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Restoring Balance: Assessing Norwegian Installation-Level Emissions under the Market Stability Reserve

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Abstract

The fight against climate change has reached a crossroads, necessitating critical assessment of the means designed to mitigate the ongoing crisis. This thesis examines the impact of the Market Stability Reserve (MSR) on installation-level emissions and emissions intensities in Norway. The MSR, introduced in 2019, was designed to address the surplus of emissions allowances in the EU Emissions Trading System (ETS) by stabilizing the uphill price trajectory of allowances and thereby incentivize firms to transition towards low-carbon technologies. Its impact is analyzed using installation-level carbon emissions data from Norway's Pollution Transfer and Release Register (PRTR) between 2013-2023. A difference-in-differences (DiD) matching methodology exploiting installation-level inclusion criteria is utilized to estimate the impact of the MSR on installations' emissions and emissions intensities. The findings show no statistically significant effects on absolute emissions, but some significant reductions in emissions intensities are observed. The general lack of significant effects may be due to data constraints, the delayed impact of market-based climate policies, and Norway's already decarbonized economy. Additional research is needed to better understand the MSR's effectiveness, ideally evaluating the policy's impact in the context of a different, less decarbonized country.

Acknowledgements

This research would not have been possible without the mentorship of my supervisor, Alexandra Lopez Cermeno. Her immense support and insightful feedback have been invaluable throughout this entire process. Additionally, I want to thank Katrina Mullan and Derek Kellenberg from the University of Montana, whose passion for environmental and climate change economics sparked my interest in this field. Lastly, I want to express my gratitude to the Emissions Trading Unit at the Swedish Environmental Protection Agency, who introduced me to the Market Stability Reserve and its pivotal role in combating climate change.

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1. Introduction

Environmental economists and policymakers alike have long advocated for putting a price on carbon dioxide emissions as a tool for mitigating climate change. Nordhaus (2018) argued that for any climate policy to be effective, it must seek to increase the price of carbon dioxide and other greenhouse gas emissions. This price adjustment corrects for the underpricing of the negative externality that is climate change. The vision of carbon pricing was realized when the EU voted to establish an emissions trading system, the EU ETS, set to take effect in 2005. Since its launch, the EU ETS has been the cornerstone of EU climate policy. This groundbreaking policy was expected to yield four outcomes. First, it would signal to the consumer about the carbon intensity of goods and services. Second, it would signal to the producers about the carbon intensity of their inputs, thereby incentivizing a shift towards low-carbon technologies. Third, it would provide market incentives for inventors, innovators, and investment bankers to reallocate their focus towards commercializing low-carbon products and processes. Lastly, it would economize on the information required to undertake all these efforts (Agiostratiti, 2019).

These outcomes are indeed desirable. However, the EU ETS has – quite literally – had its ups and downs, caused by a multitude of institutional flaws as well as demand shocks. The main issue being that the prices of emissions allowances have been too low – far below what is believed to be the true social cost of carbon and too low to steer capital investment towards low-carbon technologies (Nordhaus, 2018; Perino & Willner, 2016). The underlying reason behind the suboptimal carbon pricing is rooted in the market surplus of emissions allowances. This surplus amounted to around 2 billion in 2013. The demand shock for emissions allowances was largely due to the economic crisis, as it reduced economic activity and the associated emissions more than what was anticipated (DG CLIMA, n.d.).

As a long-term solution to this, the European Commission (n.d.) introduced the concept of a *market stability reserve (MSR)*, which began operating in 2019 (EUR-Lex, 2015). The MSR adjusts the supply of emissions allowances in the trading system, either by withholding them from auction, storing them, releasing them, or cancelling them. It is tasked with restoring balance between supply and demand of emissions allowances, strengthen compatibility with other external climate policies (e.g., carbon taxation, renewable support, and coal phaseouts), and provide a clear price signal for transitioning towards low-carbon technologies (Perino et al, 2022). Since its launch, the price of emissions allowances has almost quadrupled (World Bank Carbon Pricing Dashboard, 2024). However, there is a current dissensus among contemporary empirical researchers regarding the effectiveness of the MSR, with anticipatory behavior among market participants prevalently mentioned as legitimate risk for market destabilization (Perino, 2018; Rosendahl, 2019; Gerlagh et al, 2020). The ultimate objective of the MSR is to reduce emissions through its price stabilization property. It is therefore reasonable to examine how the policy is performing with respect to this overarching objective.

1.1 Aim and scope

The purpose of this thesis is to analyze the response of regulated firms to the market stability reserve (MSR). More specifically, this thesis asks the following question: what is the effect of the MSR on fossil carbon dioxide emissions of regulated firms? This question is answered through a quantitative analysis of emissions from Norwegian installations between 2013 to 2022. The rationale behind selecting Norwegian installations for this thesis is rooted in the uniquely low reporting threshold of its emissions database – the Pollution Transfer and Release Register (PRTR). In Norway, the threshold is set to 1 kilotonne of fossil carbon dioxide per year. This ensures that the sample comprises a sufficient number of treated and untreated installations. In contrast, the European PRTR only includes installations emitting more than 100 kilotonnes of fossil carbon dioxide per year. Consequently, most installations reporting into the European PRTR would be subject to the MSR, making the empirical analysis unfeasible due to the lack of control installations.

I employ a difference-in-difference (DiD) design, complemented with Kernel matching and fixed effects. The MSR represents a neat setting to investigate the impact of market-based climate policy on environmental performance, given that was designed to solely regulate installations above a specific capacity threshold. Installations falling below this threshold are not included in the system, despite some of them sharing a lot of similarities with regulated entities. These inclusion criteria are exploited in this study, with the aim to compare installations with similar characteristics but which are heterogeneously affected by the regulatory pressures of the MSR. Unfortunately, data on installation-level capacity is not available. To address this, I adopt a similar methodology to that of Dechezleprêtre et al (2018), utilizing pre-MSR emissions growth rates and pre-MSR emissions averages as proxies for capacity-based inclusion criteria. MSR and non-MSR installations are subsequently matched conditioned on these proxy covariates, thereby mitigating the biases associated with selection into treatment.

1.2 Outline of the thesis

The remainder of this thesis is structured as follows. First, some essential context on climate policy, emissions trading, and the MSR is provided in chapter two. In chapter three, the research aim and question are supplemented by a literature review of relevant previous research, comprising thorough investigation into the key theoretical concepts applied to the analysis. Chapter four provides an overview of the data and its respective limitations. Thereafter, a detailed description of the method, i.e., the DiD-Kernel matching estimator, is provided in chapter five. The empirical analysis is carried out in chapter six, providing an explanation of the main findings. Chapter seven discusses the results in the context of the broader empirical spectrum. Lastly, chapter eight concludes the thesis by reflecting on the main findings.

2. Context

2.1 History of climate policy

To fully understand the MSR and the contemporary debate around climate policy, one must delve into the history of environmental issues in the global agenda and the progression of climate change within that particular context.

It was not until 1972 that issues stemming from climate change started to gain recognition in the global scientific and political agenda. Since then, the discourse around climate policy has experienced a lot of turbulence, both in terms of how the problem has been framed and how solutions have been crafted (Gupta, 2010). Held in Stockholm, Sweden in June 1972, the UN Scientific Conference agreed to adopt a declaration that specified an action plan addressing the need for international environmental action. This declaration was the first of its kind to raise the issue of climate change, advocating for governments to be precautious when proceeding with harmful climate change inducing activities and to start evaluating the potential magnitude of anthropogenic climatic effects (Jackson, 2007). The first international policy instrument on climate – the Convention on Long-Range Transboundary Air Pollution – was adopted in 1979. The following years of international environmental policymaking were largely defined by the concern for damages to the ozone layer. Climate change was not a central issue at the time. In fact, the topic remained rather overshadowed in the global political arena all the way up until the ratification of the United Nations Framework Convention on Climate Change (UNFCCC), signed by 158 states in 1992. This was the most significant international action on climate change thus far, as the Convention set out to stabilize atmospheric concentrations of greenhouse gases in a manner that would prevent harmful influence on the climate (Jackson, 2007).

Flash forward to 1997. This year comprised one of the most significant milestones in international climate action up to date, i.e., the adoption of the Kyoto Protocol. The main objective of the Kyoto Protocol was to operationalize the UNFCCC by formally committing industrialized economies, along with economies in transition, to limit and reduce greenhouse gas emissions in line with agreed individual targets (UNFCCC, n.d.). In Annex B of the Kyoto Protocol, binding abatement targets are established for 37 industrialized economies, transitioning economies, and the EU. An essential component of the Kyoto Protocol was the establishment of flexible market-based mechanisms, based on the *trade of emissions permits*. This mechanism was designed to incentivize emissions abatement where it is most cost-effective based on the notion that as long as emissions are removed from the atmosphere, the location of emissions reduction is irrelevant (UNFCCC, n.d.). It was this particular element of the Kyoto Protocol that paved the way for the EU ETS.

2.2 The EU Emissions Trading System

To meet the obligations under the Kyoto Protocol in 1997, the EU decided to establish a scheme for greenhouse gas emission allowance trading (Directive 2003/87/EC). The system was launched in 2005, with the objective of reducing emissions from large emitting industries and the energy sector. An EU-wide harmonized carbon tax was also considered, but it was deemed politically unfeasible due to the relinquishment of sovereignty in taxation of its member states (Borghesi et al, 2023).

The EU ETS operates under a framework of mutual EU-legislation and covers all member states, as well as Norway, Iceland, and Liechtenstein. Approximately 13,000 installations are included in the EU ETS at present. Manufacturing industry and incineration plants producing heat and electricity make up the majority of the installations. The inclusion of an installation within the EU ETS largely depends on its production and energy capacity (Swedish Environmental Protection Agency, n.d.).

The following section will focus on the key features of the EU ETS, providing a holistic perspective to facilitate the sensemaking of the MSR and its role in climate change mitigation.

2.2.1 Economic theory

Understanding the economic theory behind emissions trading can help us gain a better understanding of the overall intentions of market participants and their potential response to the introduction of the MSR. Ultimately, it is the theoretical foundation of emissions trading that shapes its outcomes.

2.2.1.1 Externality theory

The externality theory lies at the theoretical core of emissions trading. When individuals, households, and firms actively engage in consumption, production, or investment opportunities, these decisions oftentimes affect third-party actors who are not directly involved in the transactions. There are times when these external effects are nothing but trivial, but there are other times when the effects are problematic. Large external effects are what economists call *externalities*, which constitutes one of the main reasons for government intervention in the economy (IMF, n.d.). Neoclassical economists have long recognized inefficiencies related to technical externalities as market failures, considering that the indirect effects of a transaction have an impact on others, but the market price of the product fails to account for those externalities (IMF, n.d.). There are two types of externalities – *positive* and *negative*. For this thesis, we will focus on the concept of negative externalities.

Greenhouse gas emissions – associated with market transactions – represent a negative externality. The decision-making of a greenhouse gas polluter is solely based on the direct costs and profit opportunities associated with production. The indirect costs levied on those harmed by the greenhouse gases, is excluded from the calculus. These indirect costs of climate change are plentiful – a lowered quality of life for instance. Given that these indirect costs are neither borne by the producer nor the end user, the social costs become larger than the private costs (IMF, n.d.).

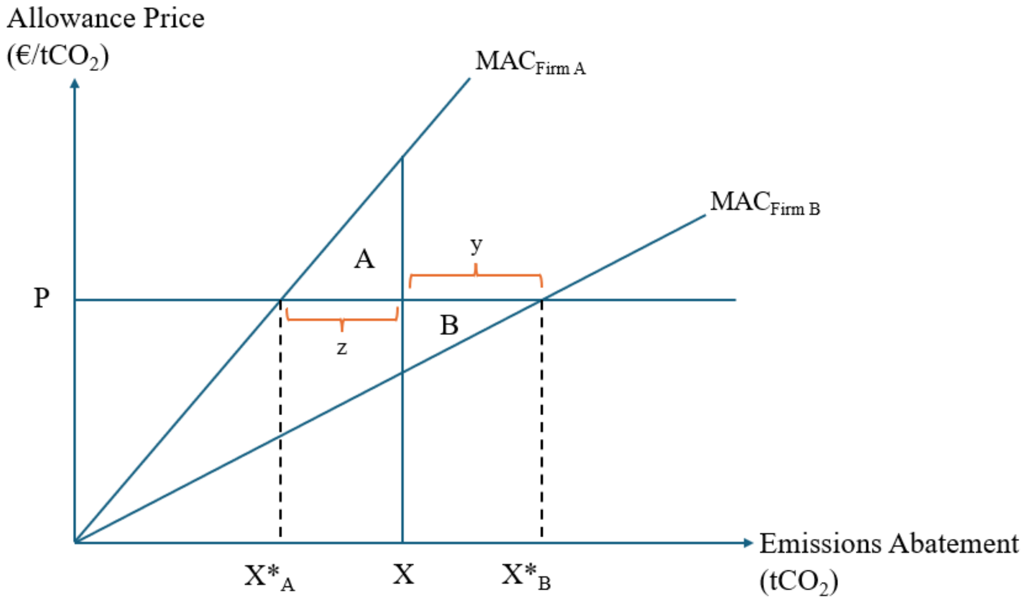
Arthur Pigou, a British economist, contributed significantly to the externality theory through his seminal work in The Economics of Welfare (1920). He argued that when the marginal social cost and the marginal private costs differ, government intervention is necessary (Zheng et al, 2023). Ronald Coase (1960), another prominent economist, utilized the theory of property rights to address the externality problem. Coase (1960) suggested that the fundamental economic function of property rights is to overcome the externality obstacle and mitigate the social costs, thus ensuring an efficient allocation of resources (Zheng et al, 2023).

2.2.1.2 Emission trading theory

Understanding emissions trading theory is crucial for understanding the role and potential impact of the MSR. The theoretical basis for carbon dioxide emissions trading refers to the interactive adjustment of carbon emissions between various sources of pollution, i.e., firms, through currency exchange (Zheng et al, 2023). An emissions trading system internalizes the negative externality of greenhouse gas emissions by putting a price on every ton of carbon emitted into the atmosphere. The market price of carbon emissions is derived through trading.

Assume we have two firms: firm A and firm B. As shown in Figure 1, these two firms display different marginal costs of abatement (MAC). Reducing emissions is more costly for firm A, given its steeper MAC curve. Reducing emissions is less costly for firm B, given its flatter MAC curve. Assuming that the emission allowances are equally distributed to the two firms, each firm is obliged to reduce X tons of carbon dioxide (tCO_2). The total emission reduction is therefore equivalent to 2X tCO_2, and the market price of an emissions allowance is set to P. In this scenario, firm B will abate X^*_B tCO_2, which indicates that the firm has abated an excess amount of y. Firm B is able to sell the excess amount of emissions allowances at the price of P, generating a net income of area B. On the other hand, firm A efficiently abates X^*_A tCO_2, and discovers that it is more beneficial to purchase z emissions allowances at the price of P instead of reducing their own emission given its steep MAC curve. By doing this, firm A saves a net cost of area A. The firms thus utilize the emissions trading system in accordance with their own interests. The total reduction post-trading is still 2X, and the MAC for both firms is equivalent to the market price of emissions allowances. The idea is that emissions reductions occur among firms where abatement is cheapest.

