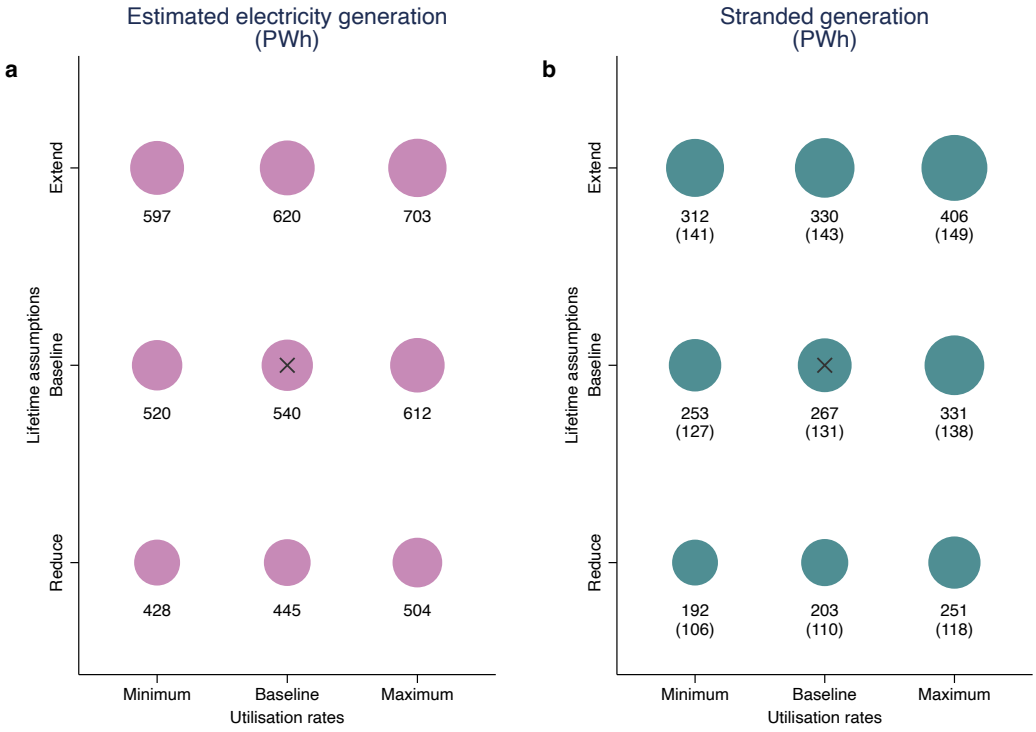


Figure 4.9: Sensitivity of estimated electricity generation (a) and stranded generation (b) to utilisation rates and asset lifetime.

All cases show the total estimated electricity generation and stranded generation of operating and in-the-pipeline fossil fuel power plants (coal-, gas- and oil-fired) from 2021 to 2100 in the "all technologies deployed" scenarios, without any conversion. The circles for panel (b) provide the mean across 7 IAMs (GCAM, IMACLIM, IMAGE, MESSAGE, POLES, REMIND, and WITCH). The standard deviation reported in parentheses accounts for the variation across models. The baseline assumption based results presented in the main results are emphasised with the "x" symbol.

Estimates of stranded generation using the IPCC SR1.5 database. Our results are obtained by using AMPERE scenarios. Other databases of IAMs could be used to calculate stranded generation with our methodology. We show the results of stranded generation using IPCC SR1.5 database in Figure 4.10 . The average amount of stranded generation in 2°C scenarios of IPCC SR1.5 database is about 300 PWh, which is 12 percent higher than the amount in AMPERE 2°C scenarios of Figure 4.2 . This is mostly because Figure 4.2 assumes that all technologies reducing the level of stranded generation are fully deployed, while 2°C scenarios of IPCC SR1.5 have varying assumptions on technology development. The amount

of stranded generation under 1.5°C scenarios is higher by about 17 percent compared to 2°C scenarios in IPCC SR1.5 databases. Besides, the order of magnitude, regional distribution and model features of stranded generation are very similar. For example, the WITCH model shows the highest amount of stranded generation, while GCAM model has the lowest amount.

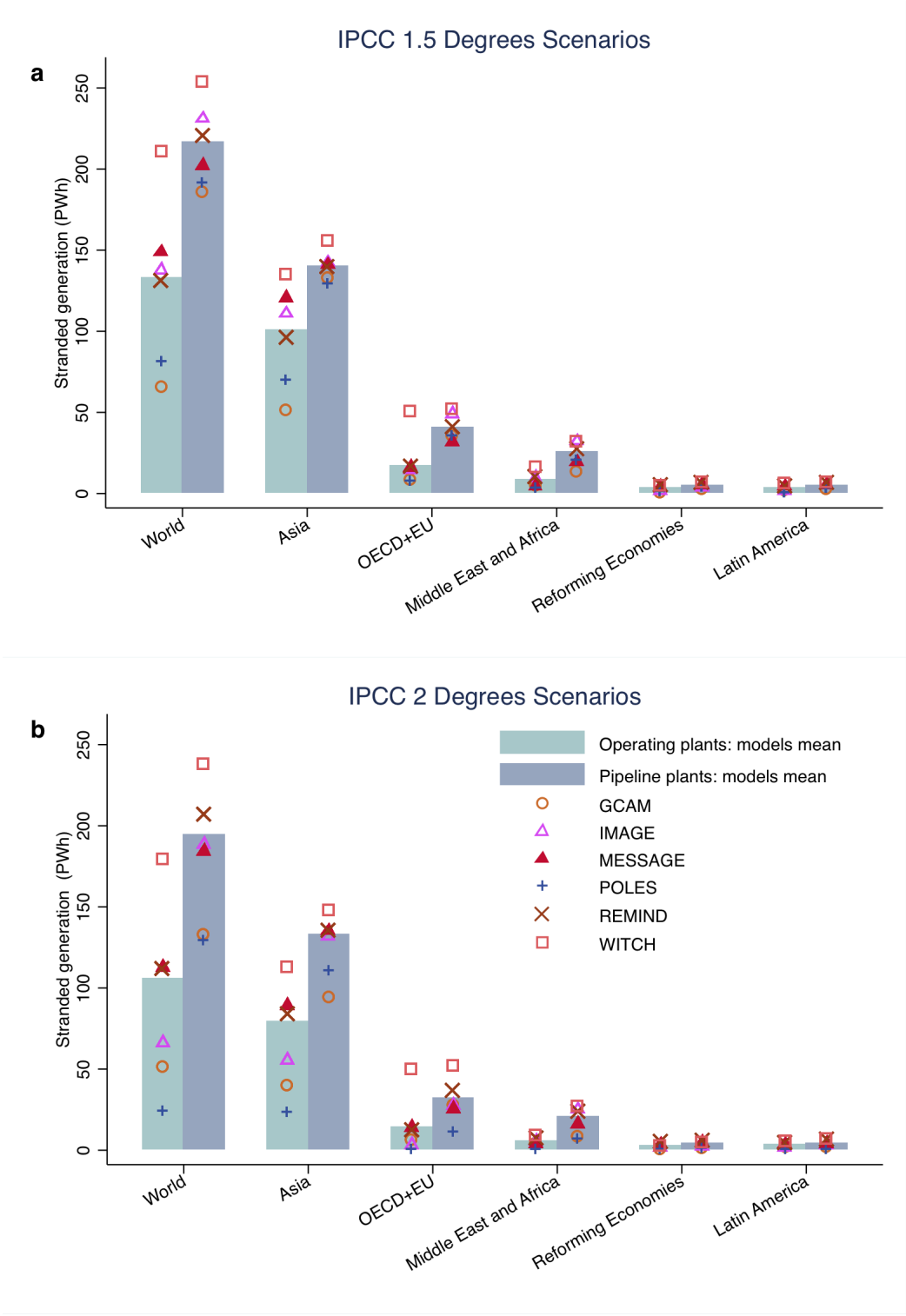


Figure 4.10: Estimated stranded generation using the IPCC SR1.5 scenarios (2021-2100)

Scatter points represent the mean value of estimations from individual IAMs and bars show the mean value across all models

Comparison of CO_2 capture. We compare the CO_2 captured in the AMPERE "all technology deployed" scenarios with the IEA's Sustainable Development Scenario (IEA, 2020a) in Figure 4.11. The AMPERE scenarios and the IEA's Sustainable Development Scenario make relatively similar assumptions for CCS take-up from all sources except biomass. However, assumptions regarding biomass with CCS are very different. The IEA's Sustainable Development Scenario assumes a late take-up of biomass with CCS, while in the "all technology deployed" scenarios of AMPERE, this technology is being deployed at scale. In Section 4.4, we discuss the possible impact of low technology diffusion on the amount of stranded generation.

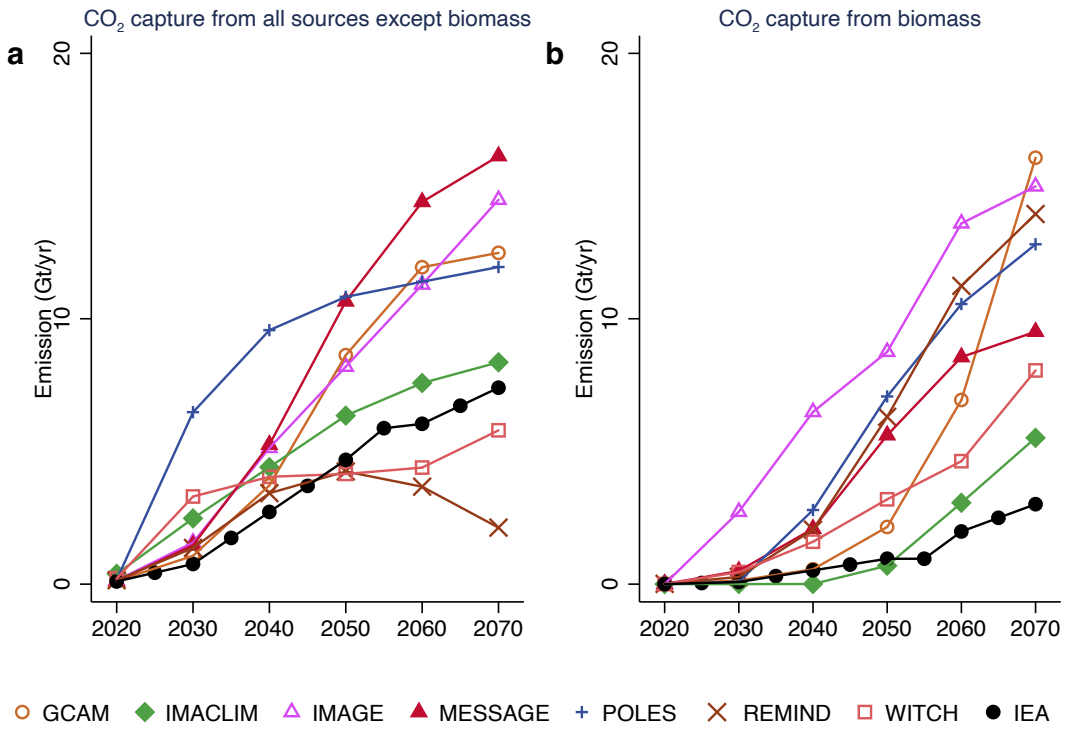


Figure 4.11: CO_2 capture from biomass and other sources in the AMPERE "all technology deployed" scenarios and the IEA Sustainable Development scenarios (2021-2070)

5 Environmental regulation and productivity are not always at odds: evidence from firms in China

Yangsiyu Lu[1], **Jacquelyn Pless**[2]

Abstract

Environmental regulations can be costly, but they can also induce innovation that improves productivity if the benefits outweigh the costs, and not all firms face the same costs. We provide the first empirical evidence of a regulation enhancing firm productivity across the industrial sector, showing how differences in compliance costs matter for understanding the impact of regulation on economic activity. We use a heterogeneous difference-in-difference research design to study not only the "dirtiest" firms facing the highest compliance costs but also those in "cleaner" (but still regulated) industries. Productivity increases by 4% for firms in cleaner industries while there is no effect on the dirtiest. This is driven partially by firm sorting. Private firms exit and sales increase for state-owned enterprises. Private firms also invest in higher-skilled labour and management yet there is no effect on new product output, suggesting that they may innovate in their processes and practices.

JEL Codes: Q52, Q58, O31, O33, D22

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5.1 Introduction

Air pollution is one of the most severe environmental threats confronting our society today. Not only does it play a pivotal role in driving climate change, but it also creates substantial costs in and of itself. It dampens worker productivity (Zivin and Neidell, 2012, Chang et al., 2016, 2019) and labour supply (Hanna and Oliva, 2015), reduces life expectancy (Ebenstein et al., 2013), and increases infant mortality (Chay and Greenstone, 2003, Arceo et al., 2016). At the same time, environmental regulations are also costly—particularly for industrial firms that often have to make large capital investments in abatement technology or adjust their production processes to comply—which has generated a long-standing "environment versus economy" discourse amongst policymakers. Indeed, empirical analyses so far have primarily found that environmental regulation hurts firm productivity (Kahn, 1997, Greenstone, 2002, Greenstone et al., 2012) and reallocates labour away from regulated industries (Walker, 2013).

However, not all firms face the same costs, and this heterogeneity has implications for developing an understanding of how regulation impacts economic activity and competitiveness. While firms typically must invest to some degree, the level of investment required and the strategies firms take to come into compliance, should they choose to do so, depend on their current pollution intensity. Firms that pollute more typically must invest more—such as by replacing old machinery—whereas those that already pollute less may only need to make minor adjustments in their production processes and practices. And in both cases, firms also have incentives innovate to reduce the cost of using pollution-intensive inputs. Such investments may even improve performance and enhance productivity if the benefits outweigh the costs.

In this paper, we study the impact of an environmental regulation on industrial firms' productivity, and importantly, we estimate the effects on firms not only in

the most pollution-intensive industries (that face the highest compliance costs) but also those that are "cleaner" (and still regulated). This allows us to capture the *net* effect, and in doing so, we provide the first empirical evidence of environmental regulation enhancing firm productivity on average across the industrial sector.[3] The findings are consistent with what is often considered the "strong" version of the Porter Hypothesis. That is, regulation creates incentives for firms to develop or adopt new technologies, processes, or practices, which can, in turn, enhance productivity (Porter and Van der Linde, 1995).

The theory behind the Porter Hypothesis is not new. However, while there is growing evidence that environmental policy and regulation spur innovative activity like R&D and patenting (i.e., the "weak" version of the Porter Hypothesis), relatively little is known about whether this translates into productivity improvements.[4] Greenstone (2002) and Greenstone et al. (2012) offer the most robust analyses of air quality regulation and industrial firm productivity to date, finding that the total factor productivity (TFP) of the most pollution-intensive manufacturing plants declined in response to the US Clean Air Act.[5] As productivity is ultimately the key input into economic growth and competitiveness, continuing to build this evidence base is of first-order importance, and it is increasingly urgent amidst productivity growth declines in developed countries and as developing countries face widespread poverty yet disproportionately bear the burden of pollution.

[3]Hafstead and Williams (2018) also point out the importance of examining the net effect and study the impact of pollution taxes on employment in a general equilibrium model, finding that employment decreased in some industries but increased in others such that the net (negative) effect is small. Our work differs by studying firm productivity and a regulation (rather than labour and taxes), by taking a reduced form approach, and in our findings that there are positive effects even on some directly regulated firms.

