

Market-based Emissions Regulation and the Evolution of Market Structure

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Abstract

We assess the long-run dynamic implications of market-based regulation of carbon dioxide emissions in the US Portland cement industry. We consider several policy designs, including mechanisms that partially offset the cost of compliance through rebating. Our results highlight two general countervailing market distortions that characterize many emissions-intensive, trade-exposed industries. First, echoing a point first made by [Buchanan \(1969\)](#), reductions in product market surplus due to market power counteract the social benefits of carbon abatement. Second, trade-exposed cement producers face competition from unregulated foreign competitors, leading to emissions “leakage” which offsets domestic emissions reductions. We find that a combination of these forces leads to social welfare *losses* for low social costs of carbon. At higher social costs of carbon, policies with production subsidies are efficient and welfare dominate more standard policy designs.

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1 Introduction

Concerns about the impacts of climate change are mounting. Rather than waiting for a coordinated global agreement to emerge, regional climate change policy initiatives are moving forward. Debates about how and when to implement these regional policies have been dominated by concerns about potentially adverse impacts on domestic industrial competitiveness, trade flows, and emissions leakage. Policy makers are looking to strike an appropriate balance between curbing domestic greenhouse gas (GHG) emissions and protecting the competitive position of the manufacturing sectors that would be subject to these regulations.

Market-based climate change policies, such as the European Union Emissions Trading Scheme (ETS) or California’s proposed GHG emissions trading program, are playing a crucial role in limiting GHG emissions. This is particularly relevant for large industrial sources. Traditionally, economic analysis of market-based emissions regulations has focused exclusively on perfectly competitive industries. In this “first-best” context, policy design is relatively straightforward. A Pigouvian tax, or an emissions trading program designed to equate marginal abatement costs with marginal damages, will generally achieve the socially optimal outcome. However, policy makers rarely, if ever, work in this first-best setting. Emissions intensive industries are generally characterized by pre-existing distortions and imperfections that complicate the design of efficient policy. This paper focuses on two distortions in particular.

First, the majority of emissions regulated under existing and planned emissions regulations come from industries that are highly concentrated.¹ In an imperfectly competitive industry, a first-best emissions policy that completely internalizes external damages will incentivize pollution abatement, but it will also exacerbate the pre-existing distortion associated with the exercise of market power. In a seminal paper, [Buchanan \(1969\)](#) asserts that the implementation of Pigouvian taxes should be limited to “situations of competition” because taxing an emissions externality further restricts already sub-optimal levels of output. In contrast, [Oates and Strassman \(1984\)](#) argue that the case for Pigouvian taxes “is not seriously compromised by likely deviations from competitive behavior” because the welfare gains from pollution control likely dwarf the potential losses from the various imperfections

¹Emissions from restructured electricity markets represent the majority of emissions currently targeted by existing cap-and-trade programs in the United States and Europe. Numerous studies provide empirical evidence of the exercise of market power in these industries, such as [Borenstein et al. \(2002\)](#); [Joskow and Kahn \(2002\)](#); [Wolfram \(1999\)](#); [Puller \(2007\)](#); [Sweeting \(2007\)](#); [Bushnell et al. \(2008\)](#). Other emissions intensive industries being targeted by regional emissions trading programs, such as cement and refining, are also highly concentrated.

in the economy.

In the context of global pollutants, such as greenhouse gases, a second consideration further complicates the welfare analysis of market-based emissions policy interventions. If emissions regulations apply to only a subset of the sources that contribute to the environmental problem, production in unregulated jurisdictions may displace production at regulated entities. This “emissions leakage” may substantially offset, or paradoxically even reverse, the reductions in emissions achieved in the regulated sector. Concerns about leakage and adverse competitiveness impacts have led to a series of policy proposals designed to penalize emissions while also providing incentives to mitigate adverse competitiveness impacts.

In this paper, we use the Markov-perfect Nash equilibrium (Maskin and Tirole, 1988; Ericson and Pakes, 1995) dynamic oligopoly framework developed in Ryan (2011) as the foundation for an analysis of market-based regulations limiting industrial emissions. Our approach allows us to assess the welfare implications of a market-based policy intervention in an industrial context characterized by both imperfect competition and exposure to competition from unregulated imports.

This paper analyzes the efficiency and distributional properties of proposed policies designed to reduce carbon dioxide emissions in the domestic Portland cement industry. For a number of reasons, this industry has been at the center of the debate about domestic climate change policy and international competitiveness. First, the industry is environmentally important: cement is one of the largest manufacturing sources of domestic carbon dioxide emissions (Kapur et al, 2009). Second, carbon regulation could result in major changes to the industry’s cost structure; complete internalization of the estimated social cost of carbon would increase average variable operating costs by more than 50 percent.² Third, the industry is highly concentrated in regionally-segregated markets, making it potentially susceptible to the Buchanan critique. Finally, import penetration in the domestic cement market has exceeded 20 percent in recent years, giving rise to concerns about the potential for emissions leakage (Van Oss, 2003 ENV; USGS Mineral Commodity Summary 2010). For these reasons, the cement industry is an interesting and important setting to study the complex interactions between industrial organization and environmental policy design.

