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Sustainability of Overlapped Emission Trading and Command-And-Control CO_2 Regulation for Korean Coal Power Production: A DEA-Based Cost-Benefit Analysis

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Regulatory policies are indispensable to efficiently curbing anthropogenic CO_2 emissions and sustainably producing emission-intensive goods. Though previous modelling practice has studied the cost and benefit of different regulatory policies, such as command-and-control (CAC) and emission trading scheme (ETS), little is known about that for overlapped emission regulation policies. Here, we built up a Data Envelopment Analysis model to study the losses and gains from the overlapped implementation of CAC and ETS for Korean coal-fired power plants during 2011–2015. We showed that the initial phase of CAC in 2012 caused a sudden loss in power plants' output, but that the loss was gradually eliminated in 2013 and 2014. Upon promulgation in 2015, ETS is expected to increase only 0.990% of output compared to CAC, yet it largely failed to deliver the potential benefit in its first year. The overlapped implementation of CAC and ETS contributes to a small share (5.567%) of the unrealized benefit. Nonetheless, we showed that implementing CAC and ETS in parallel tends to disproportionately affect less efficient power plants by restricting their strategies to meet regulatory measures. Therefore, we suggest that the integration of CAC and ETS can be a transitory measure as ETS provides only marginal welfare benefits, but ETS must be fully adopted and strengthened in the near future to economically and equitably mitigate CO_2 emissions.

Keywords: cost-benefit analysis, emission regulation, command and control, emission trading, data envelopment analysis, activity analysis, CO_2 emission, distributional effect

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INTRODUCTION

CO_2 emissions regulation spurs green technology innovation, which subsequently induces economy-wide energy savings and productivity growth (Magazzino et al., 2022; Shahzad et al., 2022; Song et al., 2022). For emissions regulation, command-and-control (CAC) system has been a predominant instrument pursued by industrial economies in the 1980s, where firms are required to limit their emissions below an authorized maximum emission level (Opschoor and Vos, 1989). Alternatively, for CO_2 emissions regulation, carbon pricing and emission trading (ET) have prevailed ever since its advent in the Kyoto Protocol, due to their cost-effectiveness. Theoretically, any emission controls, no matter CAC or ET, will pose economic costs to regulated entities as they will have to pay for additional efforts to regulate emissions. However, ET creates tradable emission permits and an emission trading market to allow emission-controlling entities to sell or buy emission permits such that they can comply with a centrally authorized emission controlling target. It is capable of equalizing marginal abatement of anthropogenic CO_2 emissions across all carbon emitters, so that lower overall cost of emission regulation. The mechanism contrasts with CAC policy instruments, where administrative measures are directly posed to emission-heavy entities to mandate their emission upper bound, causing staggered cost to different firms (Tietenberg and Tietenberg, 1985; Stavins, 2003).

A number of studies have attempted to estimate costs of CAC emission regulations and savings from alternative ET programs. For example, Yang et al. (2021) studied the cost of CAC regulation on Chinese transport sector, and found that the regulation has cost China around 2000 billion RMB during 2013–2017. Zhang and Xie (2015) found that CAC regulation on Chinese information technology industry could potentially cost China up to 29.3 billion RMB. Some have focused on the savings of ET. Brännlund et al. (1998) estimated the potential gains of the Swiss pulp industry from oxygen-demanding substance emission trading in 1989 to be 6%. Färe et al. (2013) studied potential gains from air pollutant ET in the US and found that spatial trading of air pollutants potentially increased the profits of all coal-fired power plants by 3–6%. Later works concentrated on CO_2 emissions, the primary cause of global warming and climate change. Wang et al. (2016) estimated potential CO_2 emission abatement cost savings in the context of the Chinese national emission trading market. They suggested that implementing national CO_2 emission trading would save 10.78% of GDP losses from emission mitigation in China over 2006–2010. Subsequently, Xian et al. (2019) estimated that the abatement cost savings with ET in China would be 13% on average during 2011–2015.

Existing papers have mostly pitted ET against CAC regulations, frequently claiming ET is more or less superior to CAC regulations. However, they have rarely paid enough attention to the interaction of emission trading and command-and-control. Yet, there are a number of examples being displayed in real-world policy experiments. For example, Korea has been implementing its CAC-style emission controlling strategy (Target Management System, TMS) in parallel to ET (Park and Hong, 2014). In Europe, a number of direct emission regulation policies are also simultaneously implemented despite the existence of the EU ETS (Braathen, 2011). A variety of considerations are in place when mixing ET and CAC. For example, ET may lead to carbon leakage in some specific sectors (Zhou et al., 2020), transaction costs involved in ET may be cost-prohibitive for small emitters (JaraitĖ et al., 2011; Park and Lee, 2020), and the like. Only few have studied the interaction of different emission regulation mechanism. For instance, Goulder and Stavins (2010) conducted an ex-ante explorative analysis of the interaction between state policies and federal policies, where impact of the interaction was found depends on specific designs, such as coverage, stringency, of policies at different levels. So far, there remains insufficient empirical studies on the cost and benefit of mixing the two regulatory mechanisms, raising concerns over the inefficiency of such a hybrid policy setting (Stavins, 2021).