[4]See Jaffe and Palmer (1997), Newell et al. (1999), Popp, D. (2002), Aghion et al. (2016), Calel and Dechezleprêtre (2016), and Calel (2020). In addition to not examining productivity, most of this work also studies market-based interventions like carbon taxes as opposed to command-and-control regulation.

[5]He et al. (2020) more recently also found that regulation reduced firm productivity in China but in the water context, and Bailey (2019) finds that carbon pricing reduces productivity of coal power plants. See Dechezleprêtre and Sato (2017) and Cohen et al. (2018) for the latest reviews of the literature. Most work so far on productivity studies it at the industry or country level. For example, Berman and Bui (2001) find that firm productivity increased in response to a regulation for refineries in Los Angeles.

To help narrow this knowledge gap, we specifically study China's Two Control Zone (TCZ) regulation, which was implemented in 1998 and set objectives for reducing sulphur dioxide (SO_2) emissions in about half of China's prefectures. We use a heterogeneous difference-in-difference research design, exploiting two main sources of variation that determine treatment status—whether a firm is located in a regulated prefecture and before/after variation based on the regulation's implementation timing—and we allow the effect to vary based on whether the firm is in one of the dirtiest or "cleaner" industries.

We find that total factor productivity (TFP) increases by 4.3% for firms in less pollution-intensive industries while there is no effect on those in more pollution-intensive industries. The net average effect is thus positive. When we limit the sample to only industries that are "responsive" (i.e., those whose productivity increases), the estimates increase to 16% and 8%, respectively. These are the first results providing evidence that environmental regulation enhances firm productivity across an entire sector to the best of our knowledge. We would have drawn different conclusions that are more in line with the existing literature if we did not also consider the effects on "cleaner" industries, highlighting the importance of considering all regulated firms to understand the implications of environmental regulation for industrial activity and growth.

A natural question that emerges from these results is whether the regulation was enforced and that firms actually took action to reduce their pollution levels. If not, such productivity improvements would not be associated with the regulation itself. We examine this by estimating the effect of the regulation on SO_2 concentration—the pollutant targeted by the TCZ regulation—using satellite-based data at the prefecture-month-year level and find that pollution did indeed decrease by about 5%. This suggests that firms did make adjustments in response to the regulation.

We conduct a series of additional tests to explore the underlying mechanisms driving our results, and taken together, they suggest that firm sorting and innovative

activity are likely at play. There are no increases or reallocation of inputs, but sales and value-added increase, leading to higher productivity of labour, capital, and intermediate inputs. Productivity gains are thus associated with an output effect, and firm exit plays a role. The propensity to exit increases by about 5% in response to the regulation but only for firms that were least productive in the pre-regulation period, resulting in a higher average productivity. This type of reallocation across firms is reasonable and brings efficiency improvements, as resource misallocation is common in the developing country context.

Lastly, we explore whether firms innovate in response to the regulation as the Porter hypothesis would suggest. We do not examine the effects on traditional innovation measures like patenting, since industrial firms are unlikely to develop new technologies unless an environmental technology is already their core business. They are more likely to innovate in their processes and practices, and these innovations are less likely to be patented (Hall et al., 2013). Rather, we investigate whether firms start investing in high-skilled workers—such as scientists and engineers—and whether they allocate more resources towards management, which is critical for successfully integrating new knowledge into processes and practices for it to affect productivity. We find that both average wages and expenditures on management do indeed increase. Furthermore, there is no change in output specifically associated with new products, suggesting that firms may be innovating in their processes and practices if average wages and management expenditures are indeed good proxies for investment in higher-skilled labour.

One caveat to this conclusion is that we cannot fully rule technology adoption as a mechanism. Although we find no effect on capital, firms could displace old machinery with more efficient and less pollution-intensive machinery, leading to a net-zero effect. That said, since technology adoption typically does not require ongoing high-skilled labour (just workers to operate the equipment) or additional ongoing management expenditures, we find process and practice innovation to be a

likely driver of productivity improvements.

To probe the underlying mechanisms further, we also study private firms and state-owned enterprises (SOEs) separately given how incentive structures vary by firm ownership. Environmental regulation evasion is common, and China has a long history of government favouritism of the state sector. Local public officials—who are responsible for enforcing the regulation—typically appoint the executives of SOEs and benefit from the relationships (Barwick et al., 2018). This creates strong incentives for the government to help SOEs survive, reducing the pressure on public sector firms to innovate.

Our results are consistent with favouritism behaviour like this. It turns out that the effects on exit and innovation are almost entirely driven by private firms, despite productivity increasing for both SOEs and private firms. There is no exit of SOEs on average whereas the propensity to exit increases by 5% for private firms, an effect that's driven by older firms only. This aligns with our previous finding that less productive firms are those that exit, as older firms are more likely to rely on older and less efficient machinery. On the other hand, the propensity to exit increases by 15% for both SOEs and private firms that are small, although the number of firms that this equates to is much smaller for SOEs. This is in line with China's "grasping the large, letting go of the small" approach to industrial reformation, but there is still a small effect on larger firm exit, suggesting that China particularly favoured larger SOEs when "grasping". Finally, we find positive effects on average wages and management expenditures only for private firms. Considering these heterogeneous effects across firm ownership together points to the regulation being enforced unequally such that SOEs benefit from protectionism, and this comes with potential threats to economic growth and competitiveness given how small private firms are often key drivers of innovation.

Our paper makes several contributions. First and foremost, by studying how environmental regulation impacts not only the most pollution-intensive industries

but also those that are "cleaner," we are able to shed new light on how environmental regulation and growth are not always at odds. The two papers closest to ours are Greenstone (2002) and Greenstone et al. (2012), who found that the U.S. Clean Air Act dampened the TFP of manufacturing plants. Our alternative approach to studying how regulation impacts productivity and the conclusions that it leads to complement this work.

The second main contribution of this paper is that we look under the hood of the firm to explore the strategies firms take to comply with environmental regulation and whether these translate into improved productivity. Our exploration of the channels through which firms achieve such outcomes suggests that process and practice innovation at least part of what drives productivity improvements. Even in non-environmental contexts, it is rare to empirically connect innovation to productivity.

Our empirical setting is also important in its own right, as China is the world's heaviest polluter and a significant contributor to global economic activity. The literature focusing on how environmental regulation and quality impacts other firm outcomes in China is growing but less attention has been paid to productivity.[6] One exception is He et al. (2020) who study a water regulation and find that it reduced productivity. Another closely related paper is Fan et al. (2019)'s examination of how regulation stringency impacts firm performance, also finding that it declined despite the adoption of new practices.[7]

Lastly, this paper is important and timely for policy in various settings. Our findings challenge the narrative that often dominates political debates, suggesting that intervention aiming to reduce pollution actually may be an effective tool for simultaneously fostering economic development and environmental quality improvements.

[6]For example, some have studied how pollution impacts labour demand (Liu et al., 2017, Gray et al., 2014, Liu et al., 2018, 2021) and supply (Fan and Grainger, 2021).

[7]Tanaka et al. (2014), an unpublished working paper, also study how the TCZ regulation impacted firm productivity but they do not have before/after variation.

This paper proceeds as follows. We firstly discuss the institutional details of our empirical setting and our research design. Then we describe our data and examine the effect of the regulation on pollution to demonstrate that it "worked" before moving on to our main analysis of productivity. We provide the main results and explore the underlying mechanisms. Finally, we explore differences across firm ownership and conclude.

5.2 Background and research design

5.2.1 Air pollution and environmental regulation in China

China's rapid economic growth has come with significant increases in air pollution. In particular, sulphur dioxide (SO_2) emissions from the industrial sector were a major contributor to China's ambient air pollution through the 1980s and 1990s, which reached 23.7 million by 1995 and created severe acid precipitation in more than 30% of the country's territory (Hao et al., 2001). According to the 8th Five-Year Plan (1991 to 1995) statistics, SO_2 pollution level exceeded the Class II of Chinese National Ambient Air Quality Standards (CNAAQS) for SO_2 in 149 out of 280 surveyed prefectures.[8] High levels of SO_2 and soot are severely detrimental to human health, with economic losses estimated to be about 95 billion yuan (real value) in the year 1995 (Johnson et al., 1997).

This reality and increasing public concern led the Chinese government to introduce a number of environmental regulations, eventually resulting in some of the most comprehensive environmental regulation in the developing world to date. The first was the Air Pollution Prevention and Control Law (APPCL) in 1987 (He et al., 2002). However, it provided only a general provision related SO_2 emissions and excluded the power sector, and consequently, had very little impact on reducing SO_2

[8] According to Chinese National Ambient Air Quality Standards, annual average SO_2 concentration level below 20 $\mu g/m^3$ is classified as Class I standard; Class II standard ranges from 20 $\mu g/m^3$ to 60 $\mu g/m^3$; Class III standard is between to 60 $\mu g/m^3$ and 100 $\mu g/m^3$.

emissions or acid rain. The government amended the law in 1995 with a new article imposing more stringent regulations on specific regions assigned as acid rain control zones and SO_2 pollution control zones, which became known as the Two Control Zones (TCZ) regulation.

5.2.2 The TCZ regulation

The Two Control Zone (TCZ) regulation was enacted in 1998 and aimed to limit China's total SO_2 emissions to be within 2000 levels by the year 2010, achieving urban ambient air sulphur dioxide concentrations that would meet national environmental quality standards. Another goal was to significantly reduce precipitation pH levels relative to 2000 levels. The national government designated prefectures as being SO_2 pollution control zones (i.e., regulated) based on whether the prefecture's average annual ambient SO_2 concentrations exceeded the national Class II standard, whether the prefecture's daily average concentrations exceeded the National Class III standard, and whether "high" SO_2 emissions were recorded. Prefectures were designated as acid rain control zones based on whether their average annual pH values for precipitation were less than or equal to 4.5, sulphate deposition was greater than the critical load, and high emissions were recorded.

These rules resulted in 175 regulated prefectures, spanning regions that accounted for 11.4% of the nation's territory, 40.6% of the population, 62.4% of GDP, and 58.9% of SO_2 emissions according to 1995 figures (Hao et al., 2001). Figure 5.1 illustrates their geographic distribution.

Although some aspects of the regulation were vague, it did lay out specific requirements for some industries, imposing relatively stringent pollution control measures according to nationally-mandated thresholds compared to previous efforts. The regulation particularly targeted industries related to the life cycle of coal, namely coal mining, processing, and combustion, given their particularly high contributions to SO_2 pollution in China. China consumed 963 metric tons carbon

equivalent of coal in 1998, accounting for about 30% of the world's coal consumption that year (IEA, 2020c). The main coal users, like coal power plants, industrial boilers, and kilns, contributed to approximately 35%, 34% and 11% of the total SO_2 emissions in TCZ regions (Hao et al., 2001).

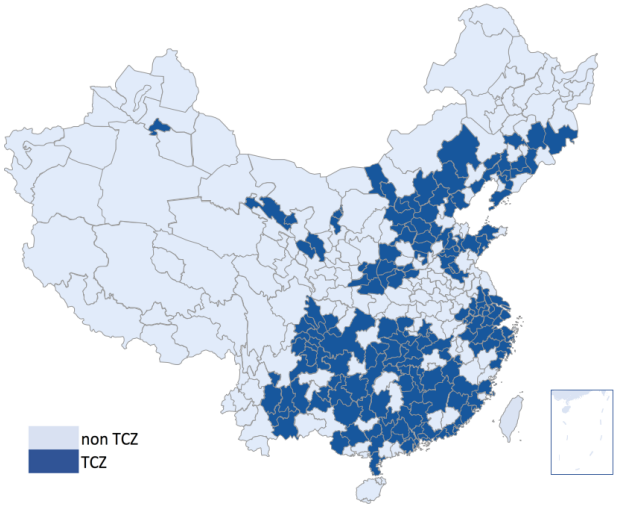


Figure 5.1: Geographic location of TCZ versus non-TCZ prefectures

The most explicit control measures were imposed on coal mining and thermal power plants. No new coal mines with sulphur content higher than 3% or no new coal-burning power plants in large and medium-sized prefectures (that were also in TCZ regions) could be built. All new and existing coal mines with sulphur content higher than 1.5% had to be equipped with coal washing facilities. Existing mines producing coal with sulphur content higher than 3% were to be gradually shut down or have output restricted. All new and existing power plants using coal with sulphur content higher than 1% had to be equipped with desulphurization facilities; existing plants were required to take action to reduce SO_2 emissions before 2000 and establish desulphurization facilities by 2010 (Hao et al., 2001).

Other polluting industries were also regulated but they were provided more

flexibility in how emissions reductions could be achieved. Firms in chemical, metallurgical, nonferrous metal (including concrete), and building materials industries in TCZs, for example, had to either construct waste gas treatment facilities (e.g., scrubbers) or "take other emissions reduction measures," such as retrofitting industrial boilers and kilns or switching to low-sulphur or washed coal. The TCZ policy also generally promoted clean production and technical renovation in all manufacturing processes to effectively reduce SO_2 emissions.

5.2.3 Research design

Our research design leverages two sources of variation created by the TCZ regulation: geographic variation based on whether a prefecture is designated as being regulated and timing variation based on before/after based on the regulation's implementation year. We examine whether productivity changes after implementation are different for firms in TCZ prefectures relative to those that are not regulated. In addition, we allow the effects to vary based on whether the firm is in a more ("dirtier") or less pollution-intensive ("cleaner") industry as measured by the proportion of total SO_2 emissions generated by that industry.[9] Although firms in "cleaner" industries face lower compliance costs, as they usually have to take less extreme measures to meet the requirements, these are polluting industrial firms, so all firms in TCZ prefectures are defined as "regulated".

This heterogeneous difference-in-differences approach allows us to identify the effect of the regulation on firms in both more and less pollution-intensive industries. We estimate the following model throughout our main analyses:

$$\begin{aligned} log(Y_{it}) =& \beta_1(TCZ_p * Post_t) + \beta_2(TCZ_p * Post_t * Polluter_s) \\ &+ \mu_p * t + \alpha_i + \gamma_{st} + \delta_{sp} + \epsilon_{it} \end{aligned} \tag{5.1}$$

[9] Our method for making these assignments are detailed in the Supplementary Information (Section 5.8).

where Y_{it} is firm i's (log) total factor productivity (TFP) (or other outcomes) in year t, TCZ_p is a "regulated" indicator equal to one for firms located in TCZ prefectures (p) and zero otherwise, and $Post_t$ is an indicator equal to one in the post-policy years (from 1999 onwards) and zero otherwise.[10] The variable $Polluter_s$ is an indicator equal to one for firms in more pollution-intensive industries (s) and zero otherwise.

The main coefficients of interest are β_1 and β_2. β_1 captures the regulation's effect on firms in less pollution-intensive industries and β_2 reflects the "extra" impact on the dirtiest firms relative to those that are cleaner. The total effect on the dirtiest firms is the sum of the two coefficients.