A distinguishing feature of our analysis is our emphasis on industry dynamics. A static, short-run analysis is ill-suited to the domestic cement industry. Capital stock turnover is expected to play an essential role in improving the environmental performance of this

²On average, domestic cement producers emit approximately one ton of carbon for each ton of cement produced. Marginal costs of cement production are estimated to be in the range of \$30-\$40/ton (Ryan, 2011).

industry ([Worrell et al., 2001](#); [Sterner, 1990](#)). This is partly due to the limited opportunities to reduce carbon intensity through process changes and disembodied capital change, and partly due to fact that some very old and inefficient kilns are still in operation in the United States. Moreover, an exclusive focus on short run outcomes would likely fail to capture the extent of emissions leakage. Although leakage can manifest immediately as firms adjust variable input and output decisions such that less (more) stringently regulated production assets are used more (less) intensively, it can also occur gradually as firms accelerate the retirement of older production technologies in more stringently regulated jurisdictions and invest in new facilities and equipment in less stringently regulated jurisdictions. Static modeling cannot capture this second leakage channel.

Our analysis begins with the specification of a theoretical model of dynamic oligopoly in which strategic domestic cement producers compete in spatially-segregated regional markets. Some of these markets are trade exposed, whereas other landlocked markets are sheltered from foreign competition. Firms make entry, exit, and investment decisions in order to maximize their expected stream of profits conditional on the strategies of their rivals. Given capital investments, producers compete each period in homogeneous quantities. Regional market structures evolve as firms enter, exit, and adjust production capacities in response to changing market conditions.

Building on the parameter estimates from [Ryan \(2011\)](#), we then turn to our investigation of the static and dynamic implications of market-based emissions regulation design decisions. We use the econometrically estimated model to simulate industry response to a series of counterfactual emissions regulations. The basic intuition underlying our counterfactual simulations is quite simple. In the benchmark model that we estimate, emissions are unconstrained. Firms invest at the level where marginal costs equal expected marginal benefits subject to covering their fixed costs. The expected benefits are a function of the period payoffs. The market-based emissions regulations we consider affect firms' production and investment choices through changes in operating cost and revenue incentives. Importantly, we assume that cement producers' past response to changes in operating costs and revenues mimics what we would observe in response to policy-induced changes.

In addition to more standard carbon tax and emissions trading programs, we are interested in analyzing policy designs that incorporate both an emissions penalty (i.e. an obligation to pay a tax or hold a permit to offset emissions) and a production incentive in the form of a rebate. Under an emissions tax regime, tax revenues can be recycled (or rebated) to producers on the basis of lagged production. In the context of cap-and-trade

programs, dynamic permit allocation updating schemes make future free permit allocations contingent on a firm’s output or emissions shares in the previous period. In a first-best setting, these contingent rebates would undermine the efficiency of permit market outcomes because the implicit subsidy conferred by allocation updating encourages firms to increase output to economically inefficient levels (Bohringer and Lange, 2005; Sterner and Muller, 2008).³ However, in second-best settings, these rebates can be used to mitigate pre-existing distortions and regulatory imperfections (see, for example, Bernard et al. (2007)).

Given the uncertainty surrounding estimates of the social cost of carbon, we consider a range of carbon dioxide (CO₂) damages. We follow the lead of a landmark interagency process which recommends a range of social cost of carbon (SCC) values for use in policy analysis (Greenstone et al., 2011).⁴ In this working paper, we simulate outcomes for approximately half of the regional markets that comprise the industry. Future versions of the paper will include all domestic cement markets.

We find that the imposition of a carbon tax or emissions trading program that fully internalizes the social cost of carbon could have negative welfare impacts for SCC values at or below the central SCC value of \$21/ton. Two primary market forces drive this result. The first intuition follows Buchanan’s insights with regards to balancing distortions from market power against those induced by pollution externalities; the US Portland cement industry is highly concentrated. The second contributing factor stems from the incompleteness of the emissions regulation which creates the potential for emissions leakage.

As the assumed value of the negative emissions externality increases, the benefits from the emissions regulation (in the form of avoided damages from emissions) exceeds the costs, emissions leakage and the constriction of economic surplus notwithstanding. Notably, policy designs that couple a carbon tax with a production subsidy (in the form of a tax rebate or contingent permit allocation) welfare dominate more standard designs. The rebate works to mitigate leakage in trade exposed cement markets and the distortion associated with the exercise of market power.

This paper makes substantive contributions to three areas of the literature. First, this paper is germane to the literature that considers the dynamic efficiency properties of market-based emissions regulations. By their very nature, medium and long-run industrial responses

³Here, “first-best” refers to a regulatory environment in which the only market distortion or imperfection is the environmental externality that the emissions regulation is designed to internalize.