Furthermore, the distributional effect of emission regulations is of tremendous concern. As put forward by Eskeland and Jimenez (1992), distributional effect analysis could inform government alternative policies that may gain the largest political support. Commonly, CAC policy instruments, which set a common emission standard for technologically heterogeneous firms, are often deemed inequitable as they restrain factor mobility and pose a disproportionally high financial burden on technologically unprepared entities (Stavins, 2003). In contrast, ET equalizes marginal abatement costs—the additional costs incurred by one incremental emission reduction—through emission permit transactions (Montgomery, 1972). In this way, firms with lower abatement costs will take action first and render emission allowances surplus. Firms with higher abatement costs will buy additional allowances from those low-abatement-cost firms. Thus, the systemic cost of all market participants is at its lowest level. The argument underlies the thriving emission trading market globally. However, empirical evidence shows that those regulations tend to be regressive (Ohlendorf et al., 2020). Big companies with an increasing return of scale tend to participate in the emission trading market more proactively and enjoy much more benefit from ET compared with their smaller counterparts, thus rendering exacerbated inequality among participating entities compared with CAC (JaraitĖ et al., 2011). This aligns with the empirical evidence by (Heindl, 2012), who found an scale effect on transaction costs among firms participating in EU ETS. Therefore, the policy practice of mixing ET and CAC steps in. Beyond this, there has been a lack of an appropriate model quantifying the distributional effect of mixing CAC and ET.

This paper contributes to existing literature by building an operational model to analyze the cost and benefit of CAC and ET, and that in the overlapped scenario. This paper is a step forward from our previous paper (Yang et al., 2021). Here, we can not only analyze the costs and benefits of CAC, but also that of ET, and the mix of ET and CAC. Another strength of our model is that it is capable of conducting cost-benefit analysis at firm or utility levels, in contrast to previous models that adopt macroeconomic data (Choi et al., 2017). It can therefore provide a more granular cost-benefit estimate at a sub-sector level. We take emission regulation of coal-fired power plants in South Korea (hereafter, Korea) as a case study and examine the costs and benefits of the Korean target management system (TMS), a CAC emission regulation mechanism, and the Korean ETS, a cap-and-trade system, as well as the distributional effect of implementing ET and CAC in parallel.

Our paper is organized as follows: theoretical models to quantify the costs and benefits of different emission regulations are presented in **Section 2**; the Korean policy background of emission regulation and data for calculation are presented in **Section 3**; empirical results are presented in **Section 4**; and discussions on potential extensions and limitations of our theoretical models and empirical findings are presented in **Section 5**.

THEORETICAL MODEL

We start by constructing a homogenous environmental production technology set, T. We denote exogenous inputs, $x = (x_1, x_2, \ldots, x_i) \in \mathfrak{R}^i$, desirable outputs, $y = (y_1, y_2, \ldots, y_m) \in \mathfrak{R}^m$, and undesirable outputs, $b = (b_1, b_2, \ldots, b_s) \in \mathfrak{R}^s$. The production possibility set, P, for environmental production technology T for coal-fired power plants can be expressed as:

$$P(x; y, b) = \{(y,b): x\ can\ produce\ (y, b)\} \tag{1}$$

where, in the Data Envelopment Analysis (DEA) formulation, $P(\cdot)$ satisfies the standard properties of a technology (monotonicity, convexity, and minimal extrapolation). Among these properties, monotonicity implies that outputs always increase, or at least not decrease, with the increase of inputs; convexity ensures that all the observations in the production possibility set P construct a convex set; and, minimal extrapolation indicates that $P(x; y, b)$ has the minimum set that constitutes all the observations. In addition, inactivity is always possible. Finite inputs can only produce finite outputs.

No Emission Regulation

Based on the DEA formulation of environmental production technology, we can first simulate maximum potential economic output if the technical inefficiency of each decision-making unit (DMU) is eliminated, considering undesirable output as freely disposable, which is a scenario for unregulated environmental technology (Färe and Grosskopf, 1983; Wang and Feng, 2014; Färe et al., 2016).

$$\begin{aligned} &\max \tilde{y}_n \\ \text{s.t.} &\sum_{n=1}^{N} \lambda_n x_{in} \le x_{in} \quad i = 1, \ldots, I \\ &\sum_{n=1}^{N} \lambda_n y_n \ge \tilde{y}_n \\ &\lambda_n \ge 0,\ n = 1, \ldots, N \end{aligned} \tag{2}$$

DMUs can freely dispose of their undesirable outputs under the current model specification. The assumption of non-negativity for the intensity variables, λ_n, restricts the model to being constant-return-to-scale. It also serves to construct a production frontier through a convex combination of the observed production factors (Brännlund et al., 1998). The sum-up of the optimized desirable output, $\tilde{y}_n^*$, of all the DMUs constitutes the maximum potential desirable output that the environmental production technology can attain (eliminating technical inefficiency) while no emission regulations are posed.