The two identifying assumptions of our research design are that: 1) trends in productivity are parallel for regulated and unregulated firms absent the regulation, and 2) there are no spillover effects on firms in non-TCZ prefectures (i.e., the stable unit treatment values assumption (SUTVA) holds). One potential threat to identification is that treated and untreated prefectures may be affected by macroeconomic shocks differently over time. For example, since treatment was not randomly assigned—it was determined by historical pollution levels—the pace of industrialisation and development may systematically differ.

To account for this, we include prefecture-specific linear time trends, $\mu_p * t$. The most flexible approach would be to use prefecture-year fixed effects instead, but doing so does not allow us to estimate the effects on less pollution-intensive (but still regulated) firms. It would be equivalent to estimating a triple-difference model and identifying the effects only on the dirtiest firms. In Section 5.4.1, we provide evidence that time trends sufficiently control for these concerns, allowing us to address this potential bias while providing a more complete picture of how the regulation affects industrial activity.

We include a rich set of additional fixed effects. Firm-level fixed effects (α_i)

[10] We treat the year 1998 as "pre-policy" since there is a delay between the policy's announcement at the central government level and implementation at the local government level.

control for time-invariant mean differences in outcomes across firms, so our estimates can be interpreted as within-firm effects. Industry-year fixed effects control for how industries may be affected differently by shocks to economic activity (γ_{st}) and industry-prefecture fixed effects control for how industries may be affected differently across prefectures (δ_{sp}). Standard errors are clustered at the prefecture level in our baseline specification, which is the conservative approach relative to clustering at the industry or firm level (see Section 5.4.2).

5.3 Data, productivity measurement, and regulation enforcement

In this section, we provide an overview of our data sources, how we address challenges with the data, and our production function estimation strategy. We also provide evidence that the regulation was enforced before moving on to our main results in the following section.

5.3.1 Data overview

To study the TCZ regulation, we match several data sets that provide firm-level, prefecture-level, and industry-level information for China's industrial sector. We start by gathering data on firms for the period 1996 to 2006 from the China Industrial Enterprise Database (CIED), which is maintained through annual surveys conducted by the National Bureau of Statistics. This database includes detailed accounting information that provides us with the variables needed for calculating TFP (value-added, labour, capital, and intermediate inputs), along with other key measures such as gross industrial output, sales, and more.

The CIED data include all state-owned enterprises (SOEs) as well as private firms with annual sales exceeding 5 million Chinese yuan. It covers 40 two-digit Chinese Industrial Classification (CIC) industries, including mining, manufacturing,

and public utilities. Although it does not contain the smallest firms in the economy, the aggregate industrial output and employment included represents about 90% and 70%, respectively, of the whole industrial sector according to 2004 figures (Brandt et al., 2012). We keep only firms that appear at least once before and once after 1999, which we mark as the first year that the TCZ policy is in effect. This allows us to include firm-level fixed effects and study the policy impacts using within-firm variation.

This database has been used in a number of economics studies so far (e.g., Hsieh and Klenow, 2009, Song et al., 2011, Brandt et al., 2012, He et al., 2020).[11] We follow the preparation procedures developed by Brandt et al. (2012) that have been widely adopted, such as their approach to matching firms over time and dropping observations that violate standard accounting principles. All nominal financial values are converted to real values (1998) using input and output deflators following Yang (2015) and He et al. (2020). This entails using annual output price indexes for every 2-digit industry to construct output deflators, and for input deflators, using industry-level intermediate input in National Input-Output tables, which allows us to account for the dynamics of input prices in different industries. See the Supplementary Information (Section 5.8) for more detail.

One difference with the panel we construct relative to others in the literature is that we extend the time covered to 1996-97 when the surveys were being piloted. During these years, mostly only SOEs were included and the sample size is much smaller.[12] Many other studies using this data start from 1998, but given the timing of the TCZ regulation, it's important to have data dating back further to probe the validity of our research design. We consider 1998 a "pre-policy" year to allow for implementation and adjustment time, so we have one year of fully comprehensive

[11] It is often referred to as the Annual Survey of Industrial Firms.

[12] Although the survey was only intended to include SOEs during these years, some of these firms are actually considered private by the definition we follow described in the Supplementary Information (Section 5.8) as they may have gone from being state-owned to private through the period we study.

data and two years of pilot year data in the pre-policy period.[13] We end up with about 24,000 firms in 1996 and 1997, increasing to around 165,000 firms in 1998 and 301,000 firms by 2006.

To determine whether firms are regulated, we obtain the list of cities designated with TCZ regulatory status from Chinese government documentation (China State Council, 1998). We assume the production site is located at the recorded address, as we do not observe whether firms have multiple sites.[14] The government's designations are made at the prefecture level for acid rain control zones and at the district/county level for SO_2 pollution control zones. We assign TCZ status at the prefecture level, defining a prefecture as regulated if it contains TCZ districts or counties, as there were several changes of administrative divisions during the sample period. Moreover, districts and counties within the same prefecture are likely to be governed under the same criteria set by the local administration.

We enhance these data with three additional sources that report prefecture-level characteristics as well as SO_2 emissions and concentration levels. From the China Statistical Yearbook, we gather industry-specific SO_2 emissions intensity information, which allows us to designate firms as being in more or less pollution-intensive industries. We define firms as more or less pollution-intensive if they belong to an industry accounting for at least 1% of total SO_2 emissions, which correlates very closely with coal consumption intensity. Table 5.9 in the Supplementary Information provides a list of all industries in our data set and identifies those that we classify as pollution-intensive, which align closely with the classifications of others in the literature studying the United States (e.g., Greenstone (2002)).

Lastly, for our examination of whether the regulation was enforced, we collect SO_2 pollution data from two sources. First, we gather prefecture-year level SO_2 emissions data from the China Environmental Yearbook. However, these figures are

[13]Many concerns with using the pilot data are alleviated by our analysis being within-firm. We also provide robustness checks dropping the pilot data throughout the paper.

[14]Brandt et al. (2012) examine this and conclude that more than 95% of all observations are single-plant firms.

reported by local government officials and may be subject to manipulation (Ghanem and Zhang, 2014, Karplus et al., 2018). Therefore, we also follow Chen and Cheng (2017) to derive satellite-based SO_2 concentration levels using data from National Aeronautics and Space Administration (NASA).[15] The data are reported monthly at the 60 by 50 kilometer grid level. We match this to Chinese prefectures by re-gridding the satellite data to their geolocations using nearest-neighbour remapping. This results in a balanced prefecture-year-month panel from January 1988 through December 2008. See the Supplementary Information (Section 5.8) for more detail.

5.3.2 Production function estimation

The CIED production and financial information allows us to measure firm-level total factor productivity (TFP) following the method developed in Ackerberg et al. (2015). We estimate production functions separately for each industry using the added value as the dependent variable, capital as a state variable, and intermediate inputs as the proxy variable (all in logs of the value plus one). We include year fixed effects to control for macroeconomic shocks and firm fixed effects to account for unobserved firm-level characteristics as well as age and an indicator for whether the firm is located in a TCZ prefecture as state variables to account for how these firms might install more equipment to reduce emissions relative to those in non-TCZ prefectures. We then use these estimates to construct firm-level TFP . More detail can be found in the Supplementary Information (Section 5.8).

We primarily rely on the Ackerberg et al. (2015) approach throughout the paper because it improves upon previously-developed control function methods in two important ways. First, it uses intermediate inputs as the proxy variable as opposed to investments, which is used in Olley and Pakes (1996), to mitigate the "lumpy investments" challenge associated with firm-level data. Second, it corrects the functional

[15]More specifically, we extract the variable "SO_2 Surface Mass Concentration" from M2TMNXAER version 5.12.4, derived from the project of Modern-Era Retrospective analysis for Research and Applications version 2 (MERRA-2).

dependence problem that both Olley and Pakes (1996) and Levinsohn and Petrin (2003) face.[16] In our robustness checks, we also measure TFP following Levinsohn and Petrin (2003), which still uses intermediate inputs as the proxy variable but assumes that firms adjust immediately after experiencing a productivity shock at no cost, and we examine single-factor productivity outcomes—labour, capital, and intermediate input productivity as measured by added-value divided by each input—as more transparent measures of productivity as well.

One potential concern with our TFP measure is that it is revenue-based rather than quantity-based, so changes in productivity could be associated with firm-specific mark-ups even though we deflate revenues using industry-specific price indices. Unfortunately, data on quantities sold as opposed to revenue are not available, however we explore whether mark-ups drive our results later by limiting the sample to the most competitive and for homogeneous goods markets. We discuss this more in Section 5.4.2.

5.3.3 Pre-regulation firm and prefecture characteristics

Table 5.1 presents firm- and prefecture-level descriptive statistics in pre-policy years (1996-1998). As expected since regulated regions were targeted based upon previous pollution levels and this is strongly correlated with the degree of industrialisation, firms in TCZ prefectures are more productive, more capital-intensive, and have higher revenues, sales, and profits (Panel A). There are also statistical differences in prefecture-level characteristics: TCZ prefectures have higher GDP per capita, populations, and SO_2 emissions intensity (Panel B).

Given these differences, an important component of our identification strategy is controlling for time-invariant prefecture characteristics as well as how they may

[16] In practice, we implement single-step approach following Rovigatti and Mollisi (2018) with an "ACF correction" rather than the two-step approach of Ackerberg et al. (2015), since the ACF method can be extremely sensitive to the starting points passed to the optimisation function. See Rovigatti and Mollisi (2018) for more detail.

evolve differently over time, as discussed in Section 5.2.3.

Table 5.1: Descriptive statistics in pre-regulation period (1996-1998)

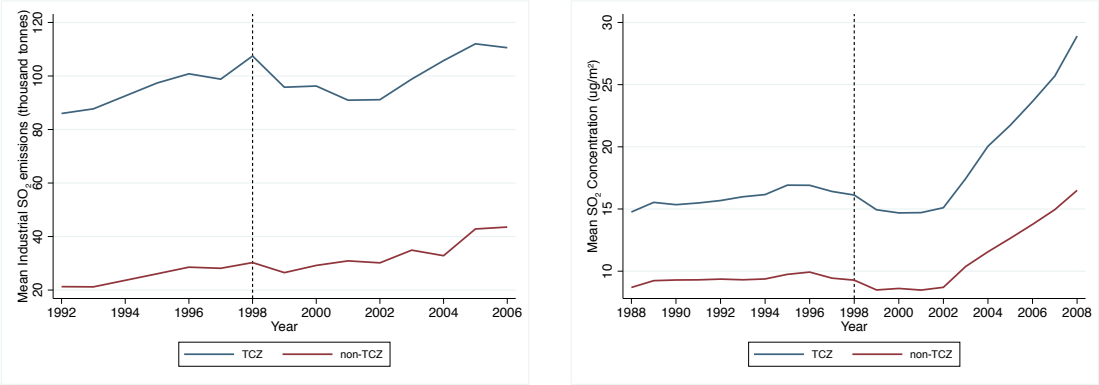
	Means			St. Deviations		Observations	
	TCZ	Non-TCZ	Difference	TCZ	Non-TCZ	TCZ	Non-TCZ
	(1)	(2)	(3)	(4)	(5)	(6)	(7)
Panel A: Firm-Level Characteristics							
TFP (log)	5.483	5.427	0.056***	1.569	1.652	122,243	38,609
Labour (log)	5.466	5.493	-0.027***	1.331	1.401	122,245	38,675
Capital (log)	9.192	9.076	0.117***	1.881	1.877	122,245	38,675
Capital-labour ratio	87.51	69.38	18.13***	308.33	225.89	122,245	38,675
Revenue (millions)	75.54	61.64	13.90***	422.95	626.97	122,245	38,675
Sales (millions)	76.24	64.24	12.00***	418.48	630.07	122,245	38,675
Profit (millions)	3.22	2.07	1.15**	52.44	126.85	122,245	38,675
Panel B: Prefecture-Level Characteristics							
GDP per capita (10,000s)	9,665	6,579	3,086**	10,252	5,130	140	81
GDP growth rate (%)	12.03	15.51	-3.48	11.47	41.17	140	81
Population (10,000s)	426.46	346.95	79.51*	320.81	244.20	140	81
SO_2 emissions (t/km^2)	60.87	31.36	29.51***	84.25	45.91	138	74
SO_2 concentration (ug/m^3)	15.94	9.36	6.58***	9.71	9.43	22,836	22,968

Notes: Table provides descriptive statistics of firm-level (Panel A) and prefecture-level (Panel B) characteristics in the pre-policy period. All monetary values are in real 1998 Yuan and variables are constructed as explained in data preparation in the Supplementary Information (Section 5.8). All prefecture-level statistics are drawn from yearly data except for the NASA SO_2 data, which are monthly. Asterisks denote $^*p < 0.10$, $^{**}p < 0.05$, $^{***}p < 0.01$.

5.3.4 Did the regulation "work"?

Before moving forward with our primary analysis of the regulation's effect on firm productivity, it is important to ensure that the policy was actually enforced and that firms thus faced compliance costs. If not, then any effects we find of the regulation on TFP are likely not actually driven by the regulation. We explore this by examining whether the regulation reduced SO_2—the targeted pollutant—expecting larger declines in TCZ prefectures relative to non-TCZ prefectures if the regulation was actually binding and changed firm behaviour.

Starting with a descriptive analysis, we plot the raw SO_2 emissions data from the China Environmental Yearbook (Panel A) and SO_2 concentration data from NASA (Panel B) over time for TCZ and non-TCZ regions in Figure 5.2. By both measures, pollution decreases by more in TCZ regions. The decline is particularly steep in Panel A, as average emissions drop substantially immediately following the regulation's implementation.[17] The NASA SO_2 concentration data in Panel B also show a slightly more significant decline for TCZ prefectures, although pollution in non-TCZ prefectures also declines.[18]



(a) SO_2 Emissions (Reported Data)

(b) SO_2 Concentration (NASA Data)

Figure 5.2: SO_2 emissions and concentration over time

This figure presents SO_2 emissions reported in the China Environmental Yearbook at the prefecture-year level (Panel A) and SO_2 concentration data from NASA reported at the prefecture-year-month level (Panel B).

To provide further confidence that the regulation has been effective to reduce pollution in regulated regions, we estimate the effect of the regulation on SO_2 concentration levels conditional on controls (using the NASA data). The NASA concentration data are less likely to be subject to manipulation (Ghanem and Zhang, 2014, Karplus et al., 2018), more comprehensive because they cover more prefectures, and have a greater time-resolution (the prefecture-year-month level). We estimate the following model:

[17] There is also a slight decrease in non-TCZ prefecture emissions for one year and they then increase again thereafter.

[18] In both sets of data, pollution increases again after 2002 for all prefectures, which is consistent with industrial activity increasing with China's accession into the WTO.

$$log(S_{ptm}) = \beta_1(TCZ_p * Post_t) + \alpha_p + \gamma_m + \delta_t + \mu_p * t + \epsilon_{pt} \quad (5.2)$$

where $\log(S_{ptm})$ is the log of SO_2 concentration levels (micrograms per square meter) in prefecture p in year t and month m. The coefficient of interest is β_1, capturing the effect of the regulation on (log) SO_2 concentration. TCZ_p is an indicator equal to one if the prefecture is regulated by the TCZ regulation and zero otherwise, and $Post_t$ is equal to one in the post-implementation period. We include month fixed effects (γ_m) to control for seasonal differences in weather and economic activity, year fixed effects (δ_t) to control for idiosyncratic shocks to economic or industrial activity in all prefectures, and prefecture-year trends ($\mu_p * t$) to control for how industrial activity may change differently over time for prefectures due to local factors.