⁴The U.S. Government recently concluded a year-long process to estimate the monetized damages caused per ton of CO₂ emissions. For 2010, the central social cost of carbon (SCC) estimate is \$21, although sanctioned estimates range from approximately \$5 to \$65.

to policy interventions are difficult to pin down. During the time it takes for policy outcomes to manifest, a host of other potentially confounding factors and processes change and evolve. The conventional approach to analyzing these dynamic responses has been to use either highly stylized theoretical models (Conrad and Wang, 2003; Lee, 1999; Requate, 2005; Sengupta, 2010; Shafter, 1999) or large, numerical simulation models (Demailly and Quirion, 2007, Fischer and Fox, 2007; Jensen and Rasmussen, 2000; Szabo et al, 2006; US EPA, 1996).⁵ In a recent review of the literature, Millimet et al.(2009) draw attention to a “striking gap in the literature on environmental regulation,” noting that advances in structural estimation of dynamic models in the industrial organization literature should be brought to bear on long-run implications of environmental regulation. This paper starts to address this gap.

Second, this paper complements a growing body of work that examines the impacts of the EU Emissions Trading Scheme (ETS) on highly concentrated, trade-exposed, and emissions-intensive industries. Several of these studies consider the cement industry in particular. For example, Szabo et al (2006) and Demailly and Quirion (2006) use a bottom-up model of the cement industry to examine impacts of alternative policy designs on industry profits, emissions, and emissions leakage. More recently, Ponssard and Walker (2008) specify a static oligopoly model of a regional European cement industry to examine the short run responses of European cement producers to the ETS. All of these studies find striking differences in impacts across grandfathering, auctioning, and rebating regimes. This paper differs from prior work in some important respects. We investigate the response of the U.S. cement sector to carbon regulation. We estimate an empirically tractable dynamic model in order to obtain estimates of key parameters such as investment costs. Finally, this paper emphasizes dynamic industry responses to policy interventions, and the interplay between emissions regulations and pre-existing distortions associated with the exercise of market power.

The paper also makes an important methodological contribution in its application of parametric value function methods to a dynamic game. We make use of interpolation techniques to compute the equilibrium of the counterfactual simulations. This allows us to treat the capacity of the firms as a continuous state. Even though parametric methods have been used in single agent problems, its application to dynamic industry models with discrete entry, exit and investment decisions has not been very successful to date ([Doraszelski and Pakes](#),

⁵One limitation of these numerical simulation models is that they must rely on the extant econometric literature to provide “off-the-shelf” estimates of important structural parameters (such as the fixed costs of entry or the elasticity of import supply). It is often the case that the econometric literature is not up to the task; models are often parameterized using outdated values or educated guesses.

2007).

The paper is organized as follows. Section 2 introduces the conceptual framework for our applied policy analysis. Section 3 provides some essential background on the US Portland cement industry. We introduce the model and a detailed description of the alternative policy designs we consider in Section 4. We present the estimation and computational methodology in Section 5. The counterfactual simulations are introduced in Section 6. Simulation results are summarized in Section 7. We conclude with a discussion of the results and directions for future research in Section 8.

2 Market-based emissions regulation in a second-best setting

A simple conceptual framework helps to lay the foundation for the applied welfare analysis that is the central focus of the paper. Figure 1 illustrates, among other things, the static welfare consequences of an emissions externality in an industry that is monopolized by a single producer.

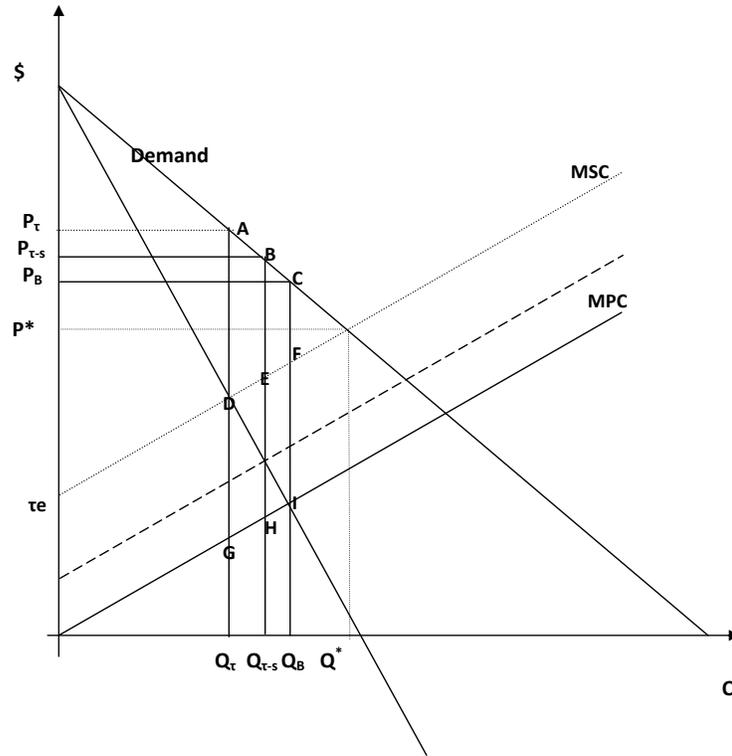
The curve labeled MPC measures the marginal private costs of production (i.e. fuel costs, labor costs, etc.) net of any environmental compliance costs. Absent any emissions regulation, this monopolist will produce output Q_B and receive a price P_B . This is the baseline (B) against which we will compare the alternative policy outcomes.