Command-And-Control

In addition, if we assume null-jointness and weak disposability for desirable and undesirable outputs:

1) For $(x;\ y,\ b) \in P$ and $0 \le \theta \le 1$, $(x;\ \theta y,\ \theta b) \in P$
2) For $(x;\ y,\ b) \in P$ and $b = 0$, $y = 0$

Then, we can simulate production activities under emission regulation with command-and-control policies (Färe et al., 2013; Zhang and Zhang, 2018; Yang et al., 2021; Zhao et al., 2022). Considering N decision making units (DMUs) with I inputs, S undesirable outputs, and one desirable output. For each DMU n, the maximum output under command-and-control policies can be expressed as:

$$\begin{aligned} &\max \tilde{y}_n \\ &\sum_{n=1}^{N} \lambda_n x_{in} \le x_{in} \quad i = 1, \ldots, I \\ &\sum_{n=1}^{N} \lambda_n y_n \ge \tilde{y}_n \\ &\sum_{n=1}^{N} \lambda_n b_{sn} = b_{sn} \quad s = 1, \ldots, S \\ &\lambda_n \ge 0, n = 1, \ldots, N \end{aligned} \tag{3}$$

The equality constraints, $\sum_{n=1}^{N} \lambda_n b_{sn} = b_{sn}$, on undesirable outputs impose weak disposability.

Under the current specification, each DMU can maximize its desirable output, y_n, to an optimal level, $\tilde{y}_n^*$, while maintaining its observed level of undesirable output, b_{sn}. Hence, the model simulates the output losses due to a binding environmental regulation. Here, the desirable output is electricity generation, and CO_2 emission is the sole undesirable output. But the model makes it easy to extend to a case of multiple undesirable outputs. Setting the desirable output as electricity generation excludes monetary value from the specification. Therefore, we do not need to account for the effect of price fluctuation on emission controlling behaviors.

Emission Trading

We can further extend the operational model to simulate emission trading. Consider a central regulator that inspects the emissions of all the participating entities. The total emissions from all the entities must comply with the upper limit set by the regulator, while those entities can freely trade their emissions at no transactional cost. We formulate a centralized DEA model as follows to estimate the cost and benefit of participating DMUs in the emission trading scenario:

$$\begin{aligned} &\max \sum_{n=1}^{N} \tilde{y}_n \\ &s.\ t. \\ For\ DMU\ 1: &\sum_{n=1}^{N} \lambda_{1n} x_{in} \le x_{i1} \quad i = 1, \ldots, I \\ &\sum_{n=1}^{N} \lambda_{1n} y_n \ge \tilde{y}_1 \\ &\sum_{n=1}^{N} \lambda_{1n} b_{sn} = b_{s1} \quad s = 1, \ldots, L \\ &\sum_{n=1}^{N} \lambda_{1n} b_{sn} = \tilde{b}_{s1} \quad s = L+1, \ldots, S \\ &\ldots \end{aligned}$$

$$
\begin{aligned}
&\text{For DMU } N: \sum_{n=1}^{N} \lambda_{Nn} x_{in} \le x_{iN} \quad i = 1, \ldots, I \\
&\sum_{n=1}^{N} \lambda_{Nn} y_n \ge \tilde{y}_N \\
&\sum_{n=1}^{N} \lambda_{Nn} b_{sn} = b_{sN} \quad s = 1, \ldots, L \\
&\sum_{n=1}^{N} \lambda_{Nn} b_{sn} = \tilde{b}_{sN} \quad s = L+1, \ldots, S \\
&\lambda_{nn} \ge 0, \; n = 1, \ldots, N \\
&\textstyle\sum_{n=1}^{N} \tilde{b}_{sn} \le B_s, \; s = L+1, \ldots, S
\end{aligned} \tag{4}
$$

In **Equation 4**, x_{in}, y_n, and b_{sn} are observed values of inputs, desirable outputs, and undesirable outputs, respectively. Some parts of b_{sn} for $s = L+1, \ldots, S$ are traded emissions among the participating DMUs, while no emission trading occurs over the other undesirable output (b_{sn} for $s = 1, \ldots, L$). Optimization occurs over λ_{nn}, $\tilde{b}_{sN}$, and $\tilde{y}_n$, to maximize the total desirable output of all the participating DMUs, $\sum_{n=1}^{N} \tilde{y}_n$. Inputs, x_{in}, are taken as fixed. $\tilde{b}_{sN}$ represents the emissions of each power plant after trading, while $\tilde{y}_n$ represents the desirable output. $B_s = \sum_{n=1}^{N} b_{sn}$ represents the total emission trading permits that exist in the trading market as regulated by the central regulator. This formulates a traditional cap-and-trade emission trading scheme.

It should be noted that **Equation 4** models the maximum gains that any cap-and-trade programs can reap, *i.e.*, a perfect market where trading is allowed without any barriers or transaction costs. It mimics the condition in which an overall emission cap is enforced by the central regulator while no cap is set for market participants. This market design can, therefore, be interpreted as emission trading with auctioning in the market. Efficient bargaining is achieved under the current model specification, where each entity can attain an efficient level of emission allowance (Cramton and Kerr, 2002).

Integrating Emission Trading and Command-And-Control

A common practice in emission trading or cap-and-trade programs is the free allocation of a portion of emission allowances to individual trading entities in the emission trading market, which existed in the initial phase of ETS to lower political barriers for its implementation (Helm, 2010). However, to regulate the overall emission cap, emission trading is often implemented in parallel to command-and-control regulation. In this scenario, binding caps are often set for each regulating entity, usually at a ratio of its historical emissions. Based on the emission trading model, we can further extend the integration of emission trading and command-and-control as follows:

$$
\begin{aligned}
&\max \sum_{n=1}^{N} \tilde{y}_n \\
&s.\ t. \\
&\text{For DMU } 1: \sum_{n=1}^{N} \lambda_{1n} x_{in} \le x_{i1} \quad i = 1, \ldots, I \\
&\sum_{n=1}^{N} \lambda_{1n} y_n \ge \tilde{y}_1 \\
&\sum_{n=1}^{N} \lambda_{1n} b_{sn} = b_{s1} \quad s = 1, \ldots, L \\
&\sum_{n=1}^{N} \lambda_{1n} b_{sn} = \tilde{b}_{s1} \quad s = L+1, \ldots, S \\
&\tilde{b}_{s1} \le \tilde{b}'_{s1} \\
&\ldots \\
&\text{For DMU } N: \sum_{n=1}^{N} \lambda_{Nn} x_{in} \le x_{iN} \quad i = 1, \ldots, I \\
&\sum_{n=1}^{N} \lambda_{Nn} y_n \ge \tilde{y}_N \\
&\sum_{n=1}^{N} \lambda_{Nn} b_{sn} = b_{sN} \quad s = 1, \ldots, L \\
&\sum_{n=1}^{N} \lambda_{Nn} b_{sn} = \tilde{b}_{sN} \quad s = L+1, \ldots, S \\
&\tilde{b}_{sN} \le \tilde{b}'_{sN} \\
&\lambda_{nn} \ge 0, \; n = 1, \ldots, N \\
&\textstyle\sum_{n=1}^{N} \tilde{b}_{sn} \le B_s, \; s = L+1, \ldots, S
\end{aligned} \tag{5}
$$

The model features a variation of the emission trading model by Färe et al. (2013). First, a common practice is that market regulators may retain a certain percentage of the total emission allowance for the purpose of market stabilization (Park and Hong, 2014). Hence, the total amount of cap emissions, $\sum_{n=1}^{N} \widetilde{b'_{sn}}$, could be different from the total tradeable emissions, B_s, which simulates the condition when the trading market regulator retains some of the permits from trading. The effect of the total amount of cap emissions on potential gains from emission allocation can also be investigated. Further, in **Equation 5**, we introduce $\widetilde{b'_{sn}}$ as an exogenous variable that determines the additional command-and-control emission cap of each DMU through the constraint, $\tilde{b}_{sN} \le \widetilde{b'_{sN}}$. This constraint mimics the command-and-control constraint, where binding caps are assigned to individual participating entities. Overall, the current model setting simulates the maximum potential gains and the minimum cost when emission trading and command-and-control policy are simultaneously implemented.

Again, due to the fact that the current work considered only CO_2 emissions, no additional constraints were posed to DMUs on other undesirable outputs. But the model can be easily extended to a case where data for multiple undesirable outputs is available. Furthermore, the model can also be easily generalized to an inter-temporal and inter-spatial model, similar to the case of Färe et al. (2013), which simulates an emission trading scheme where borrowing and banking are allowed (Boemare and Quirion, 2002). We leave it to future research.

POLICY BACKGROUND AND DATA

Policy Background

Korea announced its national CO_2 emission abatement target in 2008, which aimed at cutting 30% of CO_2 emissions by 2020 compared to a business-as-usual scenario. Therefore, in 2010, a presidential decree ratified the target and established the TMS, a command-and-control policy setting that was initiated in 2011 and formally began to operate in 2012, which directly enforced firms' compliance with emission reduction targets (Park and Hong, 2014). Major energy producers were all included in TMS. Following the enactment of TMS, there was the promulgation of the *Act on the Allocation and Trading of Greenhouse Gas Emission Allowances* (ETS Act) in 2012, which began its trading in 2015, 3 years after TMS. The two regulatory schemes operate in parallel but cover mostly different categories of entities. The ETS act covers the major emitters of CO_2, while TMS complements it with smaller but still relatively large (or medium-sized) emitters as well as public organizations that cannot be easily workable in ETS. Nonetheless, there are still chances for any DMU to overlap in this parallel system. For example, the company may participate in ETS as a major player, while one of the facilities of the company may be located under TMS. From the managerial perspective, therefore, strategic consideration may happen in these overlapping cases, and all the other companies may consider this kind of by-passing way between ETS and TMS, resulting in the integrated, yet parallel governance of the emission management policies. The Korean government may consider the TMS as a complementary mechanism to support higher abatement of ETS, but in reality, it allows the participating companies in ETS to enter the TMS. Nonetheless, TMS has a different condition as an additional target of energy consumption, which is not in the ETS, resulting in a very complicated, yet complex alleyway between these two measures.

The first commitment period of the ETS system is designated to begin on 1 January 2015 (Oh et al., 2016). For the initial allowance allocation, 100% of the previous year's emissions were retained as the total allowance, with an additional 3% of the total allowances being set aside as a reserve for market stabilization. For the second allowance period between 2018 and 2020, 97% of the allowance was freely allocated to each entity, with a 3% abatement target to be tradable in the ETS market. ETS entered the third stage in 2021, with 90% of the free allowance and a 10% abatement target to be tradable in the market. In the year 2021, the initial year of the third stage, the participating entities in ETS will be 684, with a total allowance of 2.9 billion CO_2. Among those abatement targets, 18.2 million tons are allocated as tradable allowances, implying that 10.8 million tons, or 62.8% of total allowances, are reserved for the government. As a parallel measure, the entities under TMS comprise 403 members in the private sector and 837 entities in the public sector in the year 2021.