The results are presented in Table 5.10 in the Supplementary Information. When using the full data set covering 1988 through 2008, we find that the TCZ regulation reduced SO_2 concentration levels by 3.7% (Column 1). Once limiting the sample to the time period that we study in our firm analysis (1996-2006), the effect is enhanced slightly to a 4% reduction (Column 2). We then aggregate the data to the prefecture-year level, using annual average concentration levels as the dependent variable, and find similar results. There is a 4.1% reduction when using data for 1988 through 2008 (Column 3) and a 4.6% reduction when using data for 1996 through 2006 (Column 4). These results suggest that the regulation reduced SO_2 concentration levels in TCZ prefectures relative to reductions that occurred in non-TCZ prefectures.

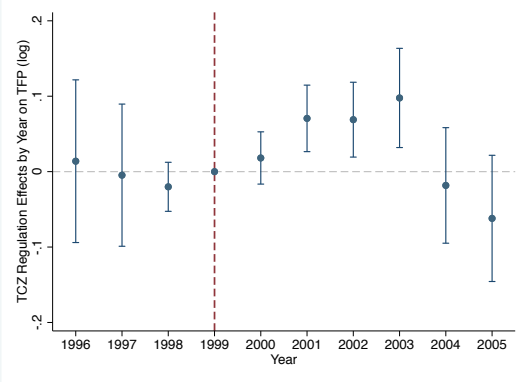
There is also evidence from previous studies examining the TCZ regulation's impact on other outcomes demonstrating that it was "effective" (Tanaka, 2015, Cai et al., 2016b), and documentation regarding firm closures and pollution treatment projects suggests that the regulation was enforced as well. About 4,492 high-

sulphur coal mines, 784 product lines in small cement and glass plants, and 404 lines in iron and steel plants were closed in TCZ zones by May 2001. Nearly 2,100 treatment projects—including boiler and kiln retrofit, waste gas treatment, flue gas desalinisation installation, and fuel-switching to low-sulphur coal—were completed in regulated areas in the first half of 2000 (He et al., 2002).

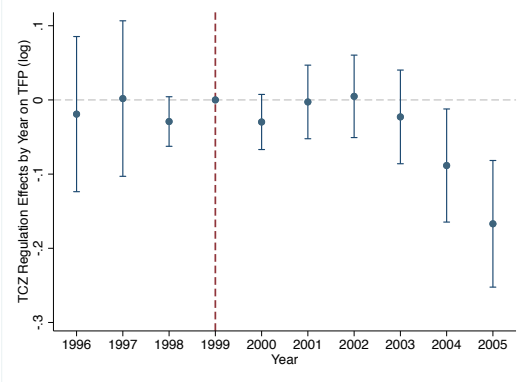
5.4 Main results

5.4.1 Effect of regulation on firm TFP

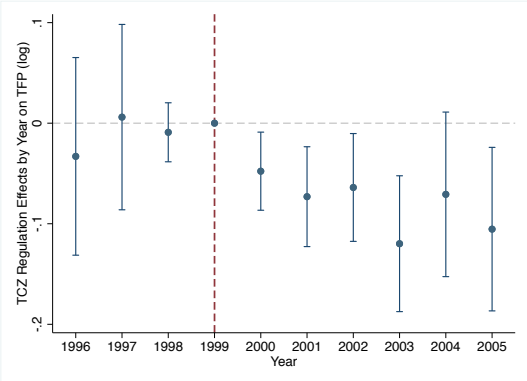
We begin by visually examining the effect of the TCZ regulation on firm (log) TFP by estimating an event study version of Equation 5.1 and plot the coefficients in Figure 5.3 (absorbing industry-prefecture fixed effects) along with their 95% confidence intervals. We allow the effects to differ for firms in less and more pollution-intensive industries following our heterogeneous difference-in-differences research design. Panel A illustrates the effects for firms in less pollution-intensive industries, which correspond to β_1 of Equation 5.1. In Panel C, we provide the effects on firms in more pollution-intensive industries *relative to the effect on firms in less pollution-intensive industries*, which corresponds to the coefficient of the interaction term (β_2) of Equation 5.1. We also estimate a separate event study for "dirty" firms only, showing the effects on firms in the most pollution-intensive industries relative to pollution-intensive firms in unregulated regions (Panel B).



(a) TCZ Regulation Effect on Firms in Less Pollution-Intensive Industries



(b) TCZ Regulation Effect on Firms in More Pollution-Intensive Industries



(c) TCZ Regulation Effect on Firms in More Pollution-Intensive Firms *Relative to Firms in Less Pollution-Intensive Industries*

Figure 5.3: Event study for effects of TCZ regulation on productivity

This figure plots the coefficients from an event study regression version of Equation 5.1 (absorbing industry-prefecture fixed effects) estimating the effect of the TCZ regulation on (log) TFP along with their 95% confidence intervals. Panel A presents the findings for less pollution-intensive firms and Panel B presents the findings for more pollution-intensive firms relative to firms in the unregulated areas. Panel C presents the effects on more pollution-intensive firms *relative to less pollution-intensive firms in regulated prefectures.*

Figure 5.3 provides three key insights. First, there are no statistical differences in TFP in pre-policy years (conditional on industry-prefecture fixed effects). The magnitudes of the effects are very close to zero and the trends appear to evolve similarly in pre-policy years. This provides confidence that our first identifying assumption—that there are no other factors generating a systematic difference in TFP trends for firms in TCZ versus non-TCZ prefectures besides the regulation—holds.[19]

Second, TFP begins to increase immediately after the TCZ regulation is implemented for firms in less pollution-intensive industries (Panel A). In Panel C, we also see that there is an immediate and sharp decrease in TFP for firms in more pollution-intensive industries *relative to those in less pollution-intensive industries*. However, there is no effect at all on firms in more pollution-intensive industries relative to dirty firms in unregulated prefectures for the first several years after the regulation is implemented (Panel B). Their productivity relative to non-TCZ firms does not decrease until 2004, which is likely due to another shift in the economy around this time, as we can see that there is a similar decline for firms in less pollution-intensive industries as well (Panel A).[20]

Third, these findings illustrate the importance of identifying the effect on both sets of firms rather than only those in more pollution-intensive industries to fully capture the regulation's impact on industrial activity. When considering them both, we can see that there is a net positive effect, as the regulation enhanced productivity for firms in less pollution-intensive industries and had zero effect on firms in more pollution-intensive industries. If we had only considered the effect on the latter and included the less pollution-intensive firms in the control group, this would produce triple-difference estimates that essentially correspond to those illustrated in Panel C. Our conclusion in that case would have been not only that the regulation reduced the productivity of firms in more pollution-intensive industries, but also that the

[19] Although the standard errors are large in the years in which data collection was in its pilot stage (1996-97), the point estimates are very similar to the 1998 estimates.

[20] For instance, this is about when all of the tariff changes related to China's accession into the WTO fully were in place, even though they entered the WTO in 2001.

overall net effect was negative. This also highlights how firms in less pollution-intensive industries in regulated areas embody a poor control group, as they would be in a triple-difference framework. Given how they are indeed affected by the regulation, including them as part of the control group would bias the results.

Turning to the regression analysis, our results from estimating Equation 5.1 on firm-level TFP are presented in Table 5.2. The estimates are consistent with the event study—there is a positive effect on firms in less pollution-intensive industries and zero effect on heavier polluters. We include only firm and year fixed effects in Column 1, and add industry by year fixed effects and prefecture-specific linear time trends in Column 2.[21] If we take Column 2 with our richest set of controls as the baseline, we find that the TCZ regulation increases the productivity of firms in less pollution-intensive industries by 4.3%. While TFP for firms in more pollution-intensive industries decreases by 4.8% *relative to those in less pollution-intensive industries*, there is no effect on them relative to firms in unregulated regions (i.e., the addition of the two coefficients). The net effect on industrial firms in TCZ prefectures is therefore positive.

Column 3 of Table 5.2 presents the results from estimating the triple-difference model[22], whereby the effect on less pollution-intensive firms is absorbed by fixed effects. These results help demonstrate the importance of taking a heterogeneous difference-in-difference approach rather than a triple-difference approach. Column 3 suggests that there is a 5.2% decrease in TFP for the heavier polluters. However, these effects capture not just the treatment in comparison to unregulated firms but also regulated ones in less pollution-intensive industries. The negative effect in Column 3 is driven by the fact that there is a significant positive effect on less

[21] While the main effects for the post-policy and regulated indicators are absorbed, the main effect for "polluter" and its two-way interactions are not (we omit their estimates from this table). They are not absorbed because industry fixed effects are at the 2-digit CIC level but we break the power industry into thermal power versus other using their 4-digit CICs.

[22] All of the variables and fixed effects in the triple-difference model are the same as in heterogeneous difference-in-difference model (Equation 5.1) except that we replace the prefecture-year trends ($\mu_p * t$) with prefecture-year fixed effects (η_{pt}) and the component capturing effects on firms in less pollution-intensive industries ($TCZ_p * Post_t$) drops out

Table 5.2: Effect of the TCZ regulation on firm TFP

Outcome Variable (log):	TFP (1)	TFP (2)	TFP (3)
TCZ * Post-Policy	0.035	0.043**	
	(0.022)	(0.022)	
TCZ * Post-Policy * Polluter	-0.037*	-0.048***	-0.052***
	(0.020)	(0.017)	(0.017)
Observations	762,957	762,957	762,922
Firm FEs	x	x	x
Year FEs	x		
Industry x Year FEs		x	x
Prefecture x Year Trends		x	
Prefecture x Year FEs			x

Notes: Table reports the effects of the TCZ regulation on (log) TFP with various sets of fixed effects. In Column 1, we include only firm and year fixed effects. We include prefecture-by-year linear trends in Column 2. In Column 3, we include prefecture-by-year fixed effects. The effect on firms in less pollution-intensive industries is in the first row and the effect on firms in more pollution-intensive industries is in the second row. The two-way interactions are included as well as the "polluter" indicator. The other two main effects are absorbed by the fixed effects. Standard errors are clustered at the prefecture level. Asterisks denote $^{*}p < 0.10$, $^{**}p < 0.05$, $^{***}p < 0.01$.

pollution-intensive firms, and these firms embody a poor control group since they are indeed affected.

Importantly, the -5.2% result of Column 3 is not statistically different from the -4.8% effect in Column 2. This provides evidence that controlling for how prefectures might respond to macroeconomic shocks differently over time with prefecture-specific trends as opposed to fixed effects is sufficient. That is, our results when using the heterogeneous difference-in-difference approach do not appear to be biased by the exclusion of prefecture-year fixed effects—they add little or no value in this case—while using prefecture-specific trends provides the benefit of allowing us to identify the effects on all firms.

Comparing Column 2 to Column 3 also illustrates why using a heterogeneous difference-in-difference approach rather than a triple-difference approach is important for understanding the implications of regulation for industrial activity. If we took the triple-difference approach, our conclusions would be reversed—the take-

away would be that there is a negative and fairly large effect—yet we see in Column 2 that the regulation's effect on these firms' productivity is actually zero. One key observation here is that, because cleaner firms in regulated regions are included in the control group in Column 3, the estimates reflect the effect on the dirtiest firms not just relative to unregulated firms but also cleaner regulated firms. This helps explain the key difference in the conclusions. Second, though, the results in Column 3 also miss the positive effects on cleaner firms, which are important for fully understanding how the regulation impacts economic activity.

5.4.2 Potential threats to baseline findings and robustness checks

We conduct a series of additional empirical analyses to explore the validity of our research design and to test whether our results are sensitive to our modelling choices or variable creation.

Addressing SUTVA

A key identification assumption of our heterogeneous difference-in-difference research design is that there are no spillovers or indirect effects of the regulation on firms in non-TCZ prefectures (i.e., the stable unit treatment value assumption (SUTVA) holds). A potential threat to this assumption is that firms in unregulated regions could become more productive once the policy is implemented, since they do not face the regulatory costs that their competitors in regulated regions face. However, this would attenuate our estimates, if anything. Spatial sorting could also introduce bias. Firms that exit could have moved and re-opened in non-TCZ regions. That said, it is extremely costly to move large manufacturing and power plants—most likely much costlier than complying with the regulation for most firms—and coal mining can only occur where coal already exists.

A more likely scenario that could bias our results in either direction is related

to migration of high-skilled workers. Recent work has shown how pollution has increased migration away from polluted cities in China, primarily for well-educated workers (Chen et al., 2017a, Khanna et al., 2019). Since non-TCZ regions are less polluted on average, high-skilled workers may have been moving out of TCZ prefectures already, which would increase productivity of firms in non-TCZ regions. Indeed, Khanna et al. (2019) find that pollution-induced migration substantially reduces aggregate productivity when high-skilled workers leave. On the other hand, the TCZ regulation could have induced migration of high-skilled workers towards TCZ regions given the pollution reductions, which could dampen aggregate productivity in non-TCZ regions.

We conduct two sets of analyses to provide evidence that the regulation did not induce migration. First, we classify firms by technology intensiveness according to the OECD (2011)'s criteria and examine whether the regulation has a differential impact on TFP for high- and low-tech firms, assuming that high-tech firms employ more high-skilled workers. We would expect high-tech firm productivity to increase by more than low-tech firms if the regulation induced migration, as more educated workers would raise average productivity. Second, we estimate the regulation's effect on the fraction of firms and workers in each prefecture that are in high-tech industries.[23] If workers or firms in high-tech industries move to TCZ regions, we would expect these measures to increase.

Table 5.11 in the Supplementary Information provides the findings, which are consistent with there being no movement of high-tech workers or firms. In Columns 1 and 2, we use our firm-level data to estimate a variation of the baseline model that interacts a dummy variable equal to one if the firm is in a high-tech industry with our two policy treatment variables. There is no differential impact of the regulation on the productivity of high- and low-tech firms when using our baseline measure of TFP as the dependent variable (Column 1) as well as our alternative TFP measure

[23]This is similar to the approach taken by Fu et al. (2021) in their study of how pollution impacts worker productivity.

(Column 2).

In Columns 3 and 4, we aggregate the data to the prefecture-year level and estimate the TCZ regulation's effects on the share of firms (Column 3) and workers (Column 4) that are in high-tech industries. We include year and prefecture fixed effects as well as prefecture-specific trends. We also control for the total number of firms (for each prefecture) in Column 3 and the total number of workers in Column 4 to account for differences in how high-tech firms and high-skilled workers are more likely to move to prefectures with these characteristics in the first place. In both regressions, the effects are not only statistically insignificant but also extremely close to zero in their magnitudes. These four tests provide confidence that there is no systematic movement of high-tech firms or high-skilled workers between TCZ and non-TCZ regions in response to the regulation.

Measuring productivity

Many methods for estimating production functions and TFP have been developed in the literature. We use the Ackerberg et al. (2015) approach in the baseline because it corrects the simultaneity concerns of the commonly used Levinsohn and Petrin (2003) and Olley and Pakes (1996) methods, and it uses intermediate materials as the proxy variable rather than investments, which can be lumpy at the firm level. That said, since there tend to be differences in estimated TFP across methods, we estimate the effects of an alternative TFP measure that we construct following Levinsohn and Petrin (2003) to see whether our results are sensitive to our choice. The results are presented in Column 1 of Table 5.12 in the Supplementary Information and are the same as in our baseline.