Figure 1. Economic theory of emissions trading. Source: Zeng et al, 2023.



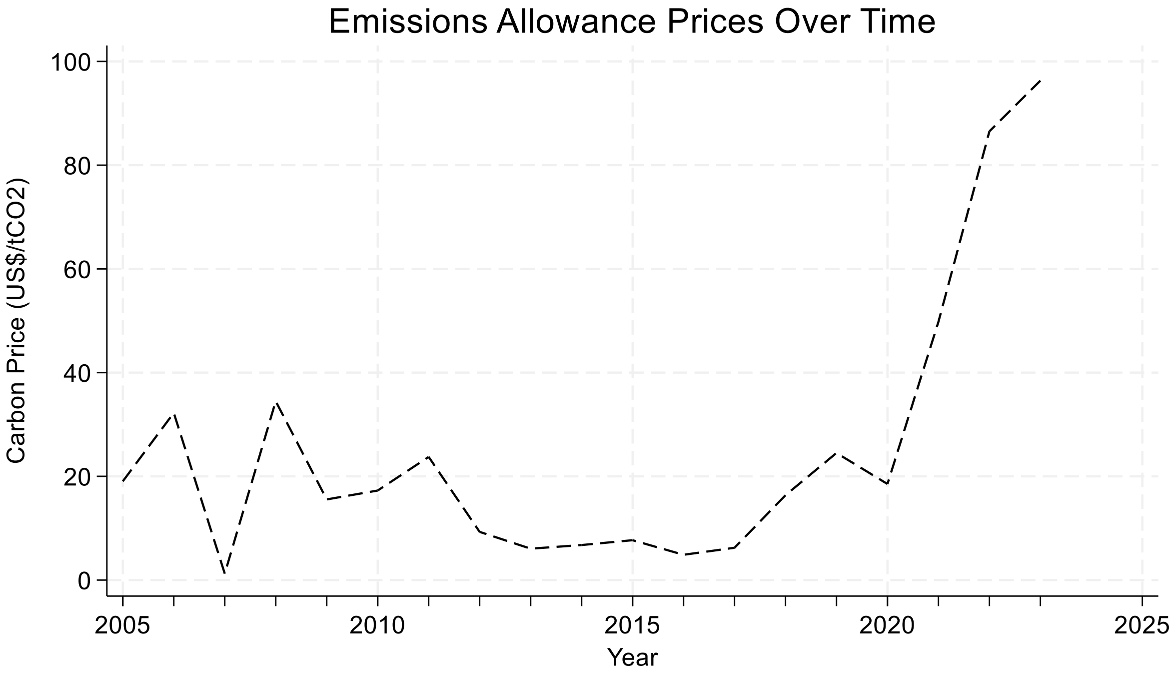
Firm B can make a profit by selling their excess emissions abatement in the form of emissions allowances. Firm A can purchase emissions allowances instead of reducing emissions, which poses a more cost-effective alternative in the meantime. The general idea behind the EU ETS is that the supply of allowances, i.e., the cap, is reduced annually. The scarcity of allowances will result in a higher value of P, thereby incentivizing abatement efforts successively over time.

2.2.2 Carbon pricing

The pricing of greenhouse gas emissions in emissions trading systems, i.e., the price of allowances, depends on the balance of supply and demand of those allowances in the market. In the EU ETS, the EU sets a cap on the maximum level of emissions within the system and creates allowances, each representing one ton of carbon dioxide allowed within the system. The cap shrinks annually at a rate in line with the EU's overarching climate targets, thereby ensuring long-term emissions reductions overtime. The cap – along with the subsequent trading that follows – offer firms certainty about the long-term scarcity of allowances, thereby ensuring that allowances have a market value (Aldy & Stavins, 2012; EU Commission, n.d.).

The price of emissions allowances is not isolated, however, meaning it is not shielded from exogenous price determinants. The relative weights of exogenous price determinants have changed over time. The carbon price in the initial stages of the EU ETS was largely affected by electricity prices, followed by oil, coal, and natural gas prices (Bu & Jeong, 2011). However, over time, other studies have found other variables, such as past allowance prices, to have the greatest effect on the current emissions allowance price (Chung et al, 2018). The importance of coal prices also seems to have surpassed that of oil prices over time (Chung et al, 2018). Figure 2 shows the changes in emissions allowance prices in the EU ETS, from its implementation in 2005 up to today.

Figure 2. Emissions Allowance Prices over Time. Source: World Bank Carbon Pricing Dashboard (2024).



2.2.3 Allocation of allowances

There are two basic methods of allocating allowances in the emission trading system: auctioning and free allocation.

Auctioning constitutes the most fundamental of the two. It has its roots in the polluter-pays principle – a principle at the core of all environmental policymaking in the EU. The auctioning is held at the European Energy Exchange (EEX), located in the German city of Leipzig. Market participants are able to purchase allowances at the real-time market price. The revenues generated from the auction mostly feed into the national budgets of member states based on historical national emissions levels (Swedish Environmental Protection Agency, n.d.). The revenues are earmarked towards investments into renewable energy, energy efficiency improvements and low-carbon technologies that facilitate further national emissions reductions (DG CLIMA, n.d.).

The other method of allocating allowances is through free allocation. The rationale behind allocating allowances for free is rooted in the concern for *carbon leakage*. Many of the European firms that participate in the EU ETS are highly exposed to international competition. Purchasing emissions allowances implies an additional cost levied on the firm, which amplifies the risk of that firm moving its operations outside the EU to a country with a more lenient regulatory regime. This is formally known as carbon leakage (Swedish Environmental Protection Agency, n.d.). The amount of free allocation is based on three factors: production levels, emissions-efficiency, and risk of carbon leakage. The allocation of free allowances thus tends to favor installations with higher levels of emissions efficiency, i.e., those with high production levels and low emissions levels (Swedish Environmental Protection Agency, n.d.). Since free allocation depends on emissions efficiency and carbon leakage risk, firms cannot be certain of receiving a sufficient amount of allowances that cover all their emissions. Purchasing additional allowances will then be necessary, either through auctioning or the secondary market.

EU has decided that for the current EU ETS phase, taking place between 2021-2030, 57 % of the emissions allowances will be auctioned and 43 % will be freely allocated (Swedish Environmental Protection Agency, n.d.).

2.2.4 Phases of the EU ETS

Since its launch back in 2005, the EU ETS has gone through a number of considerable revisions. To successfully scale up the trading system, it has been split up into different phases. As of now, there are four distinct phases in the EU ETS: Phase I, Phase II, Phase III, and Phase IV. Each phase is characterized by a distinct emissions cap that decreases in accordance with a so-called *linear reduction factor (LRF)*. The LRF is a key mechanism in the EU ETS, referring to the annual rate at which the total number of allowances available in the system decreases over time.

2.2.4.1 Phase I

Phase I (2005-2007) served as a pilot phase, where the main objective was to establish the infrastructure essential for monitoring, reporting, and verifying emissions from the firms included in the EU ETS. Almost all allowances were allocated for free (Allen & Overy, 2023). The total emission cap was calculated bottom-up, i.e., based on the aggregation of each member state's national allocation plan. Phase I proceeded with an initial cap of 2,096 $MtCO_2e$ (ICAP, 2022).

2.2.4.2 Phase II

Phase II (2008-2012) introduced more stringent regulations, requiring member states (along with newcomers Iceland, Norway and Liechtenstein) to meet concrete emissions targets. Free allocation remained the dominant method of distributing allowances, constituting 90% of all allocations (Allen & Overy, 2023). Phase II proceeded with an initial cap of 2,049 $MtCO_2e$ (ICAP, 2022).

2.2.4.3 Phase III

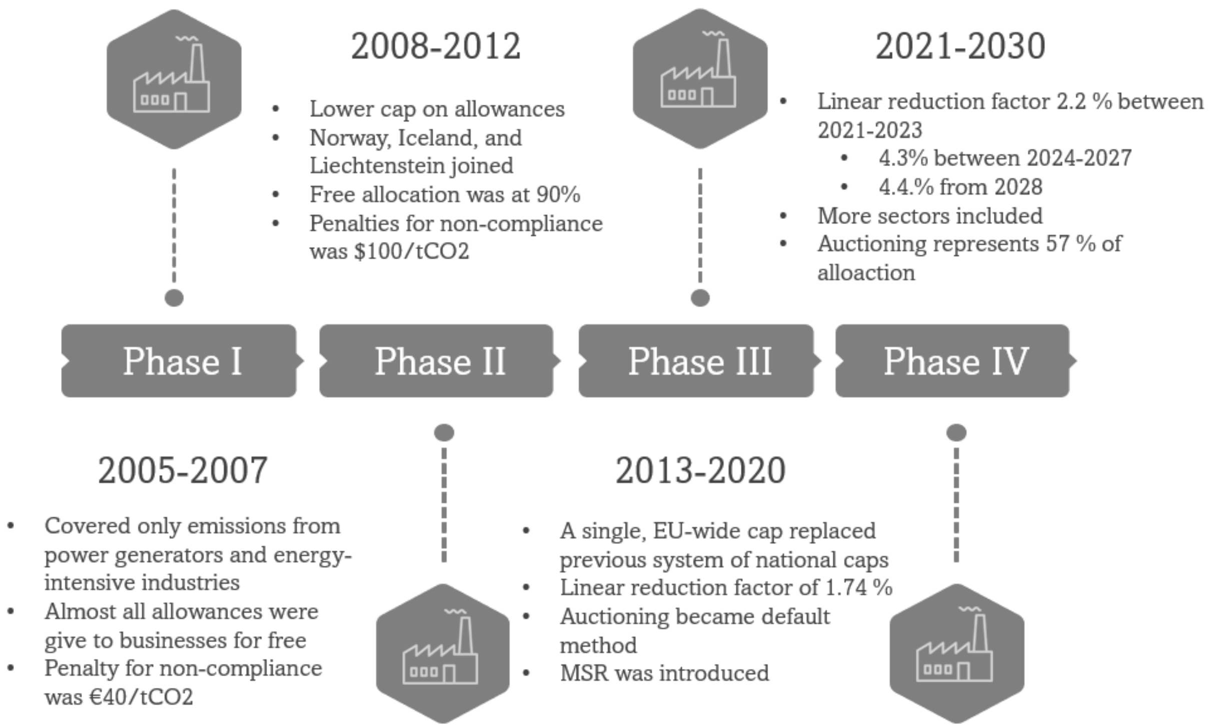
Phase III (2013-2020) underwent substantial revisions in order to address the weaknesses observed in previous phases. Auctioning became the default allocation mechanism instead of free allocation, with remaining free allowances directed towards firms vulnerable to carbon leakage. Additionally, the national emissions caps were replaced by a single, EU-wide cap. This cap was initially set to 2,084 $MtCO_2e$, decreasing in accordance with a LRF of 1.74%. Ultimately, this resulted in a cap of 1,816 $MtCO_2e$ in 2020 (ICAP, 2022).

Last but not least, Phase III also introduced the Market Stability Reserve (MSR). It was first announced in 2015, underwent significant revisions in 2018, and was ultimately implemented in 2019, marking a crucial step towards stabilizing the volatile market (Allen & Overy, 2023).

2.2.4.4 Phase IV

Phase IV (2021-2030) reflects a significant push towards decarbonization, putting the EU ETS at the forefront of EU's decarbonization agenda. The LRF was set to 2.2% between 2021-2023, 4.3% for the period between 2024-2027, and 4.4% from 2028 onwards. Additionally, the cap is set to reduce in two steps of Phase IV – by 90 million allowances in 2024 and 27 million allowances in 2026. Following these revisions, the 2024 amounts to 1,386 $MtCO_2e$ (ICAP, 2022). The scope of the EU ETS was also expanded in 2024 to include maritime emissions and to remove barriers for deploying new low-carbon technologies such as green hydrogen and fossil-free, hydrogen-based steel. Phase IV will also facilitate the phase-out of free allocation, through the introduction of the carbon border adjustment mechanism (CBAM). The CBAM mitigates the risk of carbon leakage by neutralizing the competition from heavy emitters outside the EU (Swedish Environmental Protection Agency, n.d.).

Figure 3. Phases of the EU ETS. Source: DG CLIMA (n.d.) and ICAP (2022).



2.2.5 Surplus issues

The functioning of the EU ETS has previously been compromised by the *surplus of emissions allowances* that started to build up back in 2009 (DG CLIMA, n.d.). The underlying cause of this surplus was particularly rooted in the financial crisis, which reduced emissions more than anticipated. This miscalculation led to lower carbon prices and thus insufficient incentives to reduce emissions (DG CLIMA, n.d.). The surplus amounted to approximately 2 billion allowances in the early stages of Phase III increased to around 2.1 billion at the end of 2013 (DG CLIMA, n.d.).

In an attempt to address this issue, the EU Commission made an amendment to the EU ETS Auctioning Regulation by introducing a *backloading scheme* (DG CLIMA, n.d.). Through the backloading scheme, the Commission postponed the auctioning of 900 million allowances until 2019-2020. This action did not reduce the aggregate number of auctioned allowances in Phase III. Instead, it solely sought to redistribute auctioning throughout the period. Auctioning volumes were reduced by 400 million allowances in 2014, 300 million in 2015, and 200 million in 2016 (DG CLIMA, n.d.). This solution proved to address the imbalance of demand and supply in the short term. However, recognizing the need for a sustainable, long-term solution, the EU Commission engaged in efforts to establish the MSR.

2.3 The Market Stability Reserve

To address the supply-demand imbalances, the Commission decided that a market stability reserve (MSR) should be established in 2018 and operational in 2019 (Decision (EU) 2015/1814). Recall that allowances are allocated annually based on the emissions cap. The purpose of the MSR is to function as a buffer, which activates when there are large differences between the number of allocated allowances and the actual level of emissions (Swedish Environmental Protection Agency, n.d.). If the surplus surpasses a certain upper threshold, emissions allowances are transferred to the reserve instead of flowing freely in the market. If there is a shortage, and that shortage surpasses a certain lower threshold, emissions allowances are transferred from the reserve to the market (Swedish Environmental Protection Agency, n.d.).

Table 1. Brief Explanation of the Mechanisms of the MSR.