Data

From the year 2011 till the year 2015, we collected and used only data for coal-fired power plants in Korea. This data fits the homogeneity assumption of environmental production technology. Therefore, there is no need to consider heterogeneity in the modelling of the production technology. The sample period includes the first year of no regulation, the three following consecutive years of command and control regulation, and the latest year of emission trading. In total, 258 observations were collected. The observations consist of two inputs (unit capacity in kW and energy usage in kcal), one desirable output (net electricity generation in MWh), and one undesirable output (CO_2 emissions in tons). The data was obtained from Statistics of Electric Power in Korea (2016). CO_2 emissions were calculated using IPCC emission factors, provided by Zhang and Choi (2013). **Table 1** reports descriptive statistics for inputs and outputs in the study period.

RESULTS

Overall Effect of Different Emission Regulation Strategies

Figure 1 shows the cumulative desirable output—electricity generation—of all the coal fired power plants in each year under different environmental regulation scenarios. Depending on the regulatory strategy of the scenarios, the implications of the results vary.

If no environmental regulations were posed (Korea in 2011), **Equation 2** calculates the maximum potential output each coal power plant can attain under current input constraints. Thus, its difference with the base case represents the maximum electricity production of Korean coal-fired power plants if technical inefficiency is eliminated. The difference between **Equation 3** and **2** is the opportunity cost each power plant may bear to *implement* CAC regulation on their emissions and restrict them to the observed level. The difference between **equation 4** and **2** is the potential costs and benefits an emission trading system brings about for these power plants. While the difference between **Equation 5** and **2** is the potential cost of additional CAC regulation combined with the ETS.

If environmental regulation has already been forced on those coal-fired power plants (Korea from 2012 to 2014), the difference between **Equation 3** and **2** implies potential gains from *rescinding* current environmental regulations. The cost and benefit of rescinding existing regulations can be substantially different from implementing a new regulation (Evans et al., 2021). When rescinding an existing regulation, compliance costs have already sunk. No immediate benefit will be paid to entities that comply. Besides, upon the elimination of a regulation, market conditions and technologies have already changed. Previously profitable operational conditions may not be viable at the time of rescinding the regulation. The difference between **Equation 4** and **3** is the maximum potential economic benefit the whole trading system can reap by mobilizing productive factors in these environmentally regulated power plants. And the difference between **Equation 5** and **3** is the gains the trading system as a

TABLE 1 | Descriptive statistics of data for Korean coal-fired power plants from 2011 to 2015.

Year (status)	Variables	Unit	Mean	Std. Dev	Minimum	Maximum
2011 (no regulation)	*Capacity*	10^9 kW	481051.0	154589.2	870000.0	125000.0
	Electricity generation	10^9 MWh	3956772.1	1318381.7	7307170.0	968094.0
	Energy	10^3 kcal	8779561.8	2786522.9	15867278.0	2456240.0
	CO_2 emissions	10^6 t CO_2-eq	2980308.7	922583.7	5296805.8	883040.5
2012 (TMS)	*Capacity*	10^9 kW	481051.0	154589.2	870000.0	125000.0
	Electricity generation	10^9 MWh	3908425.8	1319859.8	7244531.0	1009662.0
	Energy	10^3 kcal	8502303.1	2829268.9	15487730.0	2158059.0
	CO_2 emissions	10^6 t CO_2-eq	2909908.4	939886.6	5263632.4	915132.3
2013 (TMS)	*Capacity*	10^9 kW	481051.0	154589.2	870000.0	125000.0
	Electricity generation	10^9 MWh	3943506.5	1326110.2	7324679.0	967109.0
	Energy	10^3 kcal	8776091.0	2765974.5	15742629.0	2442416.0
	CO_2 emissions	10^6 t CO_2-eq	2906220.6	897309.2	5089644.3	876401.8
2014 (TMS)	*Capacity*	10^9 kW	495728.3	169049.1	870000.0	125000.0
	Electricity generation	10^9 MWh	3844630.1	1306256.4	6902839.0	931289.0
	Energy	10^3 kcal	8305163.5	3177250.0	14706172.0	988740.0
	CO_2 emissions	10^6 t CO_2-eq	2718142.3	1020846.6	4749363.6	310674.2
2015 (ETS)	*Capacity*	10^9 kW	498942.3	169055.4	870000.0	125000.0
	Electricity generation	10^9 MWh	3927200.4	1389797.7	7269962.0	969147.0
	Energy	10^3 kcal	8577507.0	3144200.7	15667311.0	989163.0
	CO_2 emissions	10^6 t CO_2-eq	2805694.9	1033091.6	5125737.5	300013.5

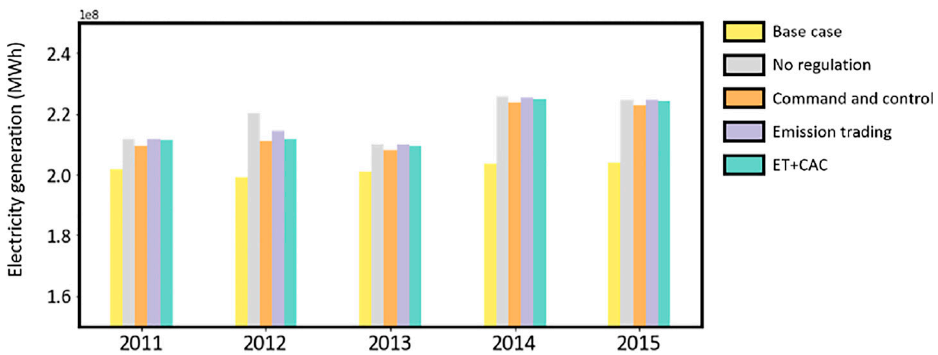

FIGURE 1 | Total electricity generation of Korean coal-fired power plants under different emission regulation scenarios. The *base case* scenario represents real electricity generation for power plants; the *no regulation* scenario stands for eliminating technical efficiency for power plants without emission regulation; the *command and control* scenario represents a scenario where technical efficiency is eliminated but emissions must be restricted to the current level; the *emission trading* scenario stands for the case where overall emissions of all power plants are restricted to the current level but emissions are tradable; and *ET + CAC* refers to the scenario where emission trading and command-and-control are implemented in parallel.

whole can reap with additional binding on the emission cap of each power plant.