Production generates harmful emissions. We assume a constant emissions rate per unit of output (e) and a constant marginal social cost of emissions τ . The curve labeled MSC captures both private marginal costs and the monetized value of the damages from the firm's emissions: $MSC = MPC + \tau e$. The social welfare maximizing level of output is Q^* . The corresponding price is P^* .

We first consider a case in which the monopolist is required to pay a Pigouvian tax of τ per unit of emissions. This increases the monopolist's variable operating costs by τe . The monopolist will choose to produce Q_τ . The equilibrium price is P_τ .

Alternatively, consider an emissions trading program in which permits are auctioned off to the highest bidder or freely distributed in lump-sum to regulated sources based on pre-determined, firm-specific characteristics (i.e. "grandfathered"). If the monopolist is sufficiently small relative to the larger emissions trading program, changes in monopolist's net supply or demand for permits will not affect the equilibrium permit price. Within our framework, a large scale emissions trading program that incorporates grandfathering and

Figure 1: Emissions-intensive Monopoly: Static case

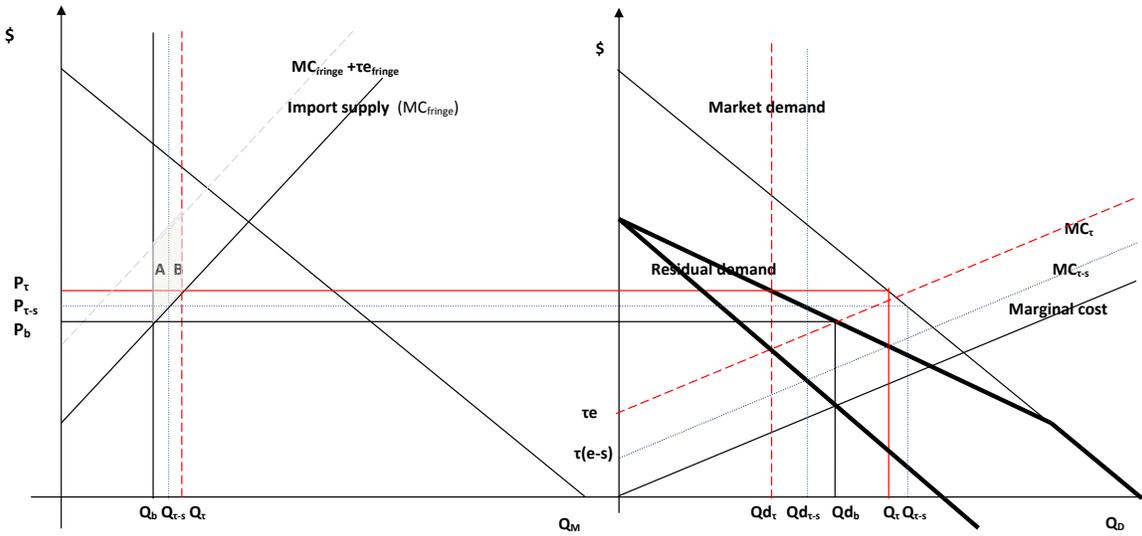


auctioning and yields an equilibrium permit price of τ is functionally equivalent to the emissions tax described in the previous paragraph.

In Figure 1, these market-based emissions regulations will *reduce* welfare because the costs associated with further restricting already sub-optimal levels of output outweigh the benefits associated with emissions abatement. This need not always be the case. If the social cost per unit of emissions is sufficiently large, the benefits from full internalization of the emissions externality will offset the costs associated with reductions in output.

If the emissions regulation were coupled with a production subsidy equal to the difference between marginal cost and marginal revenue at the socially optimal level of output, the efficient outcome could be achieved. Traditionally, it has been assumed that environmental regulators do not have the authority to subsidize the production of the industries they regulate (Cropper and Oates, 1992). However, policy makers have started to experiment with rebating tax revenues (in the case of an emissions tax) or allocating emissions permits (in the case of a cap-and-trade program) on the basis of production. These contingent rebates

Figure 2: Emissions Intensive, Trade Exposed Monopoly



affect marginal production incentives, and can thus be used to mitigate—or eliminate—the distortion introduced by the exercise of market power.

The equilibrium outcome under a market-based emissions regulation that incorporates an output-based rebate (or subsidy) s is denoted $\tau - s$ in Figure 1. The monopolist’s profit maximizing choice of output under contingent rebating is $Q_{\tau-s}$. In this case, the subsidy does not achieve the first best outcome, although it does mitigate the negative welfare impact of the policy.