Case	**Course of action**
Aggregate surplus of allowances exceeds 833 million.	12% of total number of allowances in circulation is deducted each year from the annual auction volumes and placed in the MSR.
Aggregate surplus of allowances drops below 400 million.	100 million allowances are reinjected from the MSR spread over a 12-month period.
Aggregate surplus lies between 400 million and 833 million.	MSR remains inactive.

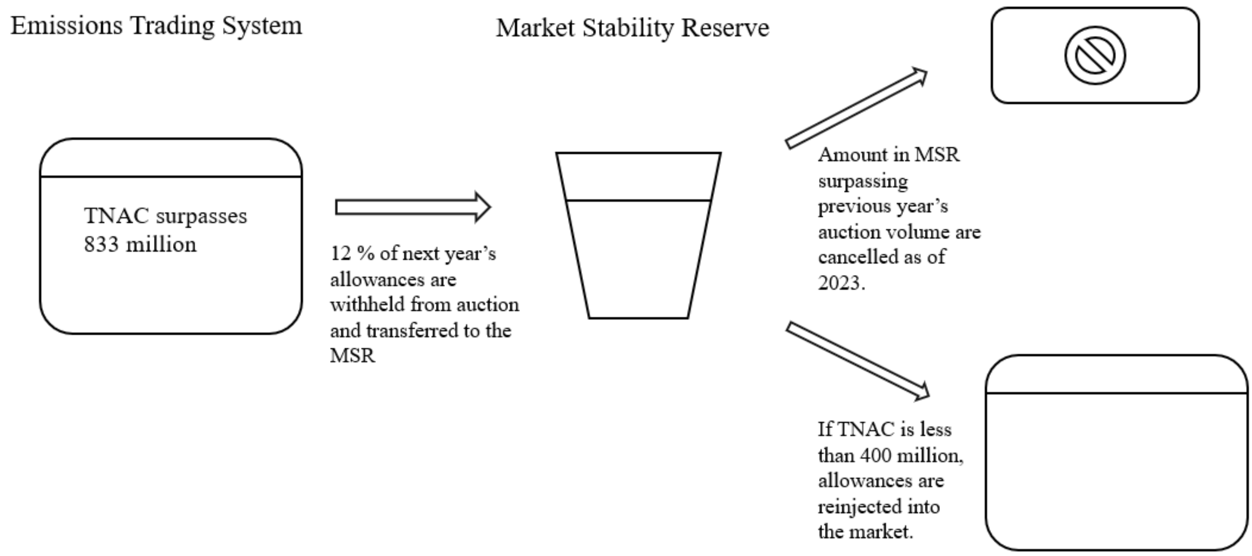
Table 1 describes the initial design of the MSR, established to institutionalize the mechanism of reducing short-term supply of allowances by placing them in the reserve and reintroducing them into those allowances into the market at a later date (Decision (EU) 205/1814). The long-term cap in the EU ETS is thus not altered.

The MSR temporarily adjusts supply by either absorbing allowances or releasing them for auctioning. Absorption of allowances is dependent on two factors – the total number of allowances in circulation (TNAC) and the rate of absorption. The first factor relates to the number of allowances that market participants decide to bank (i.e., set aside for future use) at the end of each year. The second factor relates to the percentage of the TNAC that will be deducted from future auctioning and transferred into the reserve. Table 1 demonstrates that the MSR functions in three distinct ways. Should the TNAC (i.e., the aggregate surplus banked by market participants) exceed the upper threshold of 833 million allowances, the MSR absorbs allowances – that would have otherwise been auctioned out – at a rate corresponding to 12% of the TNAC each year. Should the TNAC drop below 400 million allowances, the MSR reinjects 100 million allowances spread over a 12-month period. When the TNAC lies in between these upper and lower thresholds, the MSR remains inactive (Borghesi et al, 2023).

The initial design of the MSR was legislated back in 2015 (Decision (EU) 2015/1814). This version had no significant impact on prices, however, which was consistent with the empirical predictions that questioned the MSR's inability to adjust the long-term emissions cap (Borghesi et al, 2023; Perino & Willner, 2016). Under this initial scheme, all allowances absorbed by the MSR would ultimately be re-released into circulation. However, this ineffective scheme never became operational. The policy design of the MSR underwent significant revision in 2018, which would substantially alter the functioning and impact of the MSR.

At the core of this revision lies the *cancellation mechanism.* The cancellation mechanism was a proposal made by Sweden during the EU negotiations to revise the initial MSR (Swedish Environmental Protection Agency, n.d.). It was designed to endogenize the emissions cap, thus making it a function of both banking behavior and of current and future demand for emissions allowances (Borghesi et al, 2023). The cancellation mechanism functions in the following way: from 2023, emissions allowances surpassing a certain threshold are cancelled annually. The threshold applied in 2023 was the auction volume of 2022. From 2024 onwards, the threshold is fixed at 400 million (ICAP, 2022). Figure 4 illustrates the core characteristics of the revised MSR with the add-on cancellation mechanism.

Figure 4. Market Stability Reserve with Cancellation Mechanism.



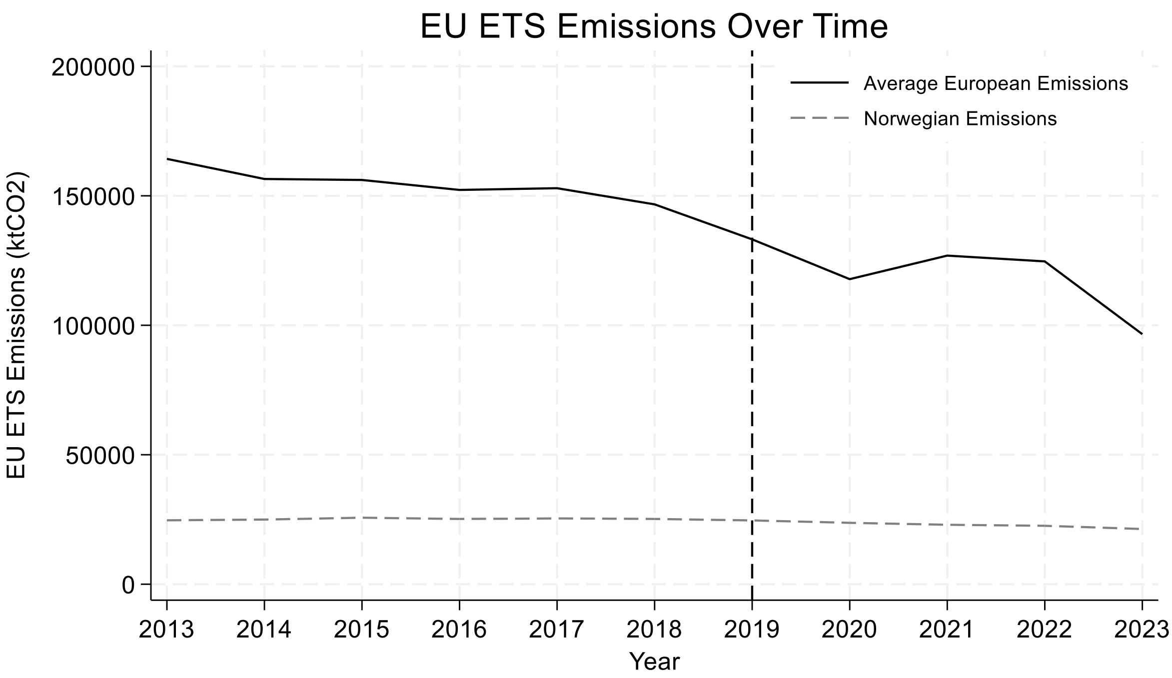
2.4 The Case of Norway

2.4.1 Norway as a testing ground

Norway has set some ambitious targets for itself regarding the reduction of greenhouse gas emissions and the establishment of a low-carbon society by 2050. With its abundance of energy resources, particularly hydropower, Norway is well-positioned in the global energy transition (IEA, n.d.). 92% of its electricity generation is covered by hydropower, making the country's power sector almost entirely reliant on renewables. Furthermore, energy demand in Norway is vastly electrified, with electricity accounting for nearly half of the country's total final consumption in 2020, constituting the highest share among all IEA countries (IEA, n.d.).

Its abundant reserves of oil and natural gas has made Norway a net exporter of energy. In fact, 87% of its aggregate energy production in 2020 was exported (IEA, n.d.). The oil and gas sector plays a pivotal role in the Norwegian economy. As most fossil fuels are exported, Norway's domestic emissions remain relatively low compared to international standards. Figure 5 illustrates Norway's EU ETS emissions relative to the average EU ETS emissions of other participating countries, highlighting Norway's significant lead in decarbonization compared to its EU ETS counterparts.

Figure 5. Norwegian Emissions versus Average European Emissions. Source: EEA (2024).



To attain its ambitious target of becoming a low-carbon society by 2050, Norway has considerable work ahead, especially since a many of the easy wins have already been achieved through widespread electrification across the country. Using Norway as a testing ground for this thesis presents both opportunities and challenges. Investigating the Norwegian response to the MSR could yield valuable insights given its strong track record in domestic decarbonization. However, there is also the risk of reaching no findings at all, given that Norway has already made substantial progress in decarbonizing its economy. The country has already begun to decouple its emissions and economic growth; between 2000-2020, energy-related emissions increased by 10% while GDP increased by 35%. This decoupling has been more evident since 2015, as energy-related emissions declined while GDP kept growing. Norway's carbon emissions per unit of GDP was the sixth lowest among all IEA countries in 2020 (IEA, 2022). These impressive numbers are reflected by what is illustrated in Figure 5.

The MSR might thus not have the desired impact on Norwegian installations, considering its high levels of electrification and that most of its fossil fuels are exported and combusted in other countries. To address this regulatory loophole, the European Commission has announced EU ETS2, scheduled to come into effect in 2027. This new system will cover upstream emissions, thereby regulating fuel suppliers releasing fuels into the market for consumption (DG CLIMA, n.d.). While the EU ETS2 and Norway's outsourcing of emissions lie beyond the scope of this thesis, it could offer additional nuance to the discussion of empirical results as it will undoubtedly be affected by the regulatory pressures of the upcoming EU ETS2.

2.4.2 Norway in the EU ETS

Norway's role in the EU ETS has not always been clear. Given its emergence as an oil-exporting country, highly dependent on resource-intensive economic activity, the EU ETS was deemed as an appropriate market-based mechanism for Norway to mitigate the substantive climate impact of its heavy industrial activities (Klemetsen et al, 2016).

During Phase I, Norway had a separate emissions trading system, not formally linked to the EU ETS given Norway's non-member status. Emissions allowances were still accepted by Norwegian authorities, however, so Norwegian installations were able to purchase emissions allowances from other installations in the EU (Klemetsen et al, 2016). Trade was sparse among Norwegian ETS installations, accounting for approximately 0.1% of total emissions from all installations. This limited trade suggests that allocation was overly generous in the initial stages of Norway's participation in the EU ETS. In fact, total allocation exceeded total emissions by 8% (Klemetsen et al, 2016). In the 1990s, Norway introduced a carbon tax in numerous sectors of the national economy, which resulted in many industries (including oil and gas industry) being exempt from Phase I of the EU ETS. From 2008 onwards, exemption of industries was no longer allowed. The share of Norwegian emissions covered by the EU ETS increased from 10% to 45% as a result (Klemetsen et al, 2016).

As mentioned earlier, auctioning began to successfully replace free allocation as the default method of allowance distribution. Norway was thus allowed to auction an increasing amount of allowances in Phase II. Worth noting is that oil and gas industry did not receive any free allowances during Phase II. But as a result of the harmonization (and the unified emissions cap) in Phase III, oil and gas industry began to receive a substantial number of allowances for free (Klemetsen et al, 2016).

3. Theory and Previous Research

Conceptual and theoretical framework

The theoretical framework on which this thesis is based draws upon the induced innovation hypothesis developed by Hicks (1932), stating that changes in relative factor prices result in innovations that lessen the need for that relatively expensive factor. The Hicksian notion of innovation inducement was later augmented by Newell et al (1999) who accounted for additional influence of government regulation. This thesis extrapolates the frameworks laid out by Hicks (1932) and Newell et al (1999) to fit into the context of the MSR and the realm of carbon pricing and emissions abatement. Firms will respond to the emissions price effects – brought about by the MSR regulation – by adopting low-carbon technologies and thus lower their emissions.

Since the MSR started operating in 2019 and the cancellation mechanism began in 2023, the aggregate supply of emissions allowances was not immediately adjusted. Environmental proactiveness thus becomes an interesting topic for the discussion of results. Hence, this thesis will also incorporate the concepts introduced by Hart (1995) and Buysse & Verbeke (2003), who utilize a resource-based view of the firm to explain various forms of strategic environmental proactiveness.

3.1 The Role of Firms in the Climate Transition

A frequently asked question among firms is how to respond to governmental actors seeking to modify industry behavior. Firm-level behavioral changes and the magnitude of those changes are a result of both internal and external factors. The internal relates to the impact on firm performance and relevant technical competencies while the external relates to the costs of non-compliance (Andreou & Kellard, 2020). On the other end, governmental and non-market actors ask themselves how to ensure that the implementation and design of their policies induce the desired firm-level response. In the case of the MSR, the EU designed the policy in hopes of stabilizing the price of emissions allowances and bringing it closer to a level that represents the social cost of carbon, a concept popularized by Nordhaus (2014). A desired firm-level response to the MSR was to induce *proactiveness*, i.e., firms beginning to adopt low-carbon technologies in the knowledge that the surplus of emissions allowances will be corrected for in the near future through the cancellation mechanism.

Whether firms are proactive or not is a crucial question to ask for the purpose of this study. Lumpkin & Dess (1996) list proactiveness as one of the five key dimensions of a firm's entrepreneurial orientation. Proactiveness, as defined by Lumpkin & Dess (1996), refers to how firms react to market opportunities by taking initiative in the marketplace. It is a forward-looking perspective characterized by acting in anticipation of future demand (Andreou & Kellard, 2020). This anticipatory behavior undoubtedly reflects a consideration for the environment, such as climate mitigation. Research on anticipatory behavior in emissions trading is abundant. Strategies in emissions trading systems can be either anticipatory or adaptive in response to regulatory and stakeholder uncertainty. Anticipatory strategies are characterized by preparing for various regulatory scenarios, while adaptive strategies are characterized by avoiding abatement efforts until regulatory uncertainty lessens (Patnaik, 2020; Engau et al, 2011).