Furthermore, if an ETS has already been implemented (Korea in 2015), the difference between **Equation 4** and **3** explains the *foregone* benefits of emission trading. The foregone benefits, different from the potential benefits in a no emission trading setting, explain potentially existing transactional barriers, e.g., transaction costs rendered by collecting information, bargaining and deciding, and monitoring and enforcement, that prohibit a competitive market from exerting its full power to maximize factor allocation efficiency (Stavins, 1995). The difference between **Equation 5** and **4** represents potential costs—the emission abatement costs and market efficiency losses—with the overlap of command-and-control regulation and emission trading.

As shown in **Figure 1**, the potential output increase from eliminating technical inefficiency in power plant operations is on average 7.434% (no regulation case). Upon implementing command-and-control CO_2 emission regulation, an increase in technical inefficiency can be seen in 2012 (9.548%), which is largely due to the curtailment of existing coal-fired power plants to meet the CO_2 emission target (201.8 TWh of electricity generation in 2011 *versus* 199.3 TWh of electricity generation in 2012). While the sudden increase (9.706%) of technical inefficiency from 2013 to 2014 is due to the installation of a new megawatt-scale power generation unit.

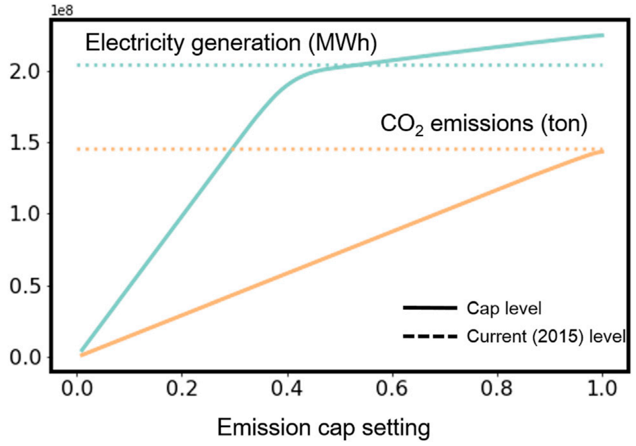

FIGURE 2 | Electricity generation and CO_2 emissions under different levels (1–100% of 2015 power plant emission level) of emission cap settings with mixed emission trading (ET) and command-and-control (CAC) policy. The cyan solid line indicates the maximum electricity generation under different CAC cap settings. The orange solid line indicates CO_2 emissions at the level of the CAC cap setting. The dot lines indicate the current levels (in 2015) of electricity generation and CO_2 emissions.

We then compare the difference between the *no regulation* scenario and with the *command-and-control* scenario. In 2011, no CO_2 emission regulations were posed to Korean coal-fired power plants, and thus the difference between these two models represents the potential cost of implementing emission regulations. The potential cost in 2011 is estimated to be only 0.969% at the time of implementing CAC (TMS). However, as the TMS regulation was implemented in 2012, the potential benefit of rescinding the policy was raised to 4.386% of total electricity generation, revealing that the emission regulation policy presents a major hurdle to power plants' production activities. However, the potential benefit of rolling back emission regulation shrank to 0.757% in 2014, suggesting that power plants rapidly reflected the command-and-control regulation and substantially improved their environmental technology to comply with the emission standard, such that the negative effect of emission regulation was alleviated. Furthermore, the potential benefits of rescinding emission regulation are the lowest in 2015, suggesting that emission trading has effectively lowered the overall cost of compliance.

In addition, we can see that emission trading has the potential to increase output by 0.990% on average in those power plants during the TMS regulation period (2012–2014). However, most of the promised benefits have not materialized. In 2015, the ETS was implemented in Korea. Yet, the potential output under the emission-trading model (emission trading with no trading friction) is still 0.861% higher than the command-and-control model. The persistent gap between the command-and-control model and the emission-trading model suggests that little or no efficiency increase is achieved through the first-year of emission trading at least. In contrast, over 85% of the promised efficiency increase by emission trading has not been realized. Färe et al. (2013) suggest that the gap—the foregone benefit in the emission trading scheme—represents the upper limit of transaction costs involved in emission trading, which impedes participating entities from sufficient trades in the allowance market to reach an efficient status. The results indicate that the transaction cost in the Korean emission trading market is on a non-negligible scale. Efforts must be made to alleviate this huge cost so it boosts the potential of emission trading more effectively.