The policy setting we are concerned with is characterized by both imperfect competition and incomplete emissions regulation. Figure 1 captures only the first consideration. Figure 2 demonstrates the potential implications of incomplete emissions regulation. Now the domestic, emissions-intensive monopolist is exposed to competition from producers in jurisdictions that are exempt from the emissions regulation. In the right panel, the thick lines represent the residual demand curve (i.e. market demand less import supply) faced by the monopolist and the corresponding marginal revenue curve, respectively. The left panel depicts import supply which is modeled as a competitive fringe.

In the absence of any regulation, import supply is given by Q_{mb} . The equilibrium output price is P_b . The introduction of market-based emissions regulation increases the operating costs of the monopolist vis a vis its import competition. In the case of an emissions tax or

a cap-and-trade program with no rebating, import market share increases to $Q_{m\tau}$ and the difference ($Q_{mb} - Q_{m\tau}$) represents leakage in production. Rebating permits or tax revenues to the monopolist based on output reduces this leakage by ($Q_{m\tau} - Q_{m\tau-s}$).

2.1 Welfare decomposition

Expositionally, it will be useful to decompose the net welfare effects of the emissions policy interventions we analyze into three parts:

1. *Changes in economic surplus.* The first part is comprised of producer and consumer surplus plus any tax revenues or auction revenues earned through the government sale of emissions permits. In Figure 1, the introduction of a carbon tax or an emissions trading program that incorporates auctioning or grandfathering reduces producer and consumer surplus by area $ACIG$. Contingent rebating reduces the reduction in consumer and producer surplus by an amount equal to area $ABGH$. Thus, the rebate serves to partially mitigate the distortion associated with the exercise of market power.
2. *Changes in damages from emissions.* An emissions tax or cap-and-trade program reduces economic surplus in the product market, but also reduces damages associated with industrial emissions. Market-based emissions regulations with no rebating reduce emissions damages by an amount equal to area $DFIG$ in Figure 1. Under a tax regime, the introduction of the rebate reduces damages by area $EFHI$.

Under a cap-and-trade program, the introduction of the rebate does not increase emissions in aggregate because emissions are constrained to equal the cap (assuming the cap binds). However, the introduction of the rebate increases emissions in this monopolized industry, thus shifting more of the compliance burden to other industries and sources subject to the cap. We assume a constant permit price, equivalent to assuming that the abatement supply curve facing the monopolist is locally flat. The additional abatement costs which must be incurred outside this industry in order to offset the emissions increase is area $DEHG$.

3. *Emissions leakage.* If the introduction of an emissions regulation increases production—and thus emissions—among producers in unregulated jurisdictions, this emissions “leakage” will offset some of the emissions reductions achieved among regulated sources. In Figure 2, the shaded parallelogram (area $A+B$) denotes the monetary cost of this leak-

age under the market-based regulation that does not incorporate rebating. This cost is reduced to area A under rebating.

Of course, the regional cement markets we analyze are considerably more complex than the stylized cases depicted in Figures 1 and 2. First, regional cement markets are served by more than one firm. Much of the intuition underlying the simple static monopoly case should apply in the case of a static oligopoly (Ebert, 1992). However, the oligopoly response to market-based emissions regulation can be more nuanced in certain situations.⁶

Second, Figures 1 and 2 depict static, short-run responses only. We are particularly interested in how market-based emissions regulations affect welfare via industry dynamics. Over a longer time frame, firms can alter their choice of production scale, technology, entry, exit, or investment behavior in response to an environmental policy intervention.

The welfare impacts of a market-based emissions policy can look quite different across otherwise similar static and dynamic modeling frameworks. On the one hand, incorporating industry dynamics into the simulation model can improve the projected welfare impacts of a given emissions regulation. Intuitively, the short run economic costs of meeting an emissions constraint can be significantly reduced once firms are able to re-optimize production processes, adjust investments in capital stock, and so forth.

On the other hand, incorporating industry dynamics may result in estimated welfare impacts that are strictly smaller than those generated using static models. In the policy context we consider, there are two primary reasons why this can be the case. First, in an imperfectly competitive industry, emissions regulation may further restrict already sub-optimal levels of investment, thus exacerbating the distortion associated with the exercise of market power. Second, a dynamic model captures an additional channel of emissions leakage. In a static model, firms may adjust variable input and output decisions such that less stringently regulated production assets are used more intensively. This leads to emissions leakage in the short run. In our dynamic modeling framework, the emissions regulation can also accelerate exit and retirement of regulated production units. This further increases the market share claimed by unregulated imports, thus increasing the extent of the emissions leakage to unregulated jurisdictions or entities.

⁶For example, if firms are highly asymmetric and the inverse demand function has an extreme curvature, it is possible (in theory) for the optimal tax rate to exceed marginal damage (Levin, 1985).

3 The Portland cement industry

Portland cement is an inorganic, non-metallic substance with important hydraulic binding properties. It is the primary ingredient in concrete, an essential construction material used widely in building and highway construction. Demand for cement comes primarily from the ready-mix concrete industry, which accounts of over 70 percent of cement sales. Other major consumers include concrete product manufacturers and government contractors.