Hart (1995) zooms in on the firm's relationship with the natural environment and utilized his findings to develop a natural-resource based view of the firm. This theory, deeply rooted in competitive advantage, comprises three interconnected capabilities: pollution prevention, product stewardship, and sustainable development. Pollution prevention is perhaps the most central factor in this thesis. The environmental driving forces behind pollution prevention include minimizing emissions, effluents, and waste, while the competitive advantage takes the form of lower costs (Hart, 1995). This theoretical approach was further nuanced by Buysse & Verbeke (2003), who listed five resource domains that draw upon Hart's (1995) classification of strategic environmental capabilities. The first domain –

investments into green technologies – is the most evident indicator of a firm's environmental proactiveness (Buysse & Verbeke, 2003; Rugman & Verbeke, 1998; Andreou & Kellard, 2020). This domain is of particular interest in the case of the MSR. Firms allocating investments into low-carbon technologies prior to MSR's cancellation mechanism kicking in, demonstrate proactiveness through Buysse & Verbeke's (2003) first domain. In fact, the main mechanism in the EU ETS as a whole is designed around the notion of proactiveness, given that unused emissions allowances can be sold in the EEX. Firms regulated by the system are thus incentivized to demonstrate proactiveness (Andreou & Kellard, 2020).

If proactiveness is on one end of the spectrum, we find *reactiveness* on the opposite end. Reactive corporate environmental practices emphasize compliance with environmental regulation at a minimum level (Kim, 2018; Buysse & Verbeke, 2003). Kim (2018) finds reactive corporate environmental practices to be associated with worse environmental performance as compared to those of proactive environmental practices. The findings are consistent with the natural-resource based view illustrated by Hart (1995), which also suggested strong association between proactiveness and superior environmental performance.

Moving onto another corner of this research field, we identify studies highlighting the importance of heterogeneous firm characteristics in explaining the type of climate change strategy pursued. These characteristics include organizational structure, firm size, industry sector, and regional affiliation (Patnaik, 2020; Damert & Baumgartner, 2017; Wang et al, 2019). Damert & Baumgartner (2017) mention regional affiliation and firm size as key determinants of the type of climate change strategy pursued, finding larger firms to exhibit more sophisticated response strategies due to higher levels of public scrutiny and access to capital. Wang et al (2019) expand on this by investigating the importance of market power in climate change strategy, finding that a firm's market power can significantly accelerate its adoption of emissions abatement technologies.

3.2 The Role of Carbon Pricing in the Climate Transition

To obtain a better understanding of the MSR and its place in the climate transition, we must delve into the role of carbon pricing. Economists have long been making a case for carbon pricing in the climate change conversation (Boyce, 2018). A price on carbon subsequently internalizes the negative externality of greenhouse gas emissions, otherwise not accounted for in conventional markets. One of the most prevalent justifications for implementing a carbon pricing scheme is related to cost-effectiveness (Boyce, 2018; Lin & Wesseh Jr, 2020; Nordhaus, 2007). Economic agents are given more freedom in how they choose to abate their emissions. The marginal cost of abatement differs between technologies and firms. Boyce (2018) sheds light on this variance, stating that installing LED lighting or converting to wind power in a favorable area come at a relatively low cost, whereas transitioning towards carbon capture and sequestration (CCS) comes at a significantly higher cost. Carbon pricing policies – especially emissions trading schemes – provide a flexibility for firms to reduce emissions in a way that financially makes sense for them. By inducing firms to determine the optimal timing of carbon emissions and allowance usage, an emissions trading scheme somewhat flirts with Hotelling's (1931) problem of efficiently extracting exhaustible resources traded in a competitive marketplace defined by conditions of uncertainty. Quemin & Trotignon (2019) expand on this notion by highlighting that as firms minimize costs over time, intertemporal emissions trading combined with an absence of arbitrage ensures that the carbon price reflects the expected long-term scarcity of permits resulting from the declining cap trajectory.

Some studies have highlighted the risks of carbon pricing, questioning its effectiveness irrespective of the price being set too high or too low (Zakeri et al, 2015; Fahimnia et al, 2013). A crucial aspect to consider in this field of research is the effectiveness of carbon pricing in *stimulating low-carbon innovation*. As regulated firms expect to face higher emissions prices relative to other production costs, they are incentivized to invest into operational changes that reduce the emissions intensity of their products (Calel & Dechezleprêtre, 2016). This has its roots in the *induced innovation hypothesis*, coined by the late Sir John Richard Hicks (1932) in his book *The Theory of Wages*. Hicks (1932, pp. 124-125) argued that "a change in the relative prices of the factors of production is itself a spur to invention, and to invention of a particular kind – directed to economizing the use of a factor which has become relatively expensive." Newell et al (1999) tested Hick's induced innovation hypothesis, augmenting it by applying it to the case of energy prices and energy-efficiency innovation. The authors also proceed to generalize the Hicksian idea of inducement to assess the influence of government regulation on energy-efficiency innovation, which becomes particularly relevant for the context of this thesis. Newell et al (1999) provide evidence for both energy prices and government regulation having an impact on the energy efficiency of firm output. The induced innovation hypothesis (Hicks, 1932) – in conjunction with the government regulation augmentation presented by Newell et al (1999) – will serve as a guiding theoretical framework for this thesis.

Another literature strand relevant to the context of this thesis is the Porter hypothesis, developed by economist Michael Porter (1991). In his hypothesis, Porter (1991) suggests that strict environmental regulation can benefit polluting firms by stimulating innovations which subsequently raises firm productivity and the end-user product value (van Leeuwen & Mohnen, 2016). Lin & Wesseh Jr (2020) used econometric modeling to test the Porter hypothesis in the context of the Chinese emission trading scheme. The authors identify a notable impact of carbon pricing on the propensity to innovate among energy-intensive firms. However, this innovation only took place only took place in the latter stages of the value chain during distribution and marketing. Energy innovations during the production stage – i.e., at the actual installation source – were few. Lin & Wesseh Jr's (2020) conclusions are in line with the findings of Liliestam et al (2020), who find little to nil evidence of the effectiveness of carbon pricing in promoting technological change and decarbonization. Policies are thus needed to induce innovation during the production stages (Lin & Wesseh, 2020; Liliestam et al, 2020). Tietenberg (2013) emphasizes that although innovation gains from emissions trading have not reached the desired impact level, many studies still find statistically significant responses to the market-based policy instrument. This finding is far from universal though, as carbon pricing may unintentionally result in exploitation of existing options (e.g., existing low-carbon fuels) rather than encouraging adoption of new technologies (Taylor et al, 2005).

There are essentially two main types of carbon pricing: emissions trading and carbon taxation. Emissions trading places a cap on the total level of emissions and lets the market set the price. Carbon taxation, on the contrary, sets the price but lets the market determine the total level of emissions (Tietenberg, 2013). Goulder & Schein (2013) critically compare the two instruments, with particular emphasis on the aspect of *price volatility*. This is not an issue under a carbon taxation scheme, as the carbon price is the tax rate. Policymakers can thus always be certain that the price is stable. When discussing emissions trading, the authors highlight that price volatility is a significant issue. As a result of the fixed emissions cap, the supply of allowances is perfectly inelastic. Shifts in demand can thus cause significant changes in allowance prices, and the irregular demand shifts can yield price volatility which can be labeled as a source of inefficiency (Goulder & Schein, 2013). Nordhaus (2007) uses this argument of price volatility to advocate for carbon taxation rather than emissions trading. The observed volatility and general instability in emissions trading has been one of the main catalysts for introducing the MSR, designed to stabilize the allowance market, enhance supply-side flexibility, and boost market resilience to external shocks (Chaton et al, 2015).

Conclusively, the empirical evidence suggests that the effectiveness of carbon pricing solely represents a theoretical argument at this point in time (Lin & Wesseh, 2020; Liliestam et al, 2020; Taylor et al, 2005). Additionally, emissions trading has been met with optimism for its favorable cost-effectiveness qualities, but skeptics advocate for caution due to the issue of price volatility (Goulder & Schein, 2013; Nordhaus, 2007). Consequently, these empirical arguments highlight the importance of studying the MSR, as it could potentially be the policy that brings forth the market stability and technological change necessary for transitioning towards the fossil-free economy, ultimately shifting the carbon pricing argument from *theoretical* to *practical*.

3.3 EU ETS and Low-Carbon Technology Investments

The EU ETS has been the cornerstone of EU's decarbonization strategy since its adoption in 2005 and was introduced with the expectation of having a central role in incentivizing the shift towards low-carbon technologies (Texidó et al, 2019; European Commission, 2017). The potential of an environmental policy to induce low-carbon technological change is empirically understood as the optimal measure for its success (Calel & Dechezleprêtre, 2016; Pizer & Popp, 2008; Kneese & Schultze, 1975). In light of this empirical illumination, theoretical and empirical economists alike have attempted to understand the link between environmental policies and technological change.

The impact of the EU ETS has not been obvious, however, as empirical evidence remains relatively ambiguous. Texidó et al (2019) conducted an overarching review of the existing empirical literature analyzing the effects of the EU ETS on low-carbon technological change. They concluded that investments into low-carbon technologies were hampered in Phase I (2005-2007) and Phase II (2008-2012) as a result of free allocation and the substantial surplus of allowances in the aftermath of the financial crisis. Löfgren et al (2014) arrived at a similar conclusion, finding no statistically significant effects on low-carbon technology investments. Calel (2018) could not identify any substantial effect on firm's adoption of off-the-shelf low-carbon technologies, but the author did identify an increase in low-carbon patenting and R&D spending. Adoption thus remained unchanged while innovation increased. Calel & Dechezleprêtre (2016) – among the more prominent contemporary researchers on innovation impacts of the EU ETS – found no compelling evidence of the system having any impact on the pace or direction of technological change when analyzing the policy impact on a firm level. The authors subsequently concluded that incentives for large-scale adoption of low-carbon technologies are insufficient at present. This begs the question of what role the MSR might have in addressing the lack of incentives for low-carbon technological change.

When looking at the effects of the EU ETS on emissions intensities, many studies find nil to modest effects (Wagner et al, 2014; Jaraité and Di Maria, 2016; Klemetsen et al, 2016; Calel, 2018). These studies are limited to Phase I or Phase II, suggesting that in its initial stages, the EU ETS had little effect on low-carbon technological change and emissions intensities. Klemetsen et al (2017) and Dechezleprêtre et al (2018) take similar approaches to studying the impact of the EU ETS on environmental performance. They are one of the relatively few studies using micro-level data to estimate the effects on environmental performance. Klemetsen et al (2020) focused on Norwegian manufacturing plants in the first three phases of the EU ETS to investigate the effects on emissions reductions but also emissions intensity. They found some evidence, although weak, of emissions reductions in the Phase II of the ETS. The effects in the other phases were statistically insignificant. Effects on emissions intensity were insignificant in all phases. Dechezleprêtre et al (2018) focused on four EU/EEA countries (France, Norway, Netherlands and the UK), to study the effects of the EU ETS on emissions reductions, finding significant emissions reductions between 2005-2012. Wagner et al (2014) and Petrick & Wagner (2014) used French plant-level data and German plant-level data respectively, and both found evidence of significant emissions reductions in the Phase II.

Emissions reductions can be a troublesome variable to observe, however, since a decline in emissions does not necessarily imply that the EU ETS is causing this decrease. Emissions reductions could have occurred in absence of the EU ETS as a result of technological or macroeconomic shifts (Dechezleprêtre et al, 2018). Emissions intensity, a variable which controls economic activity, could thus be more suitable in analyzing the impact that the MSR has had on installation-level environmental performance.

3.4 The Market Stability Reserve

Given the nascent stage of the MSR, research in this area is still relatively scarce. The MSR is part of EU's efforts to deliver more ambitious goals as formulated in the EU's Green Deal and Fit-for-55 package (Osorio et al, 2021).

Perino et al (2022) shed light on the MSR by summarizing the main strengths and weaknesses of the MSR. The MSR has been subject to controversy since is announcement, as environmental economists have identified several shortcomings with the policy. There is risk of price volatility increasing, external climate policies overlapping with the EU ETS being undermined, and speculative behavior negatively influencing the system as a whole. Perino (2018) brings up the notion of the *waterbed effect* in the EU ETS, i.e., the problem of overlapping climate policies in the EU influencing who emits but not how much is emitted in total. The waterbed effect alludes to sitting down on a waterbed, which in turn changes the distribution of water in the bed, but the bed as a whole does not lose any water, i.e., carbon emissions (see Appunn, 2019). The MSR was designed to puncture the waterbed effect and thus address the problem of overlapping climate policies being undermined, but scholars have questioned the extent to which this puncturing occurs (Perino, 2018; Rosendahl, 2019; Gerlagh et al, 2020). Perino (2018) argues that the puncturing induced by the MSR is temporary, and that the problematic waterbed effect in a few years' time returns in full force as the MSR no longer accepts allowances (Rosendahl, 2019). Gerlagh et al (2020) also pointed to the temporary nature of the MSR when investigating it in the context of external shocks. The authors claim it to be effective in dealing with short-term demand shocks but is very sensitive to long-term expectations. This aligns with the findings of Rosendahl (2019), who emphasized the risks of speculative anticipatory behavior of market participants undermining the MSR by increasing cumulative emissions. Rosendahl (2019) underscores that his findings do not suggest that the MSR yields counterproductivity, and stresses that the MSR still has immense potential to reduce gigatons of cumulative emissions.

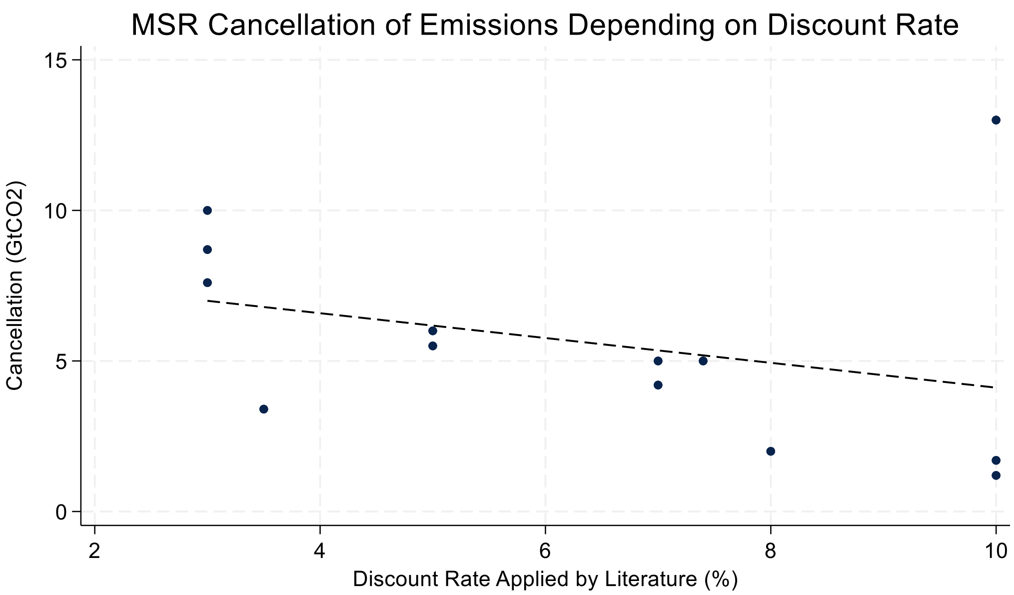
The initial MSR policy, announced October 2016, was designed to reduce the supply of allowances when the total number of allowances in circulation (TNAC) reaches a certain upper threshold, transferring these allowances to the MSR. If the TNAC drops below the lower threshold, allowances are released from the MSR (Osorio et al, 2021). Perino & Willner (2016) highlighted the fact that the initial MSR was cap-neutral, thus having a limited effect on the allowance price, and could also disincentivize long-term investments (Perino & Willner, 2016). The risk of increased price volatility was also given serious thought (Mauer et al, 2020; Richstein et al, 2015; Perino et al 2022). However, these findings were dismissed by Fell (2016), who found the opposite.