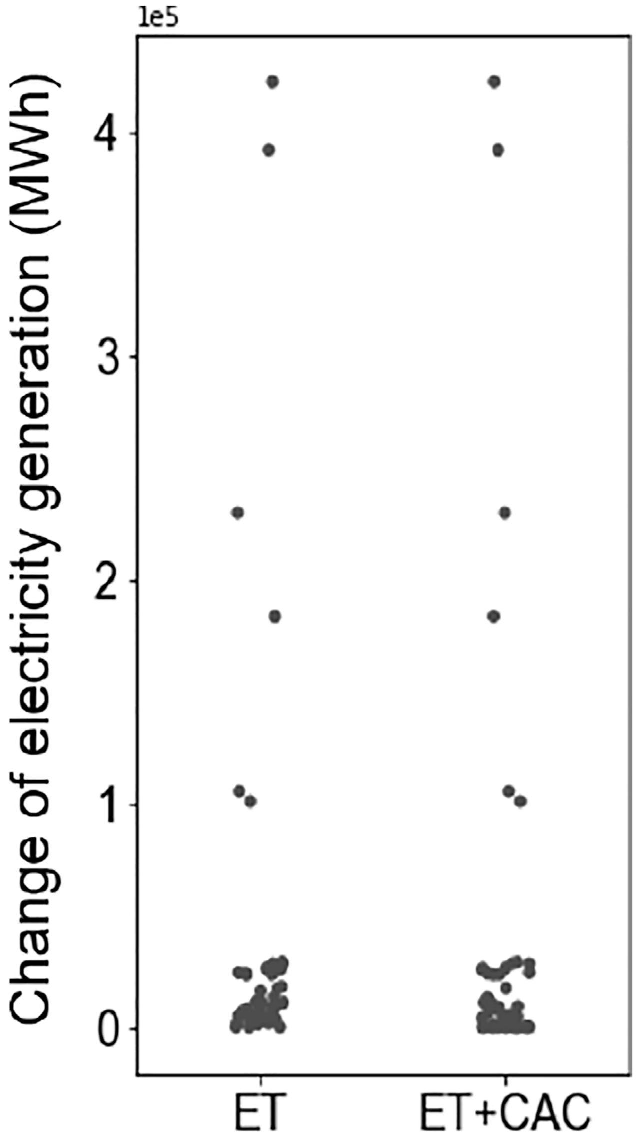

FIGURE 3 | Changes of each power plant in electricity generation under the emission trading scenario and the mixed emission trading and command-and-control (*ET* + *CAC*) scenario.

Furthermore, the integration of command-and-control and emission trading requires a cap that restrains plants' emissions at their observed levels. In this way, we can distinguish the effect of binding emission caps on the foregone emission trading benefit, which can be accounted for as a regulatory cost when integrating command-and-control and emission trading. Our results show that electricity generation under the integrated model in 2015 was about 224.5 TWh, compared to 224.6 TWh for the emission trading model and 222.7 TWh for the command-and-control model. Command-and-control regulation deviates emission trading from its efficient level, accounting for only 5.567% of the total unrealized gains under emission trading in 2015.

Another concern is marginal abatement cost under different emission regulation schemes. The marginal abatement cost could dictate the policy-making over emission mitigation (Wu and Ma, 2018). Besides, marginal abatement cost curves is a widely accepted tool to characterize the overall cost if a certain level of abatement is to be achieved (Kesicki and Strachan, 2011). **Figure 2** presents the marginal abatement cost curve—abatement cost denominated by electricity generation losses under different levels of binding emission cap setting—for Korean coal power. Interestingly, electricity output reduction does not linearly respond to a tighter emission cap, indicating a diminishing rate of increase in the marginal abatement costs at a certain point. The marginal cost of reducing the first 1% of emissions (cap from 100 to 99%) under the emission trading and cap mechanism is 0.098% of electricity generation decreased; while the last 1% of emissions reduction (cap from 2 to 1%) costs as much as 2.18% of electricity generation. If policymakers were to regulate emissions levels with the current level (the 2015 level) of electricity generation (minimizing undesirable output while keeping the current desirable output), an emission cap and trading scheme could cut about 46% of CO_2 emissions.

Distributional Effect of Different Emission Regulation Strategies

This distributional effect of an emission regulation policy is also of paramount importance to policy makers. A progressive emission regulation policy would prevail and is politically more tenable, since it delivers the good to the larger (Eskeland and Jimenez, 1992). **Figure 3** presents the distribution of output gains (or losses) by power plants from emission trading. Detailed results are shown in Appendix **Supplementary Table A1**. In the *emission trading* scenario, Korean coal-fired power plants could, in total, increase electricity output by 1.934 TWh of potential electricity output without increasing CO_2 emissions. All the participating units will at least be better off in the *emission trading* scenario. Five power plants reap a dominant share of the total gains. They will increase their electricity output by 1.333 TWh in the *emission trading* scenario compared with the *command-and-control* scenario, equivalent to 68.9% of the total benefit from ET.

Upon integrating CAC with ET, those that reap the dominant gains of ET tend not to be affected by additional CAC. But there are another 15 power plants whose welfare will be disproportionately affected. In total, these 15 power plants generated 71.95 TWh of electricity in 2015 (equivalent to 38.9% of total electricity generation). In the *emission trading* scenario, their total gains are 0.108 TWh, which is barely 8.10% of the total gains from emission trading. In the CAC + ET scenario, their potential gains are reduced to zero. This is due to the fact that the CAC restricts their strategies to response to emission mitigation compliance. In the *emission trading* scenario, they may buy residual emission allowances left by those efficient producers so that they can expand their production capacity. Under the *ET + CAC* scenario, fewer emission allowances will be available in the market due to the restriction of CAC. Those efficient producers will retain their additional un-used allowances. Finally, those less efficient producers acquire a smaller number of allowances, thus bearing the cost of CAC. Therefore, the implementation of ET + CAC may result in disproportionately high costs for a part of the market due to the invalidation of abatement cost equalization by CAC.