A second and perhaps more challenging concern related to our TFP measure is that it is revenue-based rather than quantities-based. This introduces the risk of enhanced productivity being associated with firms increasing mark-ups rather than actually improving productivity. Unfortunately, we do not have access to data on

quantities, which is frequently the case in the literature. However, we indirectly explore whether mark-ups might be driving the results by seeing whether they hold for homogeneous goods markets specifically and for markets that are most competitive. We would expect firms to not increase mark-ups (or at least not by as much) in these contexts, as their market power is limited.

We first estimate our baseline model including only industries producing homogeneous goods, such as of electricity and water, smelting and pressing of metals, petroleum processing, and mining[24]. The results become stronger (see Column 2 of Table 5.12 in the Supplementary Information). Productivity increases for both clean and dirty firms by 8%, although the coefficient for dirty firms relative to clean is negative and just not statistically significant.

Next, we estimate the effects for firms in industries that appear to be the most competitive in the year just prior to policy implementation. Our first approach is to calculate the average market share and Herfindahl–Hirschman Index (HHI), a measure of market concentration (Rhoades, 1993), for each industry—using sales to determine market shares—and include only industries that fall in the bottom quartiles of these distributions. The findings are presented in Columns 3 and 4 of Table 5.12 in the Supplementary Information and are very similar to the baseline. Lastly, we find the number of firms in each industry in the last year just before policy implementation and estimate the effects only for firms in industries in the top quartile of the distribution (Column 5). The results are again about the same as in the baseline.

Additional robustness checks

We conduct a few final tests related to some potential issues in the database and other economic activities throughout the time period that we are studying and

[24]Examples of industries that we do not define as those with homogeneous goods are manufacturing of medical and pharmaceutical products, food and beverage manufacturing, and special machinery manufacturing.

present the findings in Table 5.13 in the Supplementary Information. In Column 1, we drop the years in which the database was in pilot mode (1996-1997), since this is not as comprehensive of a sample and primarily includes SOEs. The magnitude and statistical significance decreases for firms in less pollution-intensive industries, but it remains positive and statistically significant, and the overall effect for firms in more pollution-intensive industries remains statistically zero. In Column 2, we control for China's accession into the the WTO by including a dummy variable for years after 2001 and also interacting this dummy with the TCZ dummy in case firms in TCZ and non-TCZ prefectures were affected differently. For example, firms that were already more advanced and in industrialised regions may have been more likely to be exporters. The results become stronger, if anything.

In Column 3, we drop prefectures for which the number of unique firms is below 200 to ensure that we are comparing similarly-industrialised prefectures. In Column 4, we drop firms for which their industry CIC changed over our sample period, and in Column 5, we drop the year 2003 since we needed to use an alternative labour variable for this year (see Section 5.8.1 in Supplementary Information for detail). The results remain statistically significant and have similar magnitudes to the baseline findings. Finally, we ensure that our results hold when clustering standard errors at different levels. We cluster by industry in Column 6 to account for how treatment intensity varies by industry, and in Column 7, we cluster at the firm level to account for possible serial correlation in the dependent variable. Our results become stronger in both cases.

Lastly, we conduct the same series of robustness checks but using the alternative TFP measure following Levinsohn and Petrin (2003) and provide the findings in Table 5.14 in the Supplementary Information. The estimates for firms in less pollution-intensive industries range from 3.5% to 5.3% and remain statistically significant, and the effects on the heavier polluters are still zero.

5.5 Mechanisms: why does productivity increase?

We now turn to investigating the underlying mechanisms of our findings and the strategies firms take to comply with the regulation. In doing so, we examine industries with productivity gains versus no effect or losses separately, and we also explore heterogeneity by firm ownership given the political economy of regulation in China. We summarise what conclusions we believe can be drawn about the mechanisms at play at the end of this section.

5.5.1 "Responsive" versus "unresponsive" industries

So we can explore the mechanisms that drive productivity gains, we start by estimating the effects on TFP for each industry separately at the 2-digit CIC level to identify industries in which the regulation enhanced productivity as opposed to reducing it or having no effect. The results are presented in Table 5.15 in the Supplementary Information. It turns out that, even though there were no effects on TFP for heavier-polluting firms on average, productivity does actually increase for three major "dirty" industries: smelting and pressing of ferrous metals (i.e., the steel industry), chemical materials and products, and textiles. This is intuitive as these are industries that are coal-intensive in their production processes, thus facing high compliance costs and having the incentive to make significant adjustments.[25]

Of the less pollution-intensive industries, TFP increases in petroleum and natural gas extraction, ferrous metals mining and dressing, beverage manufacturing, and tobacco processing. The first two industries are not particularly coal-intensive polluters on their own, but they are upstream to pollution-intensive industries. The second two fall within the broader category of industrial polluting-firms being

[25]Steel and chemicals, for instance, each account for 5-6% of total SO_2 emissions in China and the textiles industry accounts for 1-2% of emissions. There are no effects on any other dirty industries. Interestingly, there is no effect on the productivity of thermal power and coal mining firms, the two industries that faced the most specific regulatory requirements with thresholds to meet.

"encouraged" to reduce SO_2 emissions under the regulation.[26]

Before moving on to explore what drives productivity improvements for "responsive" industries as well as whether non-responsive industries may have responded to the regulation but in ways that did not enhance productivity, we re-estimate our baseline model and main set of robustness checks to find the average effect on responsive industries and non-responsive industries separately. The results are provided in Table 5.3. Unsurprisingly, the positive effects become much stronger for firms in less pollution-intensive firms and become positive and large for firms in more pollution-intensive industries. The effects on TFP are 16% and 8%, respectively, when using our baseline model in Column 1. On the other hand, when including the "non-responsive" firms together as a sub-sample, we can see that productivity decreases for firms in the dirtiest industries by about 6%. These findings therefore suggest that they do indeed respond as well but in ways that do not enhance their productivity, which we further explore in the next sections.

[26]The only two industries with declines in TFP were natural gas suppliers (CIC 45) and transportation equipment manufacturing (CIC 37). Both of these are low SO_2 emitters. The latter has a small sample size, and when grouped with other equipment manufacturing, the effect goes away. We cannot point to an explicit mechanism for natural gas suppliers' decreased productivity, but one potential explanation is substitution between gas and cleaner coal.

Table 5.3: Effects on TFP for firms in "responsive" versus "non-responsive" industries

Outcome Variable (log):	TFP (1)	TFP (2)	TFP (3)	TFP (4)	TFP (5)	TFP (6)	TFP (7)	TFP (8)
Panel A: "Responsive"								
TCZ * Post-Policy	0.160***	0.117***	0.161***	0.174***	0.141***	0.155***	0.160***	0.160***
	(0.041)	(0.039)	(0.042)	(0.042)	(0.041)	(0.041)	(0.040)	(0.035)
TCZ * Post-Policy * Polluter	-0.077*	-0.032	-0.077*	-0.092**	-0.056	-0.069*	-0.077*	-0.077**
	(0.041)	(0.041)	(0.041)	(0.042)	(0.042)	(0.041)	(0.032)	(0.038)
Observations	144,451	135,835	144,451	142,595	138,311	132,264	144,451	144,451
Panel B: "Non-Responsive"								
TCZ * Post-Policy	0.025	0.021	0.037	0.026	0.023	0.026	0.025*	0.025***
	(0.022)	(0.021)	(0.023)	(0.022)	(0.021)	(0.022)	(0.014)	(0.009)
TCZ * Post-Policy * Polluter	-0.063***	-0.068***	-0.063***	-0.063***	-0.061***	-0.060***	-0.063**	-0.063***
	(0.020)	(0.021)	(0.020)	(0.021)	(0.021)	(0.020)	(0.028)	(0.014)
Observations	618,501	592,466	618,501	607,662	583,479	564,245	618,501	618,501
Baseline	x							
Drop Pilot Data		x						
Control for WTO Entry			x					
Drop if Low Firm Count				x				
Drop Changed Industry					x			
Drop 2003						x		
Cluster SEs by Industry							x	
Cluster SEs by Firm								x

Notes: Table reports the regulation's effects on TFP (log) when including only firms in "responsive" industries (i.e., industries for which the regulation enhanced TFP). Column 1 is comparable to the baseline main results in Column 2 of Table 5.2. In Columns 2-7, we conduct the same set of robustness checks as we do for for the full sample in Table 5.13 in the Supplementary Information. All baseline FEs are included. Standard errors are clustered at the prefecture level in all cases except Columns 7-8. Asterisks denote *p <0.10, **p <0.05, ***p <0.01.

5.5.2 Inputs, output, and single-factor productivity

One potential explanation of changes in productivity is input reallocation or changes in input levels. In the industrial sector context, technology adoption is often an important channel for reducing emissions (Popp, 2011, Dechezleprêtre et al., 2013), and end-of-pipe and scrubber solutions are large capital investments. Labour could also increase since continuous operation of pollution abatement equipment tends to require more workers. On the other hand, firms could find innovative ways to use their capital, labour, and intermediate inputs more efficiently, such as by using lower sulphur-content fuel in their production processes (which tends to be more efficient) or adopting new management practices that improve operational efficiency without changing inputs, resulting in increased single-factor productivity via an output channel.

We test whether productivity increases through the input or output channels by estimating the regulation's effects on inputs (labour, capital, and intermediate inputs), output (value-added and sales), and single-factor productivity, which we define as value-added divided by each input. The results are presented in Panels A and B of Table 5.4 for responsive and non-responsive industries, respectively.

The overall takeaway from this set of results is that productivity gains for firms in responsive industries appear to be driven by an output effect whereas firms in non-responsive industries start scaling down production. For responsive firms, there are no changes in input (Columns 1-3), but both value-added and sales increase by 17% and 10%, respectively (Columns 7 and 8), leading to increases in all three measures of single-factor productivity (Columns 4-6). On the other hand, labour and intermediate inputs both decrease for dirty firms in non-responsive industries, but capital increases for relatively clean. As value-added and sales decrease in these industries for dirty firms while there is no change in output for their relatively clean counterparts, these findings suggest that the cleaner firms invest in new ma-

chinery—and they can bear the costs without it hurting performance—whereas the dirtiest firms (facing the highest compliance costs) start to scale down production but not yet retiring old equipment.

Table 5.4: Effects on inputs, output, and single-factor productivity

Outcome variable (log):	labour	Capital	Intermed.	VA/L	VA/K	VA/M	VA	Sales
	(1)	(2)	(3)	(4)	(5)	(6)	(7)	(8)
Panel A: "Responsive"								
TCZ * Post-Policy	0.011	0.030	0.065	0.159***	0.140***	0.106***	0.170***	0.102***
	(0.023)	(0.036)	(0.041)	(0.041)	(0.045)	(0.033)	(0.044)	(0.039)
TCZ * Post-Policy * Polluter	-0.007	-0.016	-0.023	-0.077*	-0.068	-0.061*	-0.084*	-0.055
	(0.025)	(0.034)	(0.040)	(0.040)	(0.044)	(0.032)	(0.044)	(0.039)
Observations	144,512	144,512	144,512	144,512	144,512	144,512	144,512	144,393
Panel B: "Non-Responsive"								
TCZ * Post-Policy	0.018	0.031**	0.018	0.018	0.005	0.019	0.037	0.026
	(0.012)	(0.014)	(0.019)	(0.022)	(0.023)	(0.019)	(0.023)	(0.019)
TCZ * Post-Policy * Polluter	-0.044***	-0.045***	-0.070***	-0.038**	-0.037	-0.012	-0.082***	-0.070***
	(0.013)	(0.015)	(0.019)	(0.019)	(0.022)	(0.018)	(0.022)	(0.018)
Observations	618,723	618,723	618,723	618,723	618,723	618,723	618,723	618,175

Notes: Table reports the effects of the TCZ regulation on inputs (labour, capital, and intermediate inputs in Columns 1-3) and single-factor productivity (measured as value-added over each input in Columns 4-6). Firms in responsive industries are included in Panel A and those in non-responsive industries are in Panel B. All baseline FEs are included. Standard errors are clustered at the prefecture level. Asterisks denote *p <0.10, **p <0.05, ***p <0.01.

5.5.3 Firm sorting

Another potential explanation for improved productivity is that the regulation drove out inefficient firms. Resource misallocation is common in developing countries, which has been shown to be the case for China (Hsieh and Klenow, 2009, He et al., 2020), and turnover and reallocation across firms could enhance average productivity. While He et al. (2002) document exit of coal mining firms, this is not one of the industries for which productivity increases, so further investigation of whether

firm sorting can explain the results is needed.

To explore whether this occurs, we estimate the effects of the regulation on exit for responsive and non-responsive firms including the full sample and then by splitting the samples by their median pre-policy productivity level (see Table 5.5). On average, there is no effect on exit for responsive and non-responsive industries, but when splitting the sample, we can see that the least productive firms do indeed exit. The regulation increases the propensity to exit by 5% for firms in responsive industries and 2.5% for those in non-responsive industries. These findings suggest that efficient firm exit may be at play and we explore this further later when examining private and state-owned enterprises separately.

Table 5.5: Effect of the TCZ regulation on firm exit

Outcome Variable:	Exit	Exit	Exit
Sample:	Full	Least Productive	Most Productive
	(1)	(2)	(3)
Panel A: "Responsive"			
TCZ * Post-Policy	0.010	0.050*	0.014
	(0.016)	(0.026)	(0.017)
TCZ * Post-Policy * Polluter	0.012	-0.015	0.011
	(0.015)	(0.025)	(0.017)
Observations	144,512	64,592	79,914
Panel B: "Non-Responsive"			
TCZ * Post-Policy	0.010	0.025**	0.007
	(0.009)	(0.011)	(0.010)
TCZ * Post-Policy * Polluter	0.012	0.008	0.008
	(0.008)	(0.010)	(0.009)
Observations	618,723	283,309	335,411

Notes: Table reports effects of the TCZ regulation on exit for responsive (Panel A) and non-responsive (Panel B) industries. The full sample is included in Column 1 and the samples are split by the median pre-policy productivity levels in Columns 2 and 3. All baseline FEs are included. Standard errors are clustered at the prefecture level. Asterisks denote *p <0.10, **p <0.05, ***p <0.01.

5.5.4 Human health and worker performance

There is increasing evidence in the literature that pollution impacts worker productivity and performance as well as cognition, and thus another potential channel through which the regulation impacts productivity could be through the pollution reduction itself. However, two pieces of evidence from preceding sections suggest that this is not at play. First, when considering single-factor productivity, we would expect only (or primarily) labour productivity to be significant if health can explain the findings, but we find that capital- and intermediate input productivity also increase substantially, and the magnitudes of the effects across the three measures are similar. Furthermore, we would expect the effects to be largest for workers in high-tech industries that are more likely to require high-skilled labour. We find no statistical differences for firms in high- and low-tech industries, however, when we access SUTVA in Section 5.4.2.