Another important strand of the MSR literature relates to the *cancellation mechanism*. This was a reform to the initial MSR to address the critique listed above. The decision to strengthen the MSR was agreed upon by the European Parliament and Council of the European Union in 2018 (Directive (EU) 2018/410). The cancellation mechanism implies the following: should the number of allowances stored in the MSR exceed the number of auctioned allowances in a given year, that surplus of allowances is *permanently cancelled* from 2023 and onwards (Osorio et al, 2021). The decision to strengthen the MSR (Directive (EU) 2018/410) was paired with a decision to increase the linear reduction factor (LRF) from 1.74% to 2.2% after 2020, i.e., the rate at which the emissions cap decreases in the EU ETS. These reforms evoked a large number of subsequent studies, particularly on the cancellation mechanism. Bruninx et al (2020) found that the cancellation mechanism would result in cumulative emissions decreasing 13 $GtCO_2$. Carlén et al (2019), on the other hand, found that the cancellation mechanism would only reduce cumulative emissions in the EU ETS by 2.4 $GtCO_2$. The differences in results can be explained by the discrepancies of assumed discount rates. If firms operate at a higher discount rate, they value the future less, thus implying less banking of emissions allowances. If firms bank less, fewer allowances are transferred to the MSR, subsequently leading to lower levels of cancellation (Osorio et al, 2019). Cancellation of emissions depends greatly on the discount rates assumed by empirical studies, and Table 2 and Figure 6 below suggest that cancellation is higher when discount rates are lower.

Table 2. Empirical Findings on Cancellation and Discount Rates

Source	**Cancellation (Gt)**	**Discount rate**
Bruninx et al. (2020)	13	10%
Quemin & Trotignon (2019)	10	3%
Quemin (2020)	8,7	3%
Tietjen et al. (2021)	7,6	3%
Beck & Kruse-Andersen (2020)	6	5%
Gerlagh et al. (2021)	5,5	5%
Silbye & Sørensen (2018)	5	7,4%
Quemin & Trotignon (2019) b	5	7%
Quemin (2020) b	4,2	7%
Carlén et al. (2019)	3,4	3,5%
Bocklet et al. (2019)	2	8%
Perino & Willner (2016)	1,7	10%
Mauer et al. (2019)	1,2	10%

Figure 6. Cancelled Emissions Allowances by Discount Rate.



The empirical studies seen above in Table 2 all focus on cumulative emissions, whereas this thesis leans more towards firm-level abatement. However, the conclusions around the MSR's influence on the banking behavior of firms with respect to discount rates will be useful in the discussion of results, to help make sense of the empirical findings derived in this thesis.

3.5 Econometric studies in environmental regulation

Econometric modeling provides a rigorous and empirical framework for analyzing the impact of environmental regulations, enabling policymakers to craft more effective policies to mitigate environmental challenges, all while simultaneously taking social and economic aspects into consideration. Several studies have leveraged the feasibility of econometric modeling to assess the impact of various environmental regulations, not least the EU ETS.

Jaraité & Di Maria (2016) empirically estimated the changes in installation-level environmental performance relative to what would have occurred had the EU ETS not been introduced. Since the authors were determined to identify a causal effect of the EU ETS, they exploited the unique characteristics of the EU ETS to derive estimates of counterfactual environmental outcome variables. Using econometrically adjusted outcome variables of firms not subject to the EU ETS over the same period, they were able to construct a counterfactual. When discussing the potential use of difference-in-differences (DiD) model to derive the average treatment effect of the EU ETS, they highlight the risk of bias as firm-level outcome variables may vary substantially across treatment and control groups. Concerned with these biases that make a simple DiD estimation unfeasible, the authors employed a matching methodology to control for confounding factors affecting both regulated and non-regulated installations as well as installation-level heterogeneity. Dechezleprêtre et al (2018) utilize the same matching estimation strategy. The matching methodology by Jaraité & Di Maria (2016) and Dechezleprêtre et al (2018) was first developed by Heckman, Ichimura, and Todd (1997) and Heckman et al (1998), who introduced the concept of the DiD matching estimator using propensity scores. Klemetsen et al (2016) adopted the same estimation strategy to study the impacts of the EU ETS on installation-level economic and environmental performance in Norway. Studying a slightly different context, List et al (2003) used propensity score matching to examine the effect of air quality regulation on economic activity, exploiting a county-level dataset to conduct the DiD matching analysis.

Heckman et al's (1997) DiD matching estimation strategy, employed by Dechezleprêtre et al (2018), Jaraité & Di Maria (2016), Klemetsen et al (2016), and List et al (2003) is rooted in the work of Rosenbaum & Rubin (1983), who laid the foundation for propensity score matching as a means to mitigate the bias in observational studies. By assigning propensity scores – representing the probability of receiving treatment given a set of observed covariates – to individuals in both treatment and control groups, they ultimately become more comparable and thus more feasible from an empirical standpoint. Imbens (2004) thereafter made a significant contribution to the literature on propensity score matching, as his findings suggested that, if unconfoundedness (i.e., conditional independence) holds, conditioning solely on the propensity score ensures independence of actual treatment and the potential outcome of non-treatment. The biases between treatment and control groups are thus removed, allowing for higher accuracy in the interpretation of results (Imbens, 2004).

A wide range of propensity-score-matching algorithms have been developed since the seminal work of Rosenbaum and Rubin (1983). In the context of this thesis, three matching methods come particularly to mind: *nearest-neighbor (NN) matching*, *Kernel matching estimators,* and *coarsened exact matching*. Jaraité & Di Maria (2016) provide a neat overview of NN estimators and contrast their suitability with Kernel matching estimators in the context of carbon-emitting installations. The authors deem the NN estimator as the most straightforward matching estimator. In short, an installation from the control group is selected as the appropriate matching counterpart to a treated installation when it displays the closest propensity score. Jaraité & Di Maria (2016) allow for replacement in their NN matching estimation, which essentially implies that installations in the control group can be selected for a match more than once. This closely resembles full matching, used by Dechezleprêtre et al (2018), which allows for different numbers of untreated and treated installations in each matched grouping (Langworthy et al, 2023). NN matching with replacement encompasses a trade-off between bias and variance. It tends to reduce bias since it allows for more flexibility in selecting suitable matches for treated installations. However, this comes at the expense of increased variance because the matched groupings may significantly differ from one constellation to another (Jaraité & Di Maria, 2016; Dechezleprêtre et al, 2018). Kernel matching strategies construct matches using all individuals in the control sample but differentiates the control units through a weighting scheme. The assigned weights depend on the distance between the control group observation and the treated observation. Distant observations are downweighed while proximate observations are upweighted (Heckman et al, 1998; Jaraité & Di Maria, 2016). Similar to NN matching with replacement, Kernel matching also involves the trade-off between bias and variance. Its main advantage is the lower variance, given that it uses more information. However, this comes at the expense of increased bias – all observations are utilized, including those that are downright inappropriate matches (Jaraité & Di Maria, 2016).

Given the concerns of inherent differences between treated and untreated installations when assessing the impact of the MSR, the abovementioned empirical evidence provides serves as a useful guide for mitigating the potential biases associated with the methodological approach in this thesis.

4. Data

The following section describes the data used for this thesis, as well as the limitations associated with those data. All in all, three sources of data were utilized: Norway's Pollution Transfer and Release Register, the European Union Transaction Log, and Bureau van Dijk's Orbis Database.

4.1 Norway's Pollution Transfer and Release Register

This thesis looks at Norwegian installation-level, fossil carbon dioxide emissions, and thus makes use of panel data (installation-year) from Norway's Pollution Transfer and Release Register (PRTR). PRTR's were introduced by most European countries in the 1990s, mandating the monitoring of specific installation-level pollutants to air, water, and soil, encompassing a wide range of industrial activities. A Europe-wide register (E-PRTR) was later established in 2001, requiring large installations to report their releases of pollutants. This extensive register covers approximately 30,000 installations, annually reporting the releases of more than 90 pollutants, including greenhouse gases such as carbon dioxide.

Assessing the effectiveness of the MSR mitigating firm-level carbon emissions necessitates two things: data on emissions before and after the announcement of the MSR and data from both treated and untreated installations. The reason for why the E-PRTR is not suitable for this thesis is due to the high reporting threshold. Only installations that emit more than 100 kilotonnes of carbon dioxide per year are required to report to the E-PRTR. This basically means that all installations reporting to the E-PRTR are subject to the MSR, leaving a limited number of untreated installations.

Norway's PRTR constitutes a more suitable dataset given its lower reporting threshold. All Norwegian installations emitting more than 1 kilotonne of fossil carbon dioxide per year are required to report to Norway's PRTR, thus making the register a suitable panel dataset that comprises a sufficient number of both treated and untreated installations. In addition to fossil carbon dioxide emissions, the dataset also provides information on the economic sector and location of the installation which will be of use for the empirical analysis. The panel dataset is restricted to the time period 2013-2023. Given that we are interested in the MSR introduction in 2019, the selected time period gives us plenty of years to assess the before-and-after trends. After adjusting the dataset to the selected time period, the dataset comprised 222 installations, with 1669 installation-year observations.

The data points included in this dataset include installation-year observations of emissions, NACE-code, and county. The NACE-codes are transformed to the 2-digit level, and subsequently used to determine which sector the installation belongs to.

4.2 European Union Transaction Log

The European Union Transactions Log (EUTL) is thereafter used to determine the regulatory status of installations in the sample and thereafter cross-check with Norway's PRTR to assign installations into treatment and control groups, i.e., those that are subject to the MSR and those that are not. Serving as the European Commission's centralized carbon emissions inventory, it contains data on compliance status, verified emissions, as well as transactional data within the EU ETS. After adjusting for regulatory status, the dataset comprised of 44 treatment installations and 136 control installations.

In addition to determining regulatory status, the EUTL was also used to obtain data on allocation of emissions allowances and allowance transactions. These data points will serve as useful covariates in the empirical analysis as they provide valuable information on firm production and trading engagement.

4.3 Bureau van Dijk's Orbis Database

Bureau van Dijk's Orbis database is used to obtain financial data on operational revenues. Given the tight association between emissions and economic activity, this data is essential to control for business cycles and overall fluctuations in the economy when looking at emissions trends. Data on operational revenues can also be used for constructing an estimate of emissions intensities. An ideal measure of emissions intensity should be calculated as emissions per unit of output produced (e.g., emissions per ton of aluminum produced). Emissions per unit of operational revenue constitutes a comparatively weak measure given its sensitivity to fluctuations in the output price.

Another significant source of impreciseness in the data obtained from Orbis is rooted in the fact that it solely includes financial information on the parent companies. The sample installations in this study, reporting emissions to the Norwegian PRTR, are subsidiaries. This essentially means that there is no financial data for them in Orbis. The financial data from parent companies, such as operational revenues and number of employees, do not necessarily tell us about the economic activity of the specific installations studied in the sample. The number of employees, transposed into person-hours, is also a sub-optimal measure due to the fluctuations in installation-level labor intensities over the estimation period.

4.4 Summary of Finalized Dataset

Table 3 provides an overview of the observations across the designated period of analysis. As illustrated by the table below, the panel dataset does not exhibit perfect balance as installations began reporting their emissions into the Norwegian PRTR at different points in time. Additionally, the panel data on operational revenues obtained from Orbis also lacks perfect balance.

Table 3. Summary of Observations by Year.

Year	MSR Participation		
	Control	**Treated**	**Total**
2013	115	44	159
2014	123	44	167
2015	132	44	176
2016	136	44	180
2017	132	44	176
2018	127	44	171
2019	120	43	163
2020	118	43	161
2021	115	43	158
2022	113	42	155
2023	3	0	3
Total	1234	435	1669

All in all, the finalized dataset comprises a total of 1669 observations: 1234 observations belong to the control group, while 435 observations belong to the treatment group. Also worth investigating is the sectoral distribution across the non-MSR and MSR installations. Table 4 depicts this sectoral distribution, indicating that a majority of the observations, regardless of treatment status, belong to the manufacturing sector. Table 4 also shows the total number of unique installations included in the finalized dataset amounts to 180 installations: 136 installations belong to the control group while 44 installations belong to the treatment group.

Table 4. Sectoral Distribution by Treatment Status

	MSR Participation		
Sectors	**Control**	**Treated**	**Total**
Electricity, gas, steam, and air conditioning supply	6	2	8

Manufacturing	117	36	153
Mining and quarrying	6	3	9
Water supply; sewerage, waste management and remediation activities	7	3	10
Total	136	44	180

Lastly, we are interested in examining at the difference across MSR and non-MSR installations in key outcome variables. Table 5 provides an overview of these differences. Given that this thesis utilizes difference-in-differences (DiD), installations are allowed to differ from one another, conditioned on these differences being constant across time. Section 5 provides further elaboration on this.

Table 5. Summary Statistics of Key Variables.

	N	Mean	SD	Min	Max
Non-MSR Installations					
Emissions (ktCO2)	1214	4733.4	12903.575	0	120000
Operational Revenues (€M)	1081	536.474	906.291	0	3938.224
Emissions Intensity (ktCO2/€M)	1058	1890558.7	58179246.764	0	1.891e+09
Ln (Emissions)	1092	6.678	2.091	-1.204	11.695
Ln (Operational Revenues)	1077	4.527	2.280	-16.162	8.278
Ln (Emissions Intensity)	960	2.108	2.866	-6.086	21.36
MSR Installations					
Emissions (ktCO2)	434	31321.916	37044.336	0	187700
Operational Revenues (€M)	402	3742.339	15995.316	0	141389.53
Emissions Intensity (ktCO2/€M)	396	1030.375	3750.750	0	32101.704
Ln (Emissions)	417	9.514	1.728	1.902	12.143
Ln (Operational Revenues)	397	5.013	2.063	.695	11.859
Ln (Emissions Intensity)	379	4.404	2.498	-3.986	10.377

5. Method

The following section describes the underlying method employed to assess the impact of the MSR on installation-level emissions. This thesis takes a quantitative approach to assessing the link between the MSR and installation-level emissions. Specifically, this study is conducted through a difference-in-differences (DiD) analysis paired with matching and fixed effects in order to mitigate the biases associated with comparing emissions of two rather heterogeneous groups. The DiD analysis focuses on Norwegian installation-level emissions between 2013-2023, providing plenty of informative installation-year observations before and after the MSR began operating in 2019.