DISCUSSIONS AND POLICY IMPLICATIONS

Large-scale rollbacks of US environmental regulation by the Trump administration (Popovich et al., 2020) in late 2020 raised concerns over the potential costs and benefits of the back-and-forth of environmental regulation around the globe. In this paper, we built a framework to retrospectively measure the costs and benefits of different emission regulation mechanisms, including CAC emission regulation, ET, and the integration of CAC and ET. In this framework, we are able to quantify the potential costs and benefits of not only implementing but also rescinding an emission regulation scheme. By operationalizing coal power plants in the DEA model, we can further analyze the distributional effect of those emission regulation schemes. Using CO_2 emission regulation in Korean coal-fired power plants as a case study, our empirical findings are as follows.

First, CAC emission regulation in the initial phase caused a heavy loss to Korean coal power plants, leading to a substantial technical efficiency drop. However, along with the enforcement of CAC during 2012–2014, power plants were able to advance their production technologies and operate at a higher-efficiency status while complying with the emission regulation. Therefore, the cost of CAC emission regulation diminished over time. We concluded that the benefit of rescinding the existing emission regulation is trivial. The Korean government should not consider following US practice and rolling back current regulations (the TMS system). Instead, policymakers may advance emission regulations based on the TMS to boost its benefits. An ongoing practice is moving those participating entities into the ETS group.

Second, we found that the potential gains from emission trading in Korean coal-power plants were quite marginal—only a 0.990% of output increase in the ET scenario. This is much lower than in the United States (Färe et al., 2013), China (Xian et al., 2019), and Switzerland (Brännlund et al., 1998). Furthermore, although ETS has been implemented in Korea since 2015, it has not brought about the promised efficiency increase. One critical reason is that institutional designs impede trade in emission allowance trading market (Stavins, 1995). In particular, we examined whether integrating ET with CAC, as a currently existing policy practice by Korea government, would cause substantial output losses. We show that implementing ET and CAC in parallel contributes to only 5.567% of the total electricity output loss. Hence, we suggest that integrating TMS and ETS in Korea is advisable due to its negligible efficiency loss. Lessons learned from the simultaneous implementation of EU ETS and direct regulation in Europe could be used to improve the integration of Korean TMS and ETS (Boemare and Quirion, 2002).

Third, though ET leads to a redistribution of welfare among these power plants, our results show that all the entities in the emission trading market will get better off. Nonetheless, efficient entities will reap many more benefits from emission trading schemes because of their high allocative efficiency. Proper market design must also take into account the distributional effect of emission trading. We also show that ET + CAC does not provide more benefit to inefficient power plants. Instead, CAC will disproportionately affect those low-efficiency power plants, as it restricts the strategy of those power plants to a response to the emission mitigation mandate. Therefore, we suggest Korea gradually abandon CAC and transit to a complete ETS for the purpose of not only economically reducing emissions but also providing equitable opportunities to those inefficient entities to benefit from emission trading.

A limitation of our model is that it does not fully cover participating entities in the Korean TMS and ETS emission regulation schemes. The first-year (2015) implementation of the ETS covered around 573 Mt of CO_2 emissions (Oh et al., 2016), yet our current data covered only coal-fired power plants with 149 Mt of CO_2 emissions. One step forward from our current model is to include both coal-fired power plants and non-coal-fired (such as renewable energy) power plants in emission trading. However, the modelling framework must take into consideration the technological heterogeneity of different types of power plants. To do these, the meta-frontier centralized data envelopment analysis model may be considered (Choi et al., 2020a; Choi et al., 2020b; Li and Wei, 2021).

Some elements in CO_2 emission regulation policies have been neglected in the modelling, one of which is transaction cost (Färe et al., 2013). The existence of transaction costs shifts the equilibrium of emission trading, which could explain some of the unrealized gains in emission trading. The significance of transaction cost in determining the effectiveness of emission trading schemes for Korean power plants requires further research (McCann et al., 2005).

Besides, our current study may be subject to data scarcity of other production factors. Electricity generation in coal-fired power plants generates an array of harmful pollutants simultaneously. Including constraints on other emissions has mixed effects on final outcomes (Färe et al., 2014; Ma and Hailu, 2016). To avoid the balloon effect, a more detailed study should incorporate all kinds of emissions as undesirable outputs in modelling practice. Labor is also not included due to a lack of data. It is desirable to include labor data so as to study the effect of labor mobility on emission trading behaviors of power plants (Färe et al., 2013).

DATA AVAILABILITY STATEMENT

Interested readers may retrieve codes and data through FY's Github repository at https://github.com/panday1995/2021_Korean_Emission_trading.

AUTHOR CONTRIBUTIONS

FY, HL and YC contributed to conception and design of the study. FY and HL organized the database. FY performed the statistical analysis. FY wrote the first draft of the manuscript. All authors contributed to manuscript revision, read, and approved the submitted version.

SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fenvs.2022.877823/full#supplementary-material

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