5.5.5 Induced innovation

Lastly, productivity improvements could be driven by innovative activity. Firms can innovate in various ways, such as by developing and adopting new technologies, processes, and practices. We first explore this potential channel by examining whether firms start investing in more high-skilled labour (e.g., scientists and engineers), estimating the effects on average wages (Column 1 of Table 5.6). Average wages should increase if firms engage in innovative activities, as this is the type of human capital required. We find that this is indeed the case for both responsive and non-responsive cleaner firms but the effect is about double the size for responsive firms—responsive firms increase average wages by 8.8% and non-responsive firms' average wages increase by 4.5%.[27] Dirty firms also increase average wages by 8.8% in responsive industries and 2.3% in non-responsive industries.

[27]Since labour does not change, these effects are driven by higher wages per worker.

If high-skilled labour contributes to productivity improvements, these findings raise the question as to why there is no change in productivity for non-responsive industries as well. One explanation could be management quality. A key input required for turning the knowledge produced by such workers into productivity gains is high-quality management of their activities and adoption of the newly created innovations. We estimate the effects on expenditures specifically on management and find that it is only cleaner responsive firms that invest more in management in response to the regulation, suggesting that more time and resources allocated to management are necessary to translate the high-skilled human capital into improved productivity.

Table 5.6: Evidence of innovative activity: effects on wages, management expenditures, and new products

Outcome Variable (log):	Avg. Wages (1)	Manage. Exp per Worker (2)	New Product Output (3)	New Product % of Output (4)
Panel A: "Responsive"				
TCZ * Post-Policy	0.088***	0.074**	0.104	-0.001
	(0.028)	(0.035)	(0.089)	(0.004)
TCZ * Post-Policy * Polluter	-0.026	-0.079*	-0.043	-0.001
	(0.027)	(0.043)	(0.085)	(0.004)
Observations	143,863	142,983	126,374	126,374
Panel B: "Non-Responsive"				
TCZ * Post-Policy	0.045***	0.007	0.036	-0.001
	(0.014)	(0.016)	(0.062)	(0.002)
TCZ * Post-Policy * Polluter	-0.022*	-0.022	-0.022	-0.001
	(0.013)	(0.018)	(0.045)	(0.002)
Observations	615,953	608,502	548,806	548,806

Notes: Columns 1 and 2 provide the effects on average wages and expenditures on management per employee, respectively. Columns 3 and 4 provide the effects on new product output level and its percentage of total output, respectively. All baseline FEs are included. Standard errors are clustered at the prefecture level. Asterisks denote *p <0.10, **p <0.05, ***p <0.01.

Finally, if these results do reflect innovative activity, understanding whether firms are producing new technologies as opposed to improving their processes and practices can provide deeper insight into the strategies they take to comply with the regulation. We estimate the effects of the regulation on new product output and its percentage of total output (Columns 3 and 4 of Table 5.6) to test this, and the results are consistent with innovating in processes and practices, as there are no effects on new product output for any set of firms. This aligns with the type of innovative activity we would expect for these firms as well—unless a firm's main product is already an environmental technology, they are unlikely to start producing one due to the regulation as opposed to either adopt new technologies or adjust their operations in ways that help reduce pollution.

5.5.6 Summary of mechanisms

Taken together, these sets of results suggest that firms achieve productivity improvements through an output effect rather than by reallocating or increasing inputs, and there are two likely mechanisms underlying the main findings. First, firms that were previously less productive exit, increasing average productivity. This is intuitive given how the additional costs imposed by the regulation could make it unmanageable to compete with firms that are already more productive. Second, although we remain cautious in our interpretation, our findings suggest that the regulation may have induced process and practice innovation. Moreover, when firms do invest in higher-skilled human capital, investing more time and resources into management appears to be important for innovation to improve productivity.

5.6 Heterogeneity by firm ownership

We now probe the underlying drivers of our findings further by examining exit and innovative activities for private firms and state-owned firms (SOEs) separately.

Given China's long history of protecting SOEs (Eaton and Kostka, 2017)—such as by favouring them in product and services purchasing (e.g., public procurement) to help them survive or subsidising them such that they face lower or zero compliance costs and have no incentive to change behaviour—the results of the previous section might mask important heterogeneity in firms' strategies that can shed light on the political economy of environmental regulation. Furthermore, such heterogeneity may have implications for growth and competitiveness given how private firms are on average significantly more productive than state-owned enterprises in China.

First, we estimate the effects of the regulation on exit for firms in responsive industries for SOEs and private firms (see Table 5.7). We start by using the full baseline samples (Column 1) and then split them based on firm age (Columns 2-3) and size (Columns 4-5), as evidence in the innovation economics literature has shown that start-up firms are more likely to contribute significant innovations relative to older and larger firms.

We find that it is almost entirely private firms that exit. On average, the propensity for private firms to exit increases by 5% whereas there is no effect on SOEs, and it is only older private firms that exit. This is in line with our previous findings of less productive firms exiting, as older firms are more likely to be using older and less efficient machinery, and it also begins to suggest that local government officials responsible for the regulation's enforcement may be protecting SOEs. Such behaviour is not uncommon in China—local officials typically appoint the executives of SOEs and benefit from the relationships (Barwick et al., 2021, Lei, 2021). This creates strong incentives for the government to help SOEs survive, reducing the pressure on state sector firms to invest in pollution-abatement technology or activities. On the other hand, the propensity to exit increases by about 15% for both state-owned and private small firms, although the sample size is much larger for private firms, so the level effect is much smaller for SOEs. This is consistent with China's known practice of protecting larger firms when new regulations are imposed, and although

the findings overall are still mostly driven by private firm exit, this does provide evidence of there being some regulation enforcement for SOEs as well.

Table 5.7: Effect on exit in responsive industries by firm ownership, age, and size

Outcome Variable: *Sample:*	Exit Full (1)	Exit Young (2)	Exit Older (3)	Exit Small (4)	Exit Larger (5)
Panel A: State-Owned Firms					
TCZ * Post-Policy	0.008	0.144	0.009	0.156*	-0.003
	(0.024)	(0.151)	(0.024)	(0.082)	(0.023)
TCZ * Post-Policy * Polluter	0.025	-0.088	0.026	0.088	0.033
	(0.024)	(0.161)	(0.024)	(0.087)	(0.024)
Observations	34,134	880	33,004	3,095	30,459
Panel B: Private Firms					
TCZ * Post-Policy	0.052**	0.043	0.054**	0.153**	0.033*
	(0.020)	(0.056)	(0.023)	(0.071)	(0.018)
TCZ * Post-Policy * Polluter	-0.018	-0.016	-0.016	-0.053	-0.003
	(0.018)	(0.060)	(0.020)	(0.075)	(0.017)
Observations	108,670	5,680	101,247	11,859	94,697

Notes: Table reports effects on exit by firm ownership, age, and size for firms in responsive industries. The full sample is included in Column 1. Firms that are younger and older than 5 years old are included in Columns 2 and 3, respectively. Firms with fewer or more than 50 employees are included in Columns 4 and 5. All baseline FEs are included and standard errors are clustered at the prefecture level. Asterisks denote *p <0.10, **p <0.05, ***p <0.01.

Finally, we examine whether both SOEs and private firms engage in innovative activity, as this was the second mechanism underlying our baseline results. The results from estimating the effects on average wages (Column 1), management expenditures per worker (Column 2), new product output per employee (Column 3), and the percentage of output associated with new products (Column 4) are presented in Table 5.8 for SOEs and private firms. If we interpret these outcomes as we did before, the findings indicate that it is only private firms that innovate. There are no effects on any of these outcomes for SOEs. Instead, the increases in aver-

age wages are entirely accounted for by private firms, as they increase by 14% for cleaner firms and 7% for dirtier firms. Furthermore, private firms' expenditures in management increase by 10% for cleaner firms (and do not change for dirty private firms). There are again no effects on new product output for either sample, suggesting that private firms' investments in higher-skilled human capital go towards improving operational efficiency rather than developing new technologies.

This heterogeneity suggests that the regulation is likely enforced primarily for private firms—and private firms that do not exit respond in ways that lead to productivity improvements—shedding light on the political economy of environmental regulation. It also provides new insight into how such favouritism and protection of the state sector impacts China's competitiveness. Given how younger and smaller firms typically are important drivers of innovation relative to their older and larger counterparts, and how private firms are far more productive than SOEs in China on average, the exit of these exact sets of firms may have negative consequences for economic growth. If their exit is indeed due to unequal enforcement for SOEs and private firms, political protectionism may be a detriment to the potential productivity gains that could be achieved through environmental regulation.

Table 5.8: Effect on wages, management expenditures, and new product output by firm ownership

Outcome Variable:	Avg. Wages (1)	Manage. Exp per Worker (2)	New Product Output (3)	New Product % of Output (4)
Panel A: State-Owned Firms				
TCZ * Post-Policy	0.027	0.070	0.129	-0.001
	(0.036)	(0.047)	(0.127)	(0.005)
TCZ * Post-Policy * Polluter	-0.003	-0.052	-0.216	-0.005
	(0.040)	(0.055)	(0.142)	(0.006)
Observations	33,863	33,712	27,933	27,933
Panel B: Private Firms				
TCZ * Post-Policy	0.143***	0.096*	0.048	-0.002
	(0.035)	(0.049)	(0.125)	(0.007)
TCZ * Post-Policy * Polluter	-0.070**	-0.107*	0.053	0.003
	(0.034)	(0.060)	(0.119)	(0.006)
Observations	108,310	107,593	96,980	96,980

Notes: Table reports effects on average wages (Column 1), management expenditures (Column 2), new product output per employee (Column 3), and the percentage of output that is from new products (Column 4) by firm ownership. All baseline FEs are included and standard errors are clustered at the prefecture level. Asterisks denote *p <0.10, **p <0.05, ***p <0.01.

5.7 Conclusion

Policy goals related to protecting the environment and driving economic growth are often pitted against each other, implying that their objectives cannot be achieved simultaneously. Our paper shows that they are not always at odds. By estimating the effects of an environmental regulation on firms across the entire industrial sector, we find that the net effect on productivity is positive, suggesting that environmental regulation can actually be an effective tool for fostering development.

Our results importantly rely on considering how regulation impacts not just the "dirtiest" industries that face the highest compliance costs. Ignoring the effects on less pollution-intensive industries leaves out a critical component of industrial activity, and if we had done so, we would have drawn the opposite conclusions. Such effects should be included when assessing the costs and benefits of environmental regulation.

5.8 Supplementary information

5.8.1 Raw data preparation

Firm-level data

We obtain annual firm-level data for the period 1996-2006 from the China Industrial Enterprise Database (CIED). The database provides three types of variables: basic information (firm ID, location, total employment etc.), production information (main product, industrial output etc.) and financial information (capital stock, revenue, profit, wage etc.). In our treatment of the data, we follow others in the literature and draw heavily from Brandt et al. (2012). We link firms over time using firms' numerical ID, and where possible, other information including firms' names, legal person, phone number, city code, founding year, industry code etc. We match the sample of two consecutive years first and then expand it to three consecutive years. For more details, please refer to Brandt et al. (2012)'s appendix. One extension that we implement beyond their approach is that we also include the pilot year data from 1996-1997. Although there were some changes in the format of firm ID codes, we are able to match a large portion of the data (approximately 70%) based on the other information, and including these data are helpful for the methods we use in this paper since they expand our pre-policy period.

The CIED contains two variables concerning employment—the number of employees at the end of the year and the average number throughout the year. We use the former to represent employment for all years except 2003, where we use the latter as the former is missing for the year 2003. We conduct a robustness check that drops 2003 in the paper to ensure that this does not affect our results.

We drop observations that appear to contain errors in the key variables that we use. That is, we drop observations for which employment, wages, capital, added value, or gross industrial output are negative (2.5% of the observations). We also

drop observations for which we are missing labour or fixed assets data and cases that violate standard accounting principles: observations for which the sum of liquid assets and fixed assets are higher than total assets, current assets are higher than total assets, or fixed assets are greater than total assets (0.08%). At this point, we have an unbalanced panel of 1,874,627 observations for 506,414 unique firms from 1996-2006.

Firm ownership

We categorise firms as being either "state-owned" or "private" according to their paid-up capital sources. If a firm receives more than 50% paid-up capital from state sources in that year, we consider it to be a state-owned enterprise (SOE). We define all other firms as being private, including foreign firms, which are those that receive less than 50% paid-up capital from the state and more than 25% from foreign sources and Hong Kong, Taiwan and Macao sources.

Firm industry

The database covers 40 two-digit Chinese Industrial Classification (CIC) industries, including those in mining, manufacturing, and public utilities. Given that the industry-level SO_2 emission data is only available at the 2-digit industry level (see below), we categorise firms at two-digit industry level. In 2003, the industrial code classification system was revised and several changes were made. To make industry codes comparable, we adjust 1996-2002 observations' industry codes according to the post-2003 version. The industry code used changes for firms sometimes, so we use the mode of industry codes for each firm as the assigned industry for that firm for all years so that we can assign it as being more or less pollution-intensive in our heterogeneous difference-in-differences framework.[28]

Firm age

We assume all firms are founded after the year 1800 and consider founding year

[28]These inconsistencies are largely due to ambiguous definitions. For example, manufacturing of fire van can be assigned to Industry 36 (manufacturing of transport equipment) or Industry 37 (manufacturing of dedicated devices).

missing if firms reported an earlier founding year. If firms have indicated different founding years in each survey, we use the mode of founding years within-firm for calculating the firm's age.

Prefecture-level data

We collect sociodemographic prefecture data from China City Statistical Yearbook. This yearbook contains variables both at prefecture and district level. As we assign TCZ status at prefecture level, we use all variables at prefecture level to be consistent. The data is primarily used to examine the pre-policy period prefecture-level characteristics, such as GDP per capita and population. We also collect prefecture-level SO_2 data from China Environmental Yearbook and NASA MERRA-2 to test the effectiveness of the policy in reducing SO_2 pollution.

Industry-level data

We use industry-level SO_2 emission and coal consumption data to assign industries as being more or less pollution-intensive, which is required for our heterogeneous difference-in-differences research design. In China, SO_2 emissions are highly correlated with coal consumption, and some of the TCZ regulation's more explicit measures for reducing SO_2 emissions specifically targeted the life cycle of coal. Therefore, we consider SO_2 emission and coal consumption as relevant indicators for deciding whether an industry is more or less pollution-intensive.

Our approach entails computing industry-level pollution intensity and then following Greenstone (2002) to define pollution-intensive. We gather data on SO_2 emissions from the China Statistical Yearbook 2002, which contains data for the year 2001. Unlike city-level emission data, which are subject to potential misreporting by local government officials (Karplus et al., 2018), industry-level emission data are less likely to be manipulated as emission levels of different industries are inherently heterogeneous, depending on the industry's characteristics i.e. raw ma-

terials used, manufacturing process, residues produced. The reason we use 2001 data is that it contains 40 industries (compared to 20 industries in 1997) and also includes the number of firms in each industry, which allows us to compute average SO_2 emissions per firm in each industry. To relieve the concern that the 2001 data may be affected by the implementation TCZ policy, we find high correlations between data in 1997 and 2001, i.e., 0.98 for SO_2 emission variable and 0.99 for coal consumption variable. This confirms that the 2001 data is a fairly good proxy of industry polluting levels, as suggested by Cai et al. (2016b).