5.1 Theoretical/Conceptual Framework

The empirical analysis is guided and facilitated by the theoretical framework laid out by Hicks (1932), who developed the notion of induced innovation. At a glance, the theory is quite simple – a change in relative factor prices yields innovations that mitigate the need for those relatively expensive factors of production. With Newell et al's (1999) augmentation of the induced innovation theory – endogenizing the importance of government regulation in inducing innovation – we are able to extrapolate the theoretical frameworks laid out by the scholarly works of Hicks (1932) and Newell et al (1999) and apply it to the context of the MSR, allowing us to generate a hypothesis based on the following mechanism: the MSR stabilizes the uphill price trajectory of emissions allowances, inducing a change in relative factor prices in production which subsequently incentivizes firms to adopt innovative, low-carbon technologies that become less costly than the conventional, emissions-intensive technologies. This will in turn bring down the emissions and emissions intensities of installations which are subject to this policy.

This thesis also draws upon another strand of literature concerning the environmental proactiveness among firms. With the MSR taking effect in 2019, and subsequent cancellation commencing in 2023, insights form the literature on firm environmental proactiveness could offer valuable explanations for strategic responses to the MSR. Specifically, this study will incorporate the resource-based view of the firm, as outlined by Hart (1995) and expanded upon by Buysse & Verbeke (2003), to facilitate the interpretation of the findings obtained in the empirical analysis.

5.2 Difference-in-Differences

A difference-in-differences model is utilized to assess the impact of the MSR on installation-level emissions. This method constitutes a nonexperimental technique to estimate the average treatment effect of the MSR on the treated, by comparing the difference across time in the differences between emissions means in the treatment and control group. The DiD technique controls for unobservable time and idiosyncratic characteristics between treated and untreated installations.

An installation can be subject to one out of two regulatory scenarios: either it is regulated under the ETS and thus subject to the MSR, or it is excluded from the EU ETS and thus exempt from the MSR. D denotes the treatment indicator, determining the installation's MSR participation status. When D_i = 1, it implies that the i^{th} installation is subject to the MSR. An installation in the control group is assumed to be unaffected by the policy and is thereby denoted as $D_i = 0$. Potential outcomes for firm i and time t, conditional on MSR participation and non-MSR participation, are denoted as $Y_{it}(1)$ and $Y_{it}(0)$. The average treatment effect on the treated can be derived from the following equation:

$$\alpha_{TT} = E[Y_{it}(1) - Y_{it}(0) \mid D_i = 1]$$

where t is any year after the implementation of the MSR, i.e., after 2019, and α_{TT} represents the average treatment effect of the MSR on annual installation-level CO_2 emissions. Installation-level CO2 emissions post-2019 can be used to identify $E[Y_{it}(0) \mid D_i = 1]$. The issue is that we cannot observe $E[Y_{it}(0) \mid D_i = 1]$, i.e., we cannot observe the emissions trajectory of MSR firms had they not been subject to the policy. I exploit the unique design features of the EU ETS, which solely regulates a subset of the largest emitters in the EU (and Norway, Iceland, and Liechtenstein). MSR participation is thus incomplete, yielding a potential comparison group that can be used to econometrically estimate the counterfactual scenario $E[Y_{it}(0) \mid D_i = 1]$. One of the simplest estimates of α_{TT} can be obtained using a standard DiD estimator. The standard DiD estimator allows us to compare two heterogeneous sets of installations, based on the assumption that in absence of treatment, the difference between treatment and control installations is constant over time. This is known as the parallel trends assumption. The installations are allowed to vary in the level of outcomes, as long as the difference between those outcomes is constant. Table 5 in the previous section illustrates the differences between MSR installations and non-MSR installations (see Appendix A for visual representations).

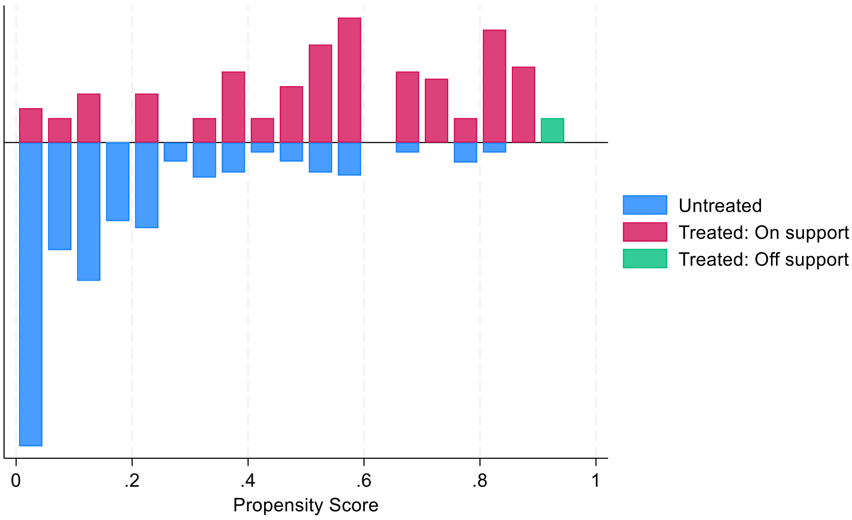
As long as the parallel trends assumption holds, a standard DiD estimator is feasible. However, there valid reasons to be concerned with bias if the variables associated with installation-level outcomes vary significantly across treatment and control groups. To mitigate this bias, I exploit the observable differences between MSR and non-MSR installations to construct *semi-parametric matching estimators*. The underlying estimation strategy is based on the work of Heckman, Ichimura, and Todd (1997) and Heckman et al (1998), who developed the following DiD matching estimator:

$$\theta_{DIDmatch} = \frac{1}{N_1} \sum_{j \in I_1} \left\{ (Y_{j1}(1) - Y_{j0}(0)) - \sum_{k \in I_0} w_{jk} (Y_{k1}(0) - Y_{k0}(0)) \right\}$$

where I_1 represents the set of MSR installations, I_0 represents the set of non-MSR installations, and N_1 represents the number of MSR installations. MSR installations are indexed by *j* and the non-MSR installations are indexed by *k*. When constructing the counterfactual observation for MSR installation *j*, each control installation *k* is weighted by w_{jk}. The fundamental purpose and function of w_{jk} is to weigh the contribution of each non-MSR installation. If a non-MSR installation exhibits similar observable characteristics X_i to those of MSR installations, their contribution as a counterfactual is assigned a heavier weight.

The statistical approach to assigning weights varies between different matching estimators. For this thesis, I draw upon the work of Rosenbaum & Rubin (1983) who developed the notion of *propensity score matching.* Numerous propensity-score-matching algorithms are available – two of the most prominent being nearest neighbor (NN) matching and Kernel matching. Given the relatively limited size of my dataset, Kernel matching was deemed as the most suitable approach as it utilizes all observations in the analysis. Each non-MSR installation is assigned a Kernel weight that reflects its similarity with MSR installations, conditioned on a set of observable covariates. When determining the set of covariates on which we match installations on, we must turn to the inclusion criteria of the EU ETS. The EU ETS inclusion criteria are equivalent to the MSR participation criteria. The inclusion criteria are based on installation-level capacity, either in terms of energy or production (Jaraité & Di Maria, 2016). An ideal dataset would include data on installation-level capacity and output. Unfortunately, these characteristics are unobservable, exacerbating the need for a proxy. I follow a similar approach to that of Dechezleprêtre et al (2018), using pre-policy emissions as a proxy for capacity and output. Altogether, I match on the log of average pre-MSR emissions and pre-MSR emission growth rate. Figure 7 shows the distribution of propensity scores for MSR installations and non-MSR installations, illustrating overlap to a certain degree. These propensity scores reflect the probability of being subject to the MSR based on the log of average pre-MSR emissions and pre-MSR emissions growth rates.

Figure 7. Distribution of Propensity Scores.



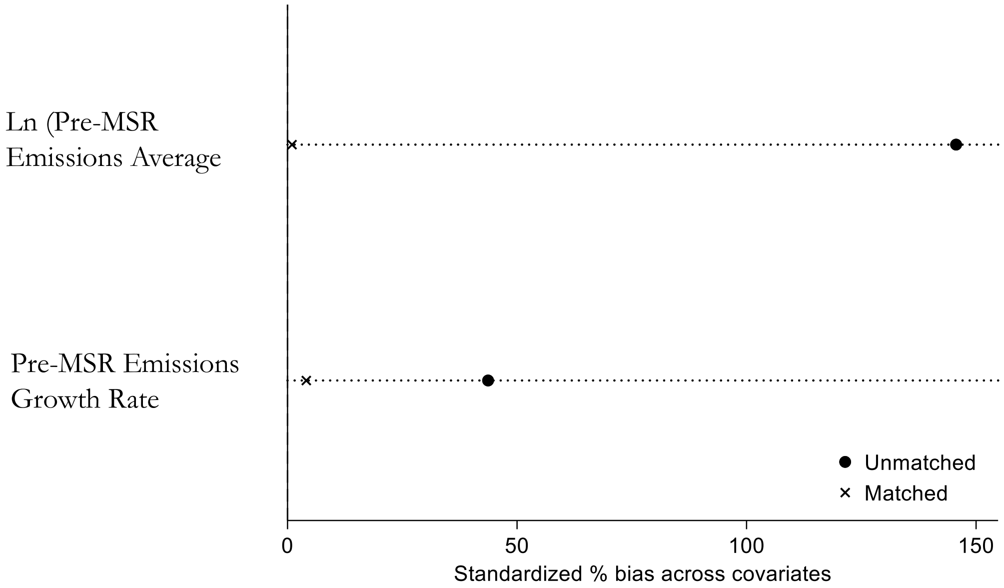
Kernel matching is conducted to mitigate the underlying bias associated with these pre-MSR emissions trends. By matching MSR installations and non-MSR installations based on these inclusion criteria proxies, we attempt to reduce the bias between treated and untreated installations in hopes of ensuring quasi-randomness in the assignment to the policy treatment. Table 6 displays a t-test indicating that post matching, the difference between MSR and non-MSR installations conditioned on the pre-MSR emissions covariates, is statistically insignificant.

Table 6. T-Tests from Kernel Matching on Pre-MSR Covariates.

Variable	**Unmatched Matched**	**Mean**		**% Reduction**		**t-test**	
		Treated	Control	% Bias	|Bias|	t	p>t
Ln (Pre-MSR Emissions Average)	U	9.511	6.902	151.20		24.21	0.000
	M	9.444	9.424	1.2	99.2	0.18	0.860
Pre-MSR Emissions Growth Rate	U	-0.018	-0.118	38.60		5.79	0.000
	M	-0.020	-0.031	4	89.7	0.70	0.487

Figure 8 illustrates the results listed in Table 6, providing a comparison between the biases of the unmatched and matched sample. As can be seen in the figure, bias is greatly reduced with Kernel matching. The constructed semi-parametric matching estimator serves the following key statistical purpose: conditional on the aforementioned pre-MSR covariates on which we match treatment and control observations, the parallel trends assumption is likely to hold (see Appendix C for visual representation).

Figure 8. Bias Reduction After Kernel Matching.



Although the Kernel matching seems to have significantly reduced the standardized bias across the aforementioned covariates, concerns about unobserved heterogeneity remain. Unfortunately, Norway's Pollution Transfer and Release Register (PRTR) lacks data on installation-level confounding economic variables (e.g., profits, revenues, employment, etc.) or the general economic circumstances (e.g., demand conditions, costs of input, gross regional product, etc.). The DiD matching strategy ensures that installations are similar in terms of pre-MSR emissions, the abovementioned economic differences persist. I mitigate this additional omitted variable bias by employing installation- and time-fixed effects, enabling me to control for the unobserved time-invariant idiosyncratic differences between MSR and non-MSR installations. The installation-fixed effects control for key factors such as sector, management quality, location-specific regulation, etc. The main advantage of employing time-fixed effects, in the context of this thesis, is that it accounts for the substantial impacts of the COVID-19 pandemic.

6. Empirical Analysis

6.1 Results

The following section describes and analyzes the findings of the study, with the aim to gain valuable insights into the impact of the MSR on installation-level fossil emissions and emissions intensities in Norway. To shed light on this pattern, I compare the emissions and emissions intensities before and after the implementation of the MSR through a regression analysis. I estimate the percentage change in emissions and emissions intensities that can be attributed to the MSR utilizing a difference-in-differences (DiD) estimation strategy, weighted by Kernel weights obtained from propensity score matching (PSM). Installation- and time-fixed effects estimators are also included in the model.

6.1.1 Impact on emissions

Table 7 reports the estimated coefficients illustrating the effect of the MSR on fossil emissions at the installation level. In line with our DiD strategy, we are mostly interested in the *Treatment x Post* coefficient, which constitutes the average treatment effect of the treated (ATT).

Table 7. Impact on MSR Installation-Level Emissions.

	(1) Ln (Emissions)	(2) Ln (Emissions)	(3) Ln (Emissions)	(4) Ln (Emissions)	(5) Ln (Emissions)
MSR Treatment	2.836*** (.318)				
Post-MSR	-.132** (.063)	-.008 (.037)	-.008 (.037)	-.129 (.161)	-.129 (.161)
Treatment x Post	.071 (.108)	-.038 (.098)	-.038 (.098)	-.137 (.145)	-.137 (.145)
Kernel Matching	No	Yes	Yes	Yes	Yes
Time-Variant Controls	No	Yes	Yes	Yes	Yes
Installation Fixed Effects	No	No	Yes	No	Yes
Year Fixed Effects	No	No	No	Yes	Yes
Constant	6.57*** (.171)	7.893*** (.917)	7.893*** (.917)	7.741*** (1.079)	7.741*** (1.079)
Observations	1509	242	242	242	242
R-squared	.z	.023	.023	.08	.08

Robust standard errors are in parentheses
**** p<.01, ** p<.05, * p<.1*

Table 7 comprises a variety of specifications. Column 1 shows the output from a simple, unrestricted DiD regression devoid of controls, matching, or fixed effects. The positive coefficient suggests an increase of emissions from MSR installations, although not statistically significant.

Moving to Column 2, I employ the DiD Kernel matching estimator, matching MSR and non-MSR installations based on their pre-MSR emissions trends. Additionally, time-variant controls (allocation and operational revenue) are incorporated into this model. The negative coefficient suggests a reduction in emissions following the MSR, albeit statistically insignificant.