Because of its critical role in construction, demand for cement tends to reflect population, urbanization, economic trends, and local conditions in the cement industry. Cement competes in the construction sector with substitutes such as asphalt, clay brick, rammed earth, fiberglass, steel, stone, and wood (Van Oss, 2003, ENV). Another important class of substitutes are the so called supplementary cementitious materials (SCMs) such as ferrous slag, fly ash, silica fume and pozzolana (a reactive volcanic ash). Concrete manufacturers can use these materials as partial substitutes for clinker.⁷

The US cement industry is fragmented into regional markets. This fragmentation is primarily due to transportation economies. The primary ingredient in cement production, limestone, is ubiquitous and costly to transport. To minimize input transportation costs, cement plants are generally located close to limestone quarries. Land transport of cement over long distances is also not economical because the commodity is difficult to store (cement pulls water out of the air over time) and has a very low value to weight ratio. It is estimated that 75 percent of domestically produced cement is shipped less than 110 miles (Miller and Osborne, 2010).⁸

3.1 Carbon dioxide emissions from cement production

Cement producers are among the largest industrial emitters of airborne pollutants, second only to power plants in terms of the criteria pollutants currently regulated under existing cap-and-trade programs (i.e. NO_x and SO₂). The cement industry is also one of the largest manufacturing sources of domestic carbon dioxide emissions (Kapur et al, 2009). World-wide, the cement industry is responsible for approximately 7 percent of anthropogenic CO₂

⁷The substitution of SCM for clinker can actually improve the quality and strength of concrete. Substitution rates range from 5 percent in standard portland cement to as high as 70 percent in slag cement. These blending decisions are typically made by concrete producers and are typically based on the availability of SCM and associated procurement costs (Van Oss, 2005, facts; Kapur et al, 2009).

⁸Most cement is shipped by truck to ready-mix concrete operations or construction sites in accordance with negotiated contracts. A much smaller percent is transported by train or barge to terminals and then distributed.

emissions (Van Oss, 2003, ENV).

Cement production process involves two main steps: the manufacture of clinker (i.e. pyroprocessing) and the grinding of clinker to produce cement. Carbon dioxide emissions from cement manufacturing are generated almost exclusively in the pyroprocessing stage. A mix comprised of limestone and supplementary materials is fed into a large kiln lined with refractory brick. The heating of the kiln is very energy intensive (temperatures reach temperatures of 1450°C) and carbon intensive (because the primary kiln fuel is coal). Carbon dioxide is released as a byproduct of the chemical process that transforms limestone to clinker. Once cooled, clinker is mixed with gypsum and ground into a fine powder to produce cement.⁹ Trace amounts of carbon dioxide are released during the grinding phase.

Carbon dioxide emissions intensities, typically measured in terms of metric tons of emissions per metric ton of clinker, vary considerably across cement producers. Much of the variation is driven by variation in fuel efficiency. The oldest and least fuel efficient kilns are “wet-process” kilns. As of 2006, there were 47 of these wet kilns in operation (all built before 1975) (PCA PIS, 2006). “Dry process” kilns are significantly more fuel efficient, primarily because the feed material used has a lower moisture content and thus requires less energy to dry and heat. The most modern kilns, dry kilns equipped with pre-heaters and pre-calciners, are more than twice as fuel efficient as the older wet-process kilns.

Because plants with different emissions intensities will respond differently to the policy interventions we analyze, it is important to capture this variation as accurately as possible. Although data limitations prevent us from estimating emissions intensities specific to each kiln in the data set, we can estimate technology-specific emissions rates. Both the IPCC and the World Business Council for Sustainable Development’s Cement Sustainability Initiative (WBC, 2005) have developed protocols for estimating emissions from clinker production. We use these protocols to generate technology-specific estimates of carbon dioxide emissions rates. Appendix A explains these emissions rate calculations in more detail.

There have been several recent studies commissioned to assess the potential for carbon emissions reductions in the cement sector.¹⁰ Using different scenarios, baseline emissions and future demand forecasts, all reach broadly similar conclusions. Although there is no one “silver bullet” on the horizon, there are four key levers for carbon emissions reductions.

⁹The US cement industry is comprised of clinker plants (kiln only operations), grinding-only facilities, and integrated (kiln and grinding) facilities. Almost all of the raw materials and energy used in the manufacture of cement are consumed during pyroprocessing. We exempt grinding only facilities from our analysis.

¹⁰A comprehensive list of studies can be found at <http://www.wbcscement.org/pdf/technology/References%20FINAL.pdf>

We summarize these here. We postpone the discussion of how these abatement options are captured by our modeling framework to section 4.

The first set of strategies involve energy efficiency improvements. The carbon intensity of clinker production can be reduced by replacing older equipment with current state of the art technologies. In the United States, it is estimated that converting wet installed capacity to dry kilns could reduce annual emissions by approximately 15 percent. Converting from wet to the semi-wet process would deliver an additional 3 percent reduction (Mahasenan et al., 2005).