We calculate each industry's share of total SO_2 emission and coal consumption. If the share is more than 1 percent for both indicators, we define this industry as being "more pollution-intensive" (see Table 5.9 in the Supplementary Information). One special case is that we break up the electricity, steam and hot water supply industry (CIC code "44") based on major differences in pollution intensity. The electricity, steam and hot water supply industry accounts for more than 50% of total coal consumption and SO_2 emission, and this is mostly contributed by the thermal power generation (CIC code "4411"). Therefore, we include thermal power in the more pollution-intensive group whereas we define other power generation industries contained in CIC code "44" (hydro, wind, solar, nuclear, etc.) as less pollution-intensive.

This results in a set of ten industries as being in the more pollution-intensive category, and the industries align closely with those that are defined as being SO_2 regulated in Greenstone (2002). Greenstone (2002) defines the following industries as being SO_2 regulated: Pulp and paper(corresponding to CIC code "22"), Inorganic chemicals(CIC code "26"), Petroleum refining(CIC code "25"), Stone, clay, glass, and concrete (CIC code "31"), Iron and steel (CIC code "32") and Nonferrous metals (CIC code "33"). These six industries are all covered in our defined pollution-intensive industries, and we include four more that qualify given the definition that we apply: coal mining and dressing (CIC code "06"), agricultural food processing

(CIC code "13"), textiles (CIC code "17"), and electricity, steam and hot water supply (CIC code "4411").

Finally, we compare the industries that we define as being more pollution-intensive with those highlighted in the TCZ policy document and targeted by the regulation. We cover three out of four industries mentioned in the policy document: chemical industry (CIC code "26"), metallurgical industry (CIC code "32"), and nonferrous metal industry (CIC code "33"). The only one that we do not cover is the building material industry, which is not included in our data.

Final preparation and the TFP Ratio

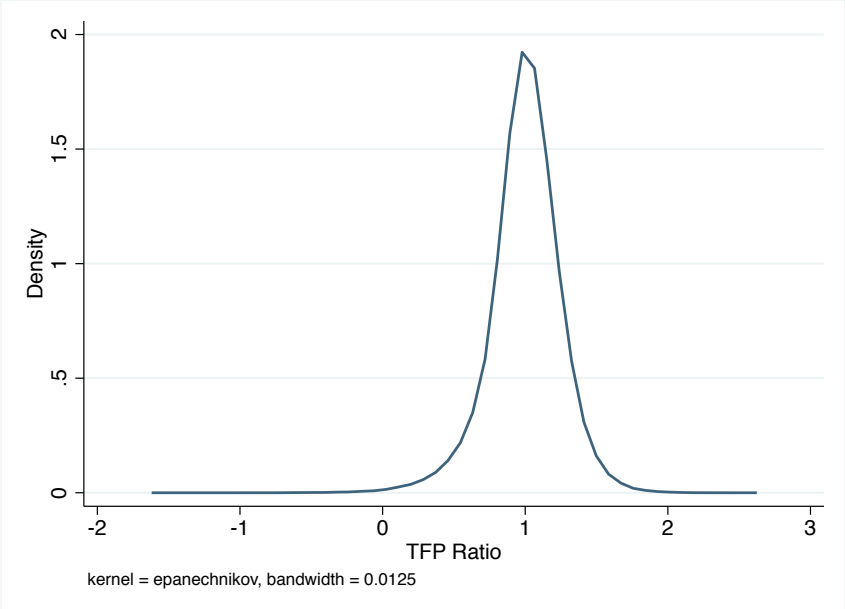
All nominal values (except industrial intermediate input) in CIED dataset and China City Statistical Yearbook are converted to real values in 1998 by using output deflators. The output deflators are constructed following Yang (2015) by using price indexes extracted from the "Urban Price Yearbook 2011" published by the National Bureau of Statistics (NBS). We convert the 2-digit industry level "total output price index (chain)" in the years 1985-2010 to fixed-base index using 1998 as the base year. The input deflators, which are used to deflate nominal industrial intermediate input, are constructed by using National Input-Output (IO) tables in 1997, 2002, and 2007. Precisely, we use the industry-level intermediate input indicated in IO tables as weights to convert output deflators to input deflators. The years before 2000 were using the 1997 IO tables, the years 2001-2005 and post-2006 were using IO tables in 2002 and 2007 respectively. As noted by Yang (2015), this approach takes into account the dynamics of input price in different sectors. We use these deflators to deflate nominal values to real values in 1998.

Once we match all of the aforementioned data and keep observations for which we have the information needed to determine treatment status, we keep only firms for which we observe at least once before and once after the policy was implemented, since we conduct a within-firm analysis. This is the step that leads to the greatest

number of observations dropped, leaving us with a little over 800,000 observations. When we go through the matching process, we also check for additional cases for which there appears to be data entry or reporting errors or when we are missing values for the key variables we need to estimate TFP and other key variables we use throughout the analysis. We also drop cases for which employment is less than eight employees, since firms that are below this threshold lack reliable audit systems. The final unbalanced panel that we use as the baseline data set includes 763,240 observations from 1996 to 2006 for 127,757 unique firms.

The mean estimated (log) TFP is about 5.47 with a standard deviation of 1.59 (for firms in TCZ and non-TCZ prefectures together). To illustrate its dispersion, we construct the "TFP Ratio" following Hsieh and Klenow (2009) and He et al. (2020)—measured as the ratio between the firm's (log) TFP and the firm's industry's average (log) TFP—and plot its distribution in Figure 5.4 in the Supplementary Information. The ratio appears to follow a normal distribution and exhibits about the amount of dispersion expected for a developing country.

Figure 5.4: Dispersion of log(TFP)



This figure plots the "TFP Ratio" for the full sample period as measured by the ratio between the firm's (log) TFP and the industry-year average (log) TFP.

5.8.2 Supplementary tables

Table 5.9: Industry list, emissions data, and pollution-intensive industry assignments

2-Digit CIC code	Industry name	SO_2 per emissions (2001 data) (t SO_2/firm/year)	SO_2 share (2001 data)	Coal consumption (2001 data)	Defined as pollution intensive?
06	Coal Mining and Dressing	79.78	1.37%	6.05%	Yes
07	Petroleum and Natural Gas Extraction	184.08	0.24%	0.70%	
08	Ferrous Metals Mining and Dressing	51.59	0.21%	0.06%	
09	Nonferrous Metals Mining	56.98	0.43%	0.08%	
10	Nonmetal Minerals Mining	122.99	0.49%	0.42%	
11	Other Mining	18.75	0.01%	0.00%	
13	Agricultural Food Processing	53.36	1.28%	1.24%	Yes
14	Food Manufacturing	38.01	0.65%	0.54%	
15	Beverage Manufacturing	57.06	0.94%	0.54%	
16	Tobacco Manufacturing	61.32	0.10%	0.11%	
17	Textile	38.29	1.70%	1.17%	Yes
18	Garments and Fiber Products	15.02	0.07%	0.11%	
19	Leather, Fur, and Feather Products	18.32	0.12%	0.06%	
20	Timber Processing and Related Products	58.60	0.24%	0.18%	
21	Furniture Manufacturing	13.45	0.02%	0.04%	
22	Papermaking and Paper Products	79.76	2.58%	1.49%	Yes
23	Printing and Related Products	4.78	0.02%	0.04%	
24	Cultural, Educational and Sports Products	43.95	0.07%	0.02%	
25	Petroleum Processing and Coking	394.67	2.75%	7.43%	Yes
26	Raw Chemical Materials and Products	112.95	5.43%	6.30%	Yes
27	Medical and Pharmaceutical Products	32.81	0.46%	0.44%	
28	Chemical Fiber Manufacturing	434.76	0.84%	0.70%	
29	Rubber Products	67.44	0.30%	0.23%	
30	Plastic Products	14.18	0.09%	0.12%	
31	Nonmetal Mineral Products	118.50	11.44%	8.01%	Yes
32	Smelting and Pressing of Ferrous Metals	414.21	6.01%	9.47%	Yes
33	Smelting and Pressing of Nonferrous Metals	490.11	4.96%	1.12%	Yes
34	Metal Product Manufacturing	6.29	0.21%	0.20%	
35	General Machinery Manufacturing	13.73	0.26%	0.31%	
36	Special Machinery Manufacturing	28.59	0.29%	0.27%	
37	Traffic Equipment Manufacturing	29.23	0.38%	0.59%	
39	Electric Apparatus Manufacturing	23.65	0.22%	0.15%	
40	Electronic Apparatus Manufacturing	13.88	0.10%	0.05%	
41	Instrument, Meter and Office Equipment	6.16	0.02%	0.02%	
42	Handicrafts and other Manufacturing	13.86	0.06%	0.19%	
44	Electricity, Steam and Hot Water Supply	4521.33	55.36%	50.59%	Yes*
45	Production and Supply of Gas	227.71	0.18%	0.83%	
46	Production and Supply of Tap Water	22.77	0.02%	0.04%	

*Only coal-fired electric power supply and production firms are classified as pollution-intensive (CIC code 4411).

Table 5.10: Effect of the TCZ regulation on SO_2 concentration levels

Sample Period:	1988-2008	1996-2006	1988-2008	1996-2006
Outcome Variable:	SO_2	SO_2	SO_2	SO_2
	(1)	(2)	(3)	(4)
TCZ * Post-Policy	-0.037***	-0.040***	-0.041***	-0.046***
	(0.011)	(0.013)	(0.012)	(0.015)
Observations	87,444	45,804	7,287	3,817
Month-year-prefecture data	x	x		
Year-prefecture data			x	x
Month FEs	x	x		
Prefecture FEs	x	x	x	x
Year FEs	x	x	x	x
Prefecture x Year Trends	x	x	x	x

Notes: Table provides the effects of the TCZ regulation on SO_2 concentration levels (ug/m^3) at the prefecture-year-month level (Columns 1-2) and the prefecture-year level (Columns 3-4) using data from NASA. All data that are available are used in Columns 1 and 3, and we limit the sample to the period we study (1996-2006) in Columns 2 and 4. Standard errors are clustered at the prefecture level. Asterisks denote $^*p < 0.10$, $^{**}p < 0.05$, $^{***}p < 0.01$.

Table 5.11: Addressing SUTVA concerns

Outcome Variable:	TFP (log)	TFPv2 (log)	% Firms High-Tech	% labour High-Tech
	(1)	(2)	(3)	(4)
TCZ * Post-Policy	0.043*	0.045**	-0.005	-0.010
	(0.023)	(0.023)	(0.007)	(0.007)
TCZ * Post-Policy * Polluter	-0.054***	-0.057***		
	(0.019)	(0.019)		
TCZ * Post-Policy * High-Tech	0.001	-0.004		
	(0.023)	(0.023)		
TCZ * Post-Policy * Polluter * High-Tech	0.037	0.040		
	(0.040)	(0.040)		
Observations	762,957	763,237	3,566	3,566
Firm-Year Data	x	x		
Prefecture-Year Data			x	x
Firm FEs	x	x		
Industry x Prefecture FEs	x	x		
Industry x Year FEs	x	x		
Prefecture x Year Trends	x	x	x	x
Prefecture FEs			x	x
Year FEs			x	x

Notes: Table reports from two sets of results addressing SUTVA concerns related to movement of firms and people in high-tech industries. In Columns 1 and 2, we use the baseline firm-year data with the two versions of (log) TFP as outcome variables. In Columns 3 and 4, we use prefecture-year data, and the outcome variable is the percent of firms that are in high-tech industries in Column 3 and the percent of workers that are in high-tech industries in Column 4. In Columns 1-2, the two-way interactions are included as well as the "polluter" indicator. The other two main effects are absorbed by the fixed effects. Standard errors are clustered at the prefecture level. Asterisks denote $^{*}p < 0.10$, $^{**}p < 0.05$, $^{***}p < 0.01$.

Table 5.12: Testing sensitivity of results to TFP measure choice

Outcome Variable (log):	LP TFP (1)	TFP (2)	TFP (3)	TFP (4)	TFP (5)
TCZpost1998	0.043**	0.077***	0.051*	0.057*	0.052*
	(0.022)	(0.027)	(0.028)	(0.029)	(0.027)
TCZpost1998dirty	-0.049***	-0.041	-0.060**	-0.072***	-0.061***
	(0.017)	(0.032)	(0.024)	(0.023)	(0.023)
Observations	763,237	158,696	407,318	362,027	436,624
Sample used for regressions:					
Full	x				
Homogeneous Goods Markets		x			
Q1 of Industry Market Share			x		
Q1 of Industry HHI				x	
Q4 Industry No. of Firms					x

Notes: Table reports results from tests of whether the results are sensitive to our TFP variable choice. In Column 1, we use the approach of Levinsohn and Petrin (2003) to construct TFP. In Columns 2-5, we limit the sample to explore whether the findings are driven by firm-level mark-ups as opposed to actual productivity increases. Only homogeneous goods markets are included in Column 2. In Columns 3 and 4, firms in industries in the bottom quartiles of the pre-regulation industry-level distribution of market shares and HHI are included, respectively. In Column 5, firms in industries in the top quartile of the industry-level pre-regulation total number of firms distribution are included. Asterisks denote *p <0.10, **p <0.05, ***p <0.01.

Table 5.13: Additional robustness checks for main TFP results

Outcome Variable (log):	TFP (1)	TFP (2)	TFP (3)	TFP (4)	TFP (5)	TFP (6)	TFP (7)
TCZ * Post-Policy	0.035*	0.053**	0.045**	0.040*	0.044**	0.043***	0.043***
	(0.021)	(0.023)	(0.022)	(0.021)	(0.022)	(0.015)	(0.009)
TCZ * Post-Policy * Polluter	-0.044***	-0.048***	-0.050***	-0.044**	-0.045***	-0.048*	-0.048***
	(0.017)	(0.017)	(0.017)	(0.018)	(0.017)	(0.025)	(0.012)
Observations	728,305	762,957	750,262	721,790	696,514	762,957	762,957
Drop Pilot Data	x						
Control for WTO Entry		x					
Drop if Low Firm Count			x				
Drop if Industry Changed				x			
Drop 2003					x		
Cluster SEs by Industry						x	
Cluster SEs by Firm							x
Firm FEs	x	x	x	x	x	x	x
Industry x Prefecture FEs	x	x	x	x	x	x	x
Industry x Year FEs	x	x	x	x	x	x	x
Prefecture x Year Trends	x	x	x	x	x	x	x

Notes: Table reports results from several robustness checks of the baseline results in Column 2 of Table 5.2. In Column 1, we drop observations in the years for which the data gathering was in its pilot stage (1996-97). In Column 2, we include a dummy variable equal to one after 2001 to control for when China entered the WTO. In Column 3, we drop prefectures that have fewer than 200 unique firms to ensure the results are not driven by differences in industrialization. In Column 4, we drop firms if their CIC changed over the sample period. In Column 5, we drop 2003 because of our labour variable being from a different source this year. In Columns 6 and 7, we cluster standard errors by industry and firm, respectively. The two-way interactions are included as well as the "polluter" indicator. The other two main effects are absorbed by the fixed effects. Standard errors are clustered at the prefecture level in all cases except Column 6-7. Asterisks denote *p <0.10, **p <0.05, ***p <0.01.