Column 3 introduces an installation fixed effects estimator to account for any other economic variables distinguishing treated from untreated installations. Notably, the coefficient and standard error remain consistent, suggesting a statistically insignificant reduction of emissions from MSR installations.

In column 4, installation fixed effects are replaced with year fixed effects to account for fluctuations the general economic environment over time, affecting both treated and untreated installations. The adjustment yields a larger negative effect on emissions from MSR installations yet remains statistically insignificant.

Lastly, column 5 incorporates both installation- and time-fixed effects, controlling for unobserved heterogeneity over time and space. Despite this additional control, the coefficient remains consistent with column 4, indicating a negative effect without statistical significance.

6.1.2 Impact on emissions intensity

These absolute measures of emissions analyzed in the section above, while intriguing from the perspective of the MSR's environmental integrity, do not enable discrimination between changes in production levels and other adjustments made at the installation level, e.g., modifying the fuel mix or production technologies. Therefore, it is important we shift focus towards emissions intensity to gain insight into these crucial factors and the role they play in the climate transition.

Table 8 reports the estimated coefficients illustrating the effect of the MSR on fossil emissions at the installation level. Similar to the table above, the key coefficient is *Treatment x Post* as it denotes the ATT. The variety of specifications are the same as those delineated in Table 7. However, in contrast to the findings presented in Table 7, the results in Table 8 below exhibit somewhat more promise.

Table 8. Impact on Installation-Level Emissions Intensities.

	(1) Ln (Emissions Intensity)	(2) Ln (Emissions Intensity)	(3) Ln (Emissions Intensity)	(4) Ln (Emissions Intensity)	(5) Ln (Emissions Intensity)
MSR Treatment	2.214***				
	(.478)				
Post-MSR	-.248***	.162***	.162***	.042	.042
	(.068)	(.009)	(.009)	(.178)	(.178)
Treatment x Post	.053	-.196*	-.196*	-.252	-.252
	(.127)	(.106)	(.106)	(.163)	(.163)
Kernel Matching	No	Yes	Yes	Yes	Yes
Time-Variant Controls	No	Yes	Yes	Yes	Yes
Installation Fixed Effects	No	No	Yes	No	Yes
Year Fixed Effects	No	No	No	Yes	Yes
Constant	2.254***	4.193***	4.193***	3.918***	3.918***
	(.285)	(.56)	(.56)	(.933)	(.933)
Observations	1339	242	242	242	242
R-squared	.z	.001	.001	.05	.05

Robust standard errors are in parentheses
**** p<.01, ** p<.05, * p<.1*

Column 1 displays the results from the unrestricted DiD regression lacking controls, matching, and fixed effects. The substantial systematic differences between MSR installations and non-MSR installations leaves little room for anticipation when regressing the full sample. The positive coefficient indicates a marginal, yet statistically insignificant increase in emissions among MSR installations.

Moving onto column 2, utilizing the Kernel DiD matching estimator along with adding time-variant controls, we detect a switch in the coefficient from positive to negative. This coefficient can be interpreted in the following way: being subject to the MSR is associated with a 19.6% reduction in emissions intensity, all else constant. This negative effect is statistically significant at the 10% level.

Column 3 enhances the model by introducing installation fixed effects, enabling us to control for time-invariant idiosyncratic effects. Findings remain consistent after the adjustment, suggesting a 19.2% emissions intensity reduction associated with the MSR, all else constant.

Replacing installation fixed effects with year fixed effects in column 4 amplifies the effect but compromises its statistical significance, with the p-value rising to 0.13. Column 5 combines both installation and year fixed effects, yielding a coefficient and standard error consistent with those in column 4.

6.2 Coarsened Exact Matching

As an alternative matching method, coarsened exact matching (CEM) was applied to check whether the estimation output turned out any different to those using Kernel matching. CEM is a monotonic matching method that mitigates imbalance between the treated and control installations. The idea behind CEM is to temporarily coarsen each covariate – in our case pre-MSR emissions growth rate and pre-MSR emissions average – into genuinely meaningful groups, thereafter exact matching on these coarsened covariates, and ultimately retaining solely the non-coarsened original values of the matched covariates. One of the main benefits of CEM lies in its property of monotonic imbalance bounding, essentially meaning that maximum imbalance is bounded to the extent the user sees fit (Blackwell et al, 2009). CEM was applied to the DiD estimation of emissions intensities. However, statistical significance is absent across all models (see Appendix D).

7. Discussion

The following section aims to discuss the main findings of this thesis. It is divided into two parts: the first part discusses the results regarding the MSR's impact on installation level emissions; the second part discusses the impact on emissions intensity. The discussion touches upon the absence of statistical significance in the results, as well as the direction of the treatment effect, drawing upon the literature synthesized in previous sections.

7.1 Installation-Level Emissions

7.1.1 Absence of statistical significance

While Table 7 reveals an observed effect, its lack of statistical significance must be addressed. The insignificance could be due to a multitude of factors. First, there are legitimate reasons to be concerned with the outcome variable in this model. Although the main objective of the MSR (and the EU ETS as a whole) is the reduction in *absolute* greenhouse gas emissions, absolute measures may not reflect changes in emissions relative to production or economic output. Even with operational revenue controlled for, unobserved time-varying installation-level covariates may influence absolute emissions. While installation- and year-fixed effects estimators help mitigate omitted variable bias, factors such as relative energy prices or the carbon tax levied on each installation remain significant drivers of absolute emissions (Klemetsen et al, 2016).

Previous studies investigating the impact of market-based policies on emissions have typically used substantially larger datasets by aggregating Pollution Release and Transfer Registers (PRTR) from multiple countries. Unfortunately, this laid beyond the scope of this thesis, resulting in a dataset of limited size and sample. A larger dataset might have produced more statistically significant findings.

Additionally, the lack of significance and modest magnitudes of the coefficients in Table 7 may be attributed to the time required for adjusting to a new regulatory regime. The price developments of emissions allowances since the implementation of the MSR has been unprecedented, providing regulated firms with a price signal unlike any they have encountered earlier. But decisions regarding levels of production and subsequent investments into low-carbon technologies take time (Rosendahl, 2019). Moreover, there is the crucial aspect of *gradual learning* – firms successfully assimilate to the new regime, gradually learning how to reduce their emissions in a cost-efficient manner (Klemetsen et al, 2016; Jaraité & Di Maria, 2016). This does not happen overnight, which could explain the absence of any statistically significant treatment effects. Liliestam et al (2020) highlight the importance of long-term carbon pricing effects when assessing market-based climate policies – the rate of technological progress and the change rate of emission reductions is more relevant than immediate emissions levels. Instead of solely focusing on immediate emissions levels, it might be worth shifting the focus towards the MSR's impact on investments into low-carbon technologies or green patents that indirectly reduces emissions.

Moreover, we can further contextualize the results by revisiting the existing literature on the MSR. The absence of statistical significance in the models of Table 7 is in line with the cautious sentiments expressed by numerous researchers regarding the MSR's effectiveness in mitigating emissions (Rosendahl, 2019; Gerlagh et al, 2020; Perino, 2018). The insignificant MSR treatment effect could potentially be attributed to the anticipatory behavior among MSR installations. Rosendahl (2019) brings up an intriguing scenario: suppose additional external abatement policies will be introduced in the future after the MSR ceases to absorb allowances. Demand for allowances during those years will decline, leading to a simultaneous decline in allowances prices. As market participants anticipate this, banking suddenly becomes less profitable. Less banking translates to fewer allowances being absorbed by the MSR, resulting in fewer allowances being cancelled. Rosendahl (2019) thus highlights the ambiguity of the policy instrument, given its susceptibility to speculatory behavior. Another potential explanation for the lack of significant differences between MSR and non-MSR installations could attributed to the idiosyncratic discount rates applied by the firms, as discussed by Osorio et al (2019). Perhaps firms value the future less, thus applying a higher discount rate to their operations, and subsequently bank fewer emissions allowances. Further research into the behavioral aspects could therefore facilitate the design and evaluation of the MSR.

Lastly, let us return to Figure 5 in Section 2.4.1, illustrating Norway's significant lead in decarbonization efforts compared to its counterparts in the EU ETS. The graph highlights a notable bias when focusing on Norwegian installations, given that its power sector relies almost entirely on renewables (IEA, n.d.). In contrast, the average EU ETS emissions of other participating countries are considerably higher than those of Norway. This suggests that an analysis of Norway – chosen due to data availability constraints – may not be the most suitable candidate for this thesis. Examining the impact of the MSR in a different context could potentially yield more significant results.

7.1.2 Direction of treatment effect

Although none of the models exhibit statistical significance, the direction of the effect is still worthy of discussion. The basic, unrestricted DiD estimation in column 1 of Table 7 suggests that the MSR led to an increase in emissions for those subject to the policy. This counterintuitive result is in line with the arguments presented by Jaraité & Di Maria (2016), who deemed the basic DiD model impractical due to the systematic differences between firm-level outcome variables. Even though the unique features of the MSR can be exploited for econometric estimation of its impact on installation-level emissions, we must acknowledge the shortcomings of comparing apples to oranges. MSR installations are typically larger and engage in economic activities associated with substantially heavier levels of emissions and production. The subsequent models in columns 2-5 employ matching methods based on covariates determining selection into treatment, all displaying negative coefficients. The direction of coefficients in columns 2-5 in Table 7 suggest that being subject to the MSR is associated with a reduction in absolute emissions.

Although not significant, the negative impact is consistent with the hypothesis derived from the theoretical frameworks laid out by Hicks (1932) and Newell et al (1999). The price stabilization feature of the MSR ensures that fossil-fuel-dependent factors of production become relatively costlier, incentivizing firms to adopt comparatively cheaper low-carbon technologies emitting less fossil carbon dioxide emissions. Furthermore, we can draw upon the conceptual framework of firm proactiveness as defined by Hart (1995) and Buysse & Verbeke (2003). The decline in emissions among MSR installations may reflect an anticipatory response to the cancellation mechanism set to take effect in 2023. Hart's (1995) theory of firms gaining competitive advantage through pollution prevention, coupled with Buysse & Verbeke's (2003) emphasis on green technology investments, collectively offer a neat explanation for the emissions reductions observed among MSR installations. Further research on the MSR's impact on installation-level investments into low-carbon technologies could validate this notion of firm proactiveness and resource-based view of the firm as outlined by Hart (1995) and Buysse & Verbeke (2003). As mentioned earlier, solely focusing on emissions as an outcome aggravates the ability to pinpoint the exact mechanism at play.

7.2 Installation-Level Emissions Intensity

7.2.1 Absence of statistical significance

While the majority of models in Table 8 do not reach statistical significance, certain features leave room for optimism. Columns 2 and 3, using illustrating the output from Kernel-weighted DiD regressions paired with installation fixed effects, indicate that being subject to the MSR is associated with a 19.6% reduction in emissions intensity, all else constant. This result is statistically significant at the 10% level. Although moderately significant, the evidence of the MSR having any meaningful impact on installation-level emissions intensities is modest. Despite presenting somewhat more promising results, Table 8 informs us that the emissions intensity changes among MSR installations were for the most part not significantly different from the behavior of non-MSR installation. The lack of significant results can be attributed to a wide array of factors.

Let's start by examining the outcome variable – emissions intensity, derived from two factors: annual fossil carbon emissions in kilotonnes and annual operational revenue in millions of euros. The data on operational revenues is inherently flawed as it solely reflects revenues generated by the parent company, while most installations reporting emissions into the Norwegian PRTR are subsidiaries. This limitation creates ambiguity in our emissions intensity measure. For instance, if an installation's emissions are increasing faster than its operational revenues, indicating a rise in emissions intensity, the parent company might report increased revenue attributed to other subsidiaries. In our case, the emissions intensity of the installation would drop since we only have financial data from the parent company. The informativity of our measure is thus ambiguous, which might explain the lack of statistical significance observed in the model. An alternative measure would be to use the number of employees translated into person-hours. However, this measure also presents challenges as it too reflects the person-hours of the parent company, offering limited insight into the subsidiaries themselves. As previously mentioned, there is considerable bias associated with analyzing the context of Norway, given that it has already progressed significantly in decarbonizing its economy (IEA, n.d.).

7.2.2 Direction of treatment effect

Moving on to the direction of the average treatment effect on MSR installations. The trends we see across the models of Table 8 closely resemble those detected in Table 7. Column 1 displays a positive coefficient of modest magnitude, indicating an increase in emissions intensity among MSR installations. However, this estimation is deemed invalid due to selection bias stemming from the systematic differences between MSR and non-MSR installations. The remaining models, specified in columns 2-5, display negative coefficients of larger magnitude than the absolute emissions estimations illustrated in Table 7. Worth noting is that intensity-based measures do not necessarily ensure reductions in absolute emissions. The larger magnitudes observed may reflect a rapid expansion of operations rather than actual abatement efforts. Furthermore, emissions intensities calculated as emissions relative to operational revenue are sensitive to changes in output prices (Klemetsen et al, 2016). The treatment effect we observe could simply reflect an increase in the prices of the products produced.

Contextualizing these results within the theoretical frameworks outlined by Hicks (1932) and Newell et al (1999), leads us to hypothesize that firms may have responded to the price signals induced by the MSR and subsequent cancellation mechanism. As illustrated in Figure 2 in Section 2, the price of emissions allowances saw an exponential increase following the MSR's introduction in 2019, translating into costlier fossil-fuel dependent factors of production. According to Hick's (1932) theory of induced innovation, firms are inclined to shift towards less costly factors of production emitting less fossil carbon dioxide. This Hicksian notion is supported and augmented by Newell et al's (1999) framework, suggesting that government regulation, such as the MSR, substantially impacts firms' energy efficiency. Given their emphasis on energy efficiency, Newell et al's (1999) theoretical framework plays a pivotal role in explaining the emissions intensity estimations.

Moreover, firms might have leveraged the price signals of the MSR to attain competitive advantage when the cancellation mechanism takes effect in 2023, as theorized by Hart (1995). By allocating investments towards low-carbon technologies and pollution-prevention measures, firms position themselves favorably as carbon dioxide emissions successfully constitutes a larger share of operational costs (Hart, 1995; Buysse & Verbeke, 2003). Resource-based firms develop strategic environmental capabilities to gain competitive advantage, demonstrating proactiveness through investments in green technologies, ultimately resulting in reductions in emissions intensities (Hart, 1995; Buysse & Verbeke, 2003).

8. Conclusion

We have reached a pivotal moment in the fight against climate change, necessitating well-designed market-based policies that effectively curb greenhouse gas emissions. The market stability reserve (MSR) is one of the more recent regulatory innovations in the sphere of climate policy, aimed at adjusting emissions allowances supply either by withholding them from auctioning, storing them, releasing them, or cancelling them later. By restoring balance to emissions trading, this policy aims to complement other climate regulations and provide a clear price signal for transitioning towards low-carbon technologies. Despite these promising features, previous research investigating the MSR has not yet reached consensus around its overall efficacy and potential to stabilize emissions allowance prices.