A second set of carbon mitigation strategies involve substitution. One approach is to simply increase the use of substitute construction materials such as wood or brick, thus reducing demand for cement. Alternatively, the amount of clinker needed to produce a given amount of cement can be reduced by the use of supplementary cementitious materials (SCM) such as coal fly ash, slag, and natural pozzolans.¹¹ It is estimated that the increased use of blended cement could feasibly reduce carbon emissions by a third over the time frame we consider (Mahasenan et al., 2005).

Fuel switching offers a third emissions abatement strategy. Less carbon intensive fuels, such as waste derived fuels or natural gas, could replace coal as the primary kiln fuel. The potential for CO₂ mitigation by fuel switching to lower carbon fuels and fuels qualifying for emissions offsets in North America has been estimated to be on the order of 5 percent of current emissions (Humphreys and Mahasenan, 2001).

Finally, carbon dioxide emissions can be separated or captured during or after the production process and subsequently sequestered. This abatement option is unlikely to play a significant role in the near term given that sequestration technologies are in an early stage of technical development or acceptance and are relatively costly.

3.2 Trade Exposure

Whereas overland transport of cement is very costly, sea-based transport of clinker is relatively inexpensive. In the 1970s, technological advances made it possible to transport cement in bulk quantities safely and cheaply in large ocean vessels. Since that time, U.S. imports have been growing steadily. Over the past fifteen years, the share of domestic demand satisfied by foreign imports has averaged 16 percent. The United States now absorbs approximately one quarter of the total global cement trade (Van Oss, 2003 ENV). In the recent past, import

¹¹When part of the cement content of concrete is replaced with supplementary cementitious materials, the extent of the emissions reduction is proportional to the extent to which SCM replaces clinker. Substitution rates as high as 75 percent are possible.

penetration rates have averaged around 20 percent (USGS Mineral Commodity Summary 2010). China is currently the largest supplier of imported cement (accounting for 22 percent of imports), followed by Canada, Korea, and Thailand (USGS, 2010 fact sheet).

Exposure to import competition in regional markets has given rise to growing concerns about unilateral climate policy. For example, an industry trade group has warned that, in the absence of measures that either relieve the initial cost pressure or impose equivalent costs of imports, California’s proposed cap on greenhouse gas emissions will “render the California cement industry economically unviable, will result in a massive shift in market share towards imports in the short run, and will precipitate sustained disinvestment in the California cement industry in the long run.”¹²

4 Model

4.1 Baseline model

The basic building block of the model is a regional cement market.¹³ Each market is fully described by the $\bar{N} \times 2$ state vector, s_t , where s_{it} describes the productive capacity of the i -th firm at time t and its associated emissions rate. We set \bar{N} to be the maximal number of firms. Firms with zero capacity are considered to be potential entrants. Time is discrete and unbounded. Firms discount the future at rate $\beta = 0.9$.

Each decision period is one year. In each period, the sequence of events unfolds as follows: first, incumbent firms receive a private draw from the distribution of scrap values, and decide whether or not to exit the industry. Potential entrants receive a private draw from the distribution of both investment and entry costs, while incumbents who have decided not to exit receive private draws on the fixed costs of investment and divestment. All firms then simultaneously make entry and investment decisions. Third, incumbent firms compete over quantities in the product market. Finally, firms enter and exit, and investments mature. We assume that firms who decide to exit produce in this period before leaving the market, and that adjustments in capacity take one period to realize. We also assume that each firm operates independently across markets.¹⁴

¹²Letter from the Coalition for Sustainable Cement Manufacturing and Environment to Larry Goulder, Chair of the Economic and Allocation Advisory Committee. Dec. 19, 2009.

¹³This section borrows heavily from [Ryan \(2011\)](#).

¹⁴This assumption explicitly rules out more general behavior, such as multimarket contact as considered in [Bernheim and Whinston \(1990\)](#) and [Jans and Rosenbaum \(1997\)](#).

Firms obtain revenues from the product market and incur costs from production, entry, exit, and investment. Firms compete in quantities in a homogeneous goods product market. Firms face a constant-elasticity aggregate demand curve:

$$\ln Q_m(\alpha) = \alpha_{0m} + \alpha_1 \ln P_m, \quad (1)$$

where Q_m is the aggregate market quantity, P_m is price, α_{0m} is a market-specific intercept, and α_1 is the elasticity of demand.

For firms in trade-exposed regional markets, the effective residual demand that they face is more elastic and potentially kinked, as they also face an import supply curve given by:

$$\ln M_m = \rho_0 + \rho_1 \ln P_m, \quad (2)$$

where M_m measures annual import supply in market m and ρ_1 is the elasticity of import supply. Here we assume that the elasticity of import supply is an exogenously determined parameter.¹⁵ For clarity, we omit the m subscript in what follows.