Table 5.14: Robustness checks for alternative TFP measure results

Outcome Variable (log):	TFP (1)	TFP (2)	TFP (3)	TFP (4)	TFP (5)	TFP (6)	TFP (7)	TFP (8)
TCZ * Post-Policy	0.043**	0.035*	0.053**	0.045**	0.041*	0.044**	0.043***	0.043***
	(0.022)	(0.021)	(0.023)	(0.022)	(0.021)	(0.022)	(0.016)	(0.009)
TCZ * Post-Policy * Polluter	-0.049***	-0.045***	-0.049***	-0.051***	-0.046**	-0.046***	-0.049*	-0.049***
	(0.017)	(0.017)	(0.017)	(0.017)	(0.018)	(0.017)	(0.025)	(0.012)
Observations	763,237	728,578	763,237	750,262	722,059	696,774	763,237	763,237
Baseline Model	x							
Drop Pilot Data		x						
Control for WTO Entry			x					
Drop if Low Firm Count				x				
Drop if Industry Changed					x			
Drop 2003						x		
Cluster SEs by Industry							x	
Cluster SEs by Firm								x
Firm FEs	x	x	x	x	x	x	x	x
Industry x Prefecture FEs	x	x	x	x	x	x	x	x
Industry x Year FEs	x	x	x	x	x	x	x	x
Prefecture x Year Trends	x	x	x	x	x	x	x	x

Notes: Table reports the regulation's effects on TFP (log) when using the alternative measure of TFP described in Section 5.3.2. Column 1 is comparable to the baseline main results in Column 2 of Table 5.2. In Columns 2-8, we conduct the same set of robustness checks as we do for for the full sample. In Column 2, we drop observations in the years for which the data gathering was in its pilot stage (1996-97). In Column 3, we include a dummy variable equal to one after 2001 to control for when China entered the WTO. In Column 4, we drop prefectures that have fewer than 200 unique firms to ensure the results are not driven by differences in industrialisation. In Column 5, we drop firms if their CIC changed over the sample period. In Column 6, we drop 2003 because of our labour variable being from a different source this year. In Columns 7 and 8, we cluster standard errors by industry and firm, respectively. The two-way interactions are included as well as the "polluter" indicator. The other two main effects are absorbed by the fixed effects. Standard errors are clustered at the prefecture level in all cases except Columns 7-8. Asterisks denote $^{*}p < 0.10$, $^{**}p < 0.05$, $^{***}p < 0.01$.

Table 5.15: Effects on TFP for firms in "responsive" industries (separately)

Outcome Variable (log):	TFP	TFP	TFP	TFP
	(1)	(2)	(3)	(4)
Panel A: Most Pollution-Intensive Industries				
	Textiles	Chemicals	Steel	
TCZ * Post-Policy	0.084*	0.085**	0.113*	
	(0.046)	(0.037)	(0.065)	
Observations	56,704	53,116	14,208	
Panel B: Less Pollution-Intensive Industries				
	Petro & Gas Processing	Ferrous Metals Mining & Dress	Beverage Manufacturing	Tobacco Manufacturing
TCZ * Post-Policy	0.435*	0.306*	0.083*	0.231*
	(0.216)	(0.166)	(0.045)	(0.119)
Observations	349	2,644	15,360	2,070
Firm FEs	x	x	x	x
Year FEs	x	x	x	x
Prefecture FEs	x	x	x	x
Prefecture x Year Trends	x	x	x	x

Notes: Table reports the regulation's effects on TFP (log) for "responsive" industries (i.e., industries for which the regulation enhanced TFP). Panel A includes the most pollution-intensive industries with positive effects (textiles, chemicals, and steel). Panel B includes the less pollution-intensive industries with positive effects (petroleum and gasoline processing, ferrous metals mining and dressing, beverage manufacturing, and tobacco manufacturing). Standard errors are clustered at the prefecture level. Asterisks denote $^{*}p < 0.10$, $^{**}p < 0.05$, $^{***}p < 0.01$.

6 Discussions

The discussion chapter summarises the main contributions, highlights the policy implications, presents major limitations, and points out several future research priorities.

6.1 Contributions

This DPhil thesis has advanced the literature on environment and economy through producing three novel and rigorous research papers, which arise from the existing theoretical framework and are motivated by practical policy issues. Precisely, the first paper suggests that environmental risks may increase the likelihood of energy investment failure, highlighting the importance of managing environmental risks through governance. The second paper shows technologies could only play a limited role in avoiding asset stranding under climate constraints, and it is unwise to rely on future technology deployment to reduce transitional risks. The third paper finds that environmental regulation could enhance regulated firms' average productivity level through firm sorting and process innovation, demonstrating that productivity gains and environmental performance could be achieved hand in hand.

These three research papers make several major contributions. The first DPhil paper, to the best of my knowledge, is one of the first empirical studies to statistically investigate the link between environmental risks and energy investment failures. Many studies have attempted to explore the potential risk factors associated with energy investments using ex-ante evaluation methods. Their findings vary with changes in assumptions and scenarios, making the implications less convincing for investors and practitioners. There is a lack of evidence-based ex-post research quantifying the impact of environmental risk on project implementation. Furthermore, despite the considerable attention to planned projects' potential impacts on the nat-

ural environment, there is limited knowledge on whether these proposed projects could be successfully commissioned. The first paper has narrowed the knowledge gap by providing evidence from Chinese overseas investments in the electric power sector. Lastly, this paper advances our understanding of energy-related mobilisations by developing georeferenced proxy variables for the main environmental conflicts synthesised in anecdotal evidence and presenting one of the first systematic statistical evidence of place-based protests' effects in altering the trajectory of power projects.

The second DPhil paper has addressed the unclear question of whether carbon abatement technologies could mitigate asset stranding risk and allow the fossil fuel business to continue for longer. Previous studies have highlighted the stranding risk of investing in fossil fuels, yet the implications may not be salient enough to stop fossil fuel financing for investors who pin their hopes on carbon abatement technologies. The optimistic technology development scenarios look promising, which may mask the uncertainties in their potential in reducing carbon emissions. As decisions have to be made today, this paper offers important insights for investors and policymakers on their decision-making.

The third DPhil paper has provided novel empirical evidence from developing country context to fuel the long literature on environmental regulations and firm productivity. This paper adds new empirical evidence that environmental regulation enhances firm productivity on average across all industrial sectors. Furthermore, this paper is one of the first studies to link productivity improvements with innovation. The exploration of the mechanisms through which firms achieve positive outcomes suggests that process and practice innovation at least partly drives the productivity gains.

Overall, this DPhil thesis is developed based on environmental economics, integrating pieces of knowledge from many other disciplines, such as environmental science, political science, geography, sociology, and climate science. It combines

fine-scale data with diverse forms of novel datasets such as geocoded measurements from satellite sources, mapping of designated protected areas, and artificial intelligence-based media lexical data. It contributes to a more thorough understanding of the interactions between environment and economy, a broad topic that needs an interdisciplinary perspective.

6.2 Policy implications

This DPhil thesis sends three critical implications to policymakers, corporates and investors.

Firstly, investing in coal is highly risky, not only on environmental grounds but also on financial grounds. The second DPhil paper has shown that continuing to invest in fossil fuel power plants is incompatible with achieving climate targets. Plant conversions and carbon abatement technologies are not sufficient to mitigate the stranding risk. In addition to these arguments from an environmental point of view, the first DPhil paper has highlighted that even from a financial perspective, investing in coal is not a wise choice. Deciding to invest in coal is already risky, as coal power projects are more likely to be suspended than solar and wind technologies. Let alone the environmental risks embedded in coal projects that may translate into cancellation or postponement, rendering financiers' investments worthless. Our findings suggest that investors should factor into account the risk of stranding, postponement or cancellation of coal projects when making decisions.

China has announced to stop building new coal power plants overseas in September 2021, a significant step that closes the chapter on public finance for coal. However, Chinese financing is involved in only 13% of the global coal power capacity outside of China that is operating or being developed between 2013 and mid-2019 (Ma and Gallagher, 2021). Lessons learnt from Chinese overseas investments and the fossil fuel stranding risk should send a wake-up call to the remaining majority

of international coal financiers — mainly private investors and commercial banks (Urgewald, 2021) — that investing in coal is risky. The private sector should join public financiers to stop financing coal and consider directing more capital toward renewable power generation technologies, such as solar and wind, which are less likely to be suspended and help achieve a global green energy transition.

Guiding the customary coal-financing towards solar and wind technologies creates opportunities for emerging economies to meet their electricity needs through climate-friendly energy resources. Accelerating investments in renewables in emerging economies is a top priority for building a global energy system capable of delivering several UN Sustainable Development Goals. If China and other energy investors could channel their tremendous capital, technology and engineering capacity in renewables into expanding green electricity throughout the developing world, it could dramatically accelerate the currently COVID-stagnated progress of increasing electricity access.

Secondly, environmental issues must be considered seriously in investors' decision making. The results in the first and second DPhil papers suggest that while investments could have significant environmental impacts, environmental issues may fail investments. Although investments could bring economic and social benefits, they should not be built at the cost of destroying the environment. This highlights the need for environmental impact assessments of new investments, not only to avoid or mitigate ecological losses, but also financial losses incurred due to project disruption. It also calls for strengthened environmental governance from authorities, putting in place safeguards and setting guidelines for sustainable investments.

Thirdly, economic growth and protecting the environment are not always at odds. The results in the third DPhil paper suggests that achieving economic prosperity in a sustainable manner – whether by exiting less productive firms or by adopting more efficient practices and processes – could decouple economic growth from their environmental impacts. This challenges the dominant political argument

that achieving environmental performance is at the cost of suppressing economic development. Besides, the findings on the heterogeneity in the mechanisms of private and state-owned firms' response to regulations indicate that protectionism is detrimental to enhancing competitiveness through environmental regulations.

This DPhil thesis calls for well-designed environmental policies, facilitating the shift to low carbon and resource-efficient growth path, and the need for equal enforcement of the regulation, regardless of regulated entities' bargaining power.

6.3 Limitations

Recognisably, the results in this DPhil thesis come with several limitations. The first DPhil paper only focuses on technology-specific environmental risks based on where the power projects are located. Other risks, such as operational, political and financial risks, may also affect the implementation of power projects (Yuan et al., 2019). Each power project is situated in its own unique technical, social, economical, and political context that ultimately influences investment's fate. Therefore, one needs to be cautious to interpret findings, as strict causation may not be established from this analysis.

The second DPhil paper assumes that sufficient plants can be converted to meet an emissions budget. In reality, there may be technical and economic barriers at the individual plant level, which mean our estimate of the potential to reduce stranded generation may be overestimated. Remarkably, there is much uncertainty regarding the potential and pace of future conversions. This paper presents a back-of-the-envelope estimation at the global level to lay the groundwork, and further analysis at the individual level is needed to strengthen the findings.

Due to data availability constraints, the third DPhil paper's data analysis only contains one year of fully comprehensive firm data and two years of pilot year firm data in the pre-policy period. The sample size of pilot year data is much smaller

compared to full years. Although the results are robust to dropping the pilot years, it remains a data limitation. The findings could be strengthened if other micro-level data in the pre-policy period become available and the results remain consistent.

6.4 Future research priorities

While this DPhil thesis has advanced our knowledge on interactions of the environmental and economic systems, it has also laid out several future research directions that are worth further investigation.

The role of cooperation in energy financing

Infrastructure projects, particularly large-scale ones, are frequently co-developed by several firms, involving financier, engineering contractor, equipment supplier, maintainer. Many Chinese overseas power plant projects are co-developed. Partners range from the host country's local firm to other countries' companies (rather than Chinese or host country's) to peer Chinese firms. projects involve co-financing from Chinese banks, either public funding from policy banks or private financing from commercial banks. It is worth further exploring the role of partners in energy financing and the effect of cooperation on the project's location choice and implementation status.

Unpack determinants of solar and wind project success

Solar and wind technologies have experienced lower suspension rates in the sample of Chinese overseas investments. This naturally raises the question: how is the comparison at a global scale? To accelerate investments in solar and wind power, stakeholders could learn from the success of implemented projects to understand the challenges developing new projects faces. A further study could be dedicated to exploring the factors that could contribute to solar and wind project success.

Extreme weather events and electricity supply vulnerability

The first DPhil paper has shown that coal power projects are less likely to succeed in

countries with severe losses from extreme weather events. The second DPhil paper confirms that fossil fuels power plants are under high stranding risk under climate constraints. On the other hand, more frequent extreme weather events attributable to climate change may impose significant risks to the electricity grid. For example, millions of people experienced power outages during a series of winter storms in Texas in the United States in February 2021. The interaction between climate change and the energy system is as complex as the linkage between environmental and economic systems. It is crucial to understand the potential risks imposed on electricity supply by climate change when shaping an accessible, affordable and low-carbon electricity system.

Learning from the suspension of biomass power plants

In most climate scenarios, large-scale deployment of BECCS (bioenergy combined with carbon capture and storage) is considered a requisite for attaining the stringent climate target (Gasser et al., 2015, Smith et al., 2016). Results in the second DPhil paper suggest the amount of electricity generation at risk of stranding would rise by 45 percent if bioenergy is not deployed to its full potential. Nevertheless, the deployment of bioenergy is experiencing many challenges, such as feedstock availability and economic cost (Smith et al., 2016), and facing disputes over sustainability because of risks of deforestation, concerns over food security and potential biodiversity loss (Creutzig et al., 2015, Ember, 2019). The suspension rate in biomass power plants in the sample of Chinese overseas investments is high, and this motivates further investigation on biomass power plants' suspension, given that the development of bioenergy is crucial for decarbonising the energy sector.

More evidence from environmental policies and Chinese firms' innovation

The third DPhil paper has provided evidence from a command-and-control type environmental policy in China to shed light on how regulations may affect firm competitiveness. China has implemented a significant number of environmental policies, covering diverse types of instrumental tools. The understanding of these

policies' effects is still limited. For example, to promote the market penetration of electric vehicles (EV), China launched a national-wide EV subsidy scheme in 2009 (Hao et al., 2014). China has become the world's largest EV market, with 1 million battery electric vehicles and 0.25 million plug-in hybrids sold in 2020 (Richter, 2021). Whether the subsidy enhanced the firm's competitiveness, particularly innovation capabilities, remains unclear.

Firm ownership and investment efficiency

The first DPhil paper discovers the difference in technology choice between state-owned enterprises (SOE) and private firms' overseas investment in electric power. The third DPhil paper suggests that mechanisms that drive the productivity gains from environmental regulations differ between SOE and private firms. These findings motivate the further investigation of the heterogeneity among firms with different ownership, particularly in investment decision-making and efficiency. For example, it is worth exploring global utility firms' investment behaviours in the electric power sector to shed light on investment trends.

7 Conclusions

This DPhil thesis explores three critical areas for which more research is needed to understand the interaction between environment and economy thoroughly. They are environmental risks embedded in energy investments, the role of technologies in mitigating capital loss under climate constraints, and the impact of environmental regulation on firm productivity. Each research question is addressed in one specific DPhil research paper.

Overall, this DPhil thesis has advanced the literature on environment and economy. It shows that environmental policies are needed to manage environmental issues. Otherwise, environmental risks may impose threats to economic development. If investments are not thought through (e.g. heavy investments in carbon-intensive sectors), stringent environmental policies may put assets at risk. Nevertheless, environmental policies can be a practical tool to achieve simultaneously productivity gains and environmental performance.

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