This thesis shed light on the MSR by asking the following question: what is the effect of the MSR on fossil carbon dioxide emissions and emissions intensities of regulated Norwegian installations? The question was answered by employing a difference-in-differences (DiD) estimation strategy. Due to legitimate concerns of selection bias, arising from the systematic differences between treated and untreated installations, sample installations were matched based on two pre-MSR covariates serving as proxies to MSR inclusion criteria: pre-MSR emissions growth rate and the log of pre-MSR emissions average.

My results indicate that the MSR is associated with modest reductions in emissions, albeit statistically insignificant. Factors such as the time lag of adjusting to new regulatory regimes, data constraints, the difficulty to isolate emissions variation, and Norway's already decarbonized economy may contribute to this insignificance. Emissions intensities seem to have been reduced to a larger extent, with some treatment effects displaying significance at the 10% level. The lack of statistical significance in the remaining emissions intensity model specifications could be attributed to factors similar to those explaining the lack of significance in the absolute emissions estimations. The intensity-based measure is imperfect due to emissions being reported at the installation level and operational revenues being reported by the parent company.

The direction of the treatment effect, although predominantly insignificant across model specifications, can be attributed to the theoretical frameworks proposed by Hicks (1932) and Newell et al (1999). Firms respond to the price signals induced by the MSR by favoring less-costly factors of production emitting less. They may also leverage the MSR to gain competitive advantage, as theorized by Hart (1995) and Buysse & Verbeke (2003), through pollution prevention strategies and investments into low-carbon technologies.

Further research is undoubtedly needed to understand the impact of the MSR and its accompanying cancellation mechanism. Since environmental strategies are typically designed for emissions reductions in the long term, it might be valuable to investigate the short-term effects of the MSR, like its impact on low-carbon investments or green patenting. Additionally, as installations select into MSR based on capacity, regression discontinuity would constitute an ideal method for estimating the effects of the MSR, assuming data on installation-level energy and production output is available. Lastly, applying this study to the context of a different, less decarbonized country may also offer valuable insight into the effectiveness of the MSR. If we are to achieve the ambitious climate targets that we have set for ourselves, it is imperative we gain a deeper understanding of the means designed to reach them.

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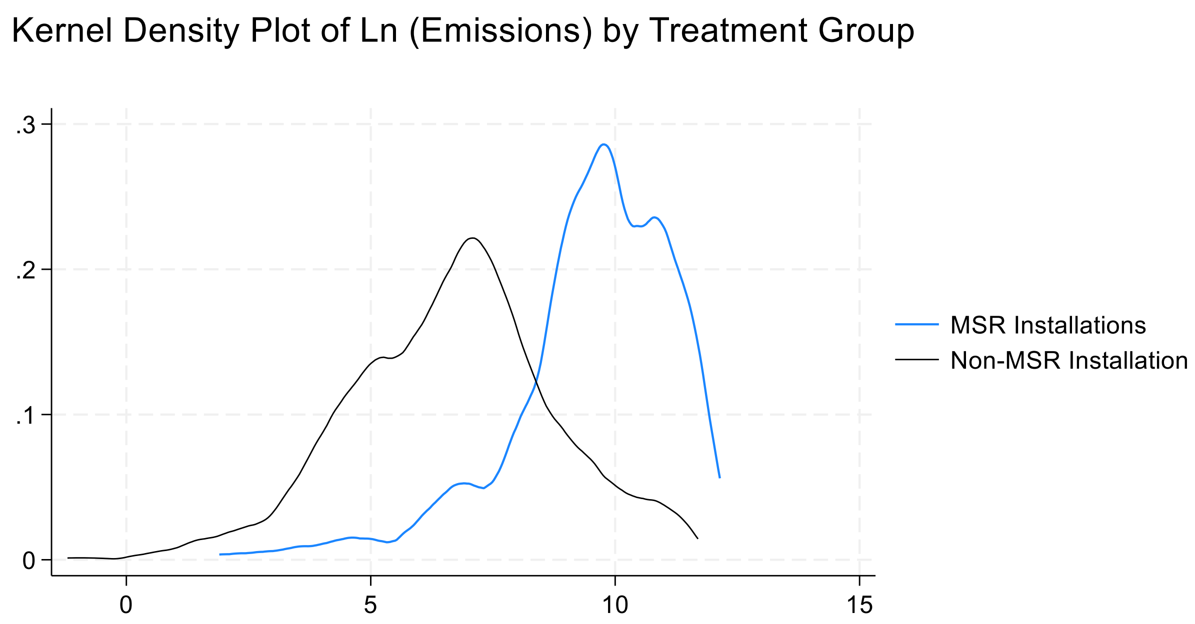
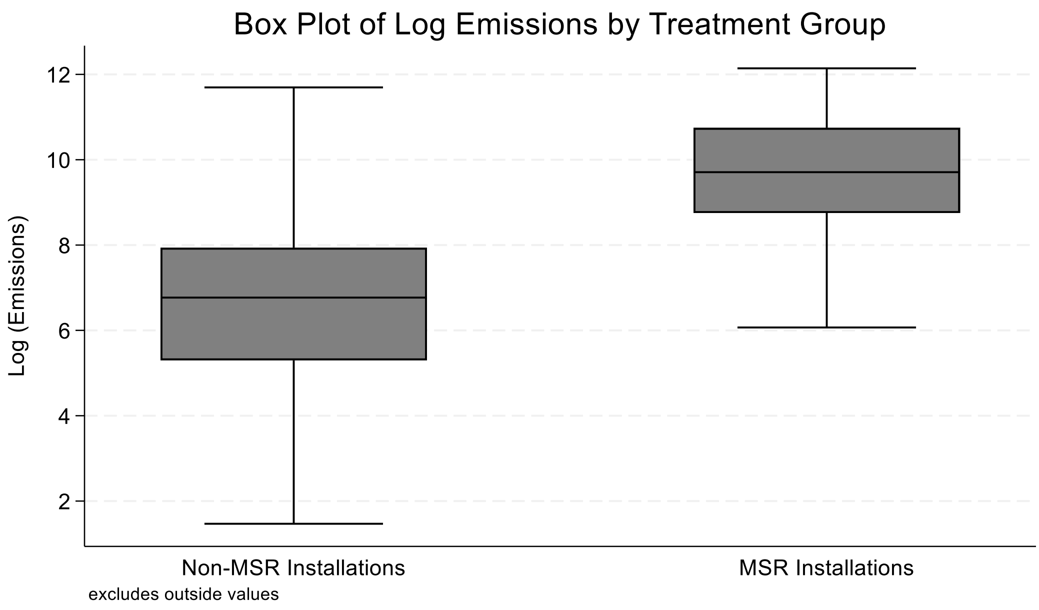
Appendix

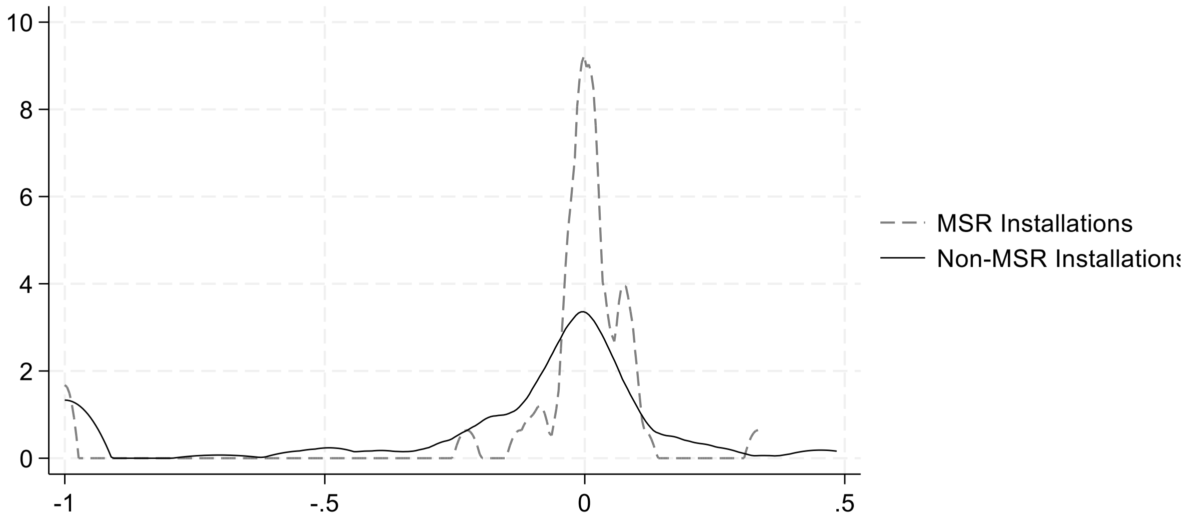
Appendix A – Differences in outcome variable by treatment group



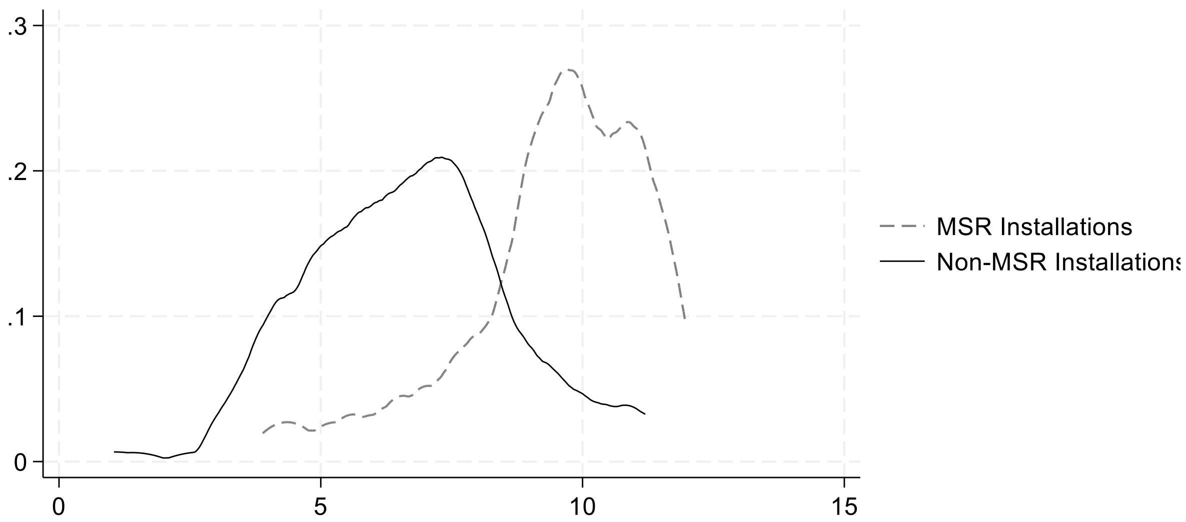


Appendix B – Common support

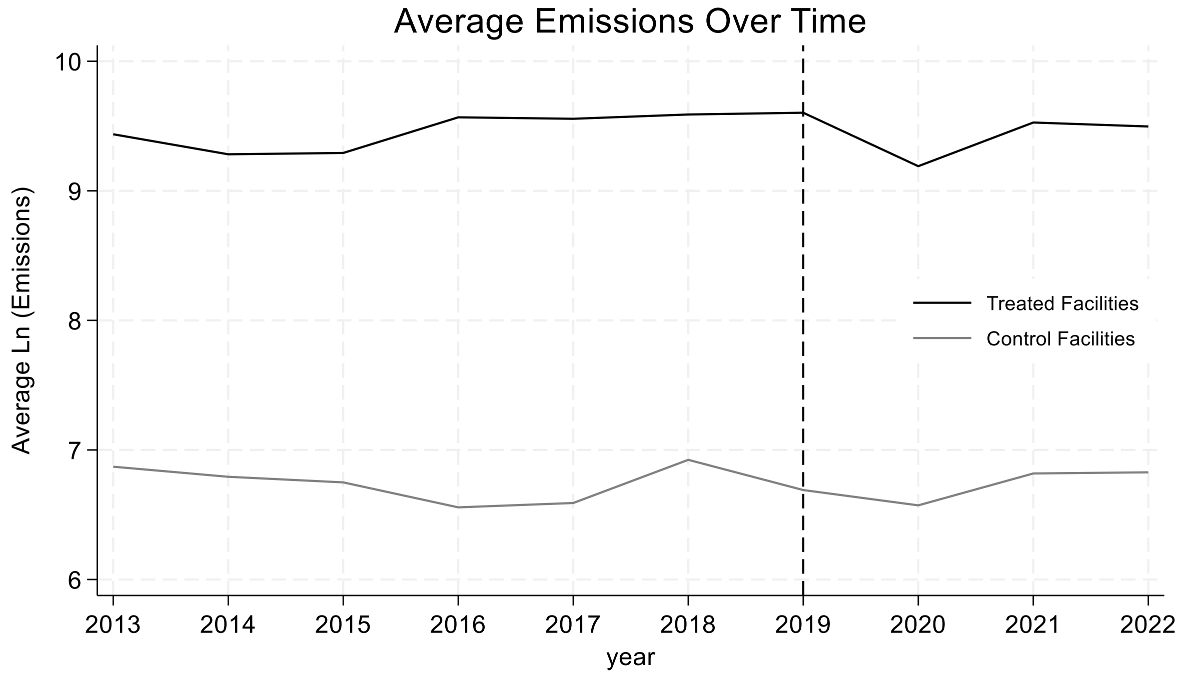
Kernel Density Plot of Pre-MSR Emissions Growth Rate



Kernel Density Plot of Ln (Pre-MSR Emissions Average)



Appendix C - Parallel trends



Appendix D – Coarsened Exact Matching

	(1) Ln (Emissions Intensity)	(2) Ln (Emissions Intensity)	(3) Ln (Emissions Intensity)	(4) Ln (Emissions Intensity)	(5) Ln (Emissions Intensity)
MSR Participation	2.214*** (.478)				
Post-MSR	-.248*** (.068)	-.167** (.068)	-.167** (.068)	-.386*** (.13)	-.386*** (.13)
Treatment x Post	.053 (.127)	-.043 (.13)	-.043 (.13)	-.042 (.131)	-.042 (.131)
CEM Matching	No	Yes	Yes	Yes	Yes
Installation Fixed Effects	No	No	Yes	No	Yes
Year Fixed Effects	No	No	No	Yes	Yes
Constant	2.254*** (.285)	4.674*** (.025)	4.674*** (.025)	4.704*** (.068)	4.704*** (.068)
Observations	1339	857	857	857	857
R-squared	.z	.035	.035	.07	.07

Robust standard errors are in parentheses
**** p<.01, ** p<.05, * p<.1*