There are essentially five variable inputs used in cement production: labor, fuel (primarily coal), electricity, feedstocks, and maintenance. These factor inputs are not substitutable (Das, 1994). The majority of variable operating costs are energy related. Because frequent heating and cooling damages the firebrick lining, kilns typically operate continuously at full capacity for 24 hours a day. Annual output is adjusted by varying the length of time the kiln is shut down for annual maintenance. In the model, each firm chooses the level of annual output that maximizes their static profits given the outputs of the competitors, subject to capacity constraints that are determined by dynamic capacity investment decisions:

$$\max_{q_i} P \left(q_i + \sum_{j \neq i} q_j; \alpha \right) q_i - C_i(q_i; \delta) - \varphi(q_i, e_i, \tau), \quad (3)$$

where $P(Q; \alpha)$ is the inverse of Equation 1. In the presence of fixed operation costs the product market may have multiple equilibria, as some firms may prefer to not operate given

¹⁵In fact, firms that own a majority of the domestic production capacity in the United States are also among the largest importers. These dominant producers presumably use imports to supplement their domestic production as needed, and to compete in markets where they do not own production facilities. Domestic cement producers have noted that increased domestic ownership of import facilities has contributed to a “more orderly flow of imports into the U.S.”

Grancher, Roy A. “U.S. Cement: Record Performance and Reinvestment”, Cement Americas, Jul 1, 1999

the outputs of their competitors. However, if all firms produce positive quantities then the equilibrium vector of production is unique, as the best-response curves are downward-sloping.

We will use this model to evaluate the impacts of alternative approaches to allocating emissions permits in an emissions trading program. Firm-specific compliance costs will be determined by kiln-specific emissions rates, e_i , production quantity, and the number of permits the firm receives free of charge. While postponing the discussion of the policy designs we consider until Section 4.2, we note here that the introduction of a tax or emissions trading program modifies the profit function in Equation 3 through the term $\varphi(q_i, e_i, \tau)$, where τ is the price paid to offset one metric ton of carbon dioxide. The precise nature of the modification will vary across policy designs.

The cost of output, q_i , is given by the following function:

$$C_i(q_i; \delta) = \delta_1 q_i + \delta_2 1(q_i > \nu s_i)(q_i - \nu s_i)^2. \quad (4)$$

Variable production costs consist of two parts: a constant marginal cost, δ_1 , and an increasing function that binds as quantity approaches the capacity constraint. We assume that costs increase as the square of the percentage of capacity utilization, and parameterize both the penalty, δ_2 , and the threshold at which the costs bind, ν . This second term, which gives the cost function a “hockey stick” shape common in the electricity generation industry, accounts for the increasing costs associated with operating near maximum capacity, as firms have to cut into maintenance time in order to expand production beyond utilization level ν . We denote the profits accruing from the product market by $\bar{\pi}_i(s; \alpha, \delta)$.

Firms can change their capacity through costly adjustments, denoted by x_i . The cost function associated with these activities is given by:

$$\Gamma(x_i; \gamma) = 1(x_i > 0)(\gamma_{i1} + \gamma_2 x_i + \gamma_3 x_i^2) + 1(x_i < 0)(\gamma_{i4} + \gamma_5 x_i + \gamma_6 x_i^2). \quad (5)$$

Firms face both fixed and variable adjustment costs that vary separately for positive and negative changes. Fixed costs capture the idea that firms may have to face significant setup costs, such as obtaining permits or constructing support facilities, that accrue regardless of the size of the kiln. Fixed positive investment costs are drawn each period from the common distribution F_γ , which is distributed normally with mean μ_γ^+ and standard deviation σ_γ^+ , and are private information to the firm. Divestment sunk costs may be positive as the firm may encounter costs in order to shut down the kiln and dispose of related materials and components. On the other hand, firms may receive revenues from selling off their infrastructure,

either directly to other firms or as scrap metal. These costs are also private information, and are drawn each period from the common distribution G_γ , which is distributed normally with mean μ_γ^- and standard deviation σ_γ^- .

Firms face fixed costs unrelated to production, given by $\Phi_i(a)$, which vary depending on their current status and chosen action, a_i :

$$\Phi_i(a_i; \kappa_i, \phi_i) = \begin{cases} -\kappa_i & \text{if the firm is a new entrant,} \\ \phi_i & \text{if the firm exits the market.} \end{cases} \quad (6)$$

Firms that enter the market pay a fixed cost of entry, κ_i , which is private information and drawn from the common distribution of entry costs, F_κ . Firms exiting the market receive a payment of ϕ_i , which represents net proceeds from shuttering a plant, such as selling off the land and paying for an environmental cleanup. This value may be positive or negative, depending on the magnitude of these opposing payments. The scrap value is private information, drawn anew each period from the common distribution, F_ϕ . Denote the activation status of the firm in the next period as χ_i , where $\chi_i = 1$ if the firm will be active next period, whether as a new entrant or a continuing incumbent, and $\chi_i = 0$ otherwise. All of the shocks that firms receive each period are mutually independent.

Collecting the costs and revenues from a firm's various activities, the per-period payoff function is:

$$\pi_i(s, a; \alpha, \rho, \delta, \gamma_i, \kappa_i, \phi_i) = \bar{\pi}_i(s; \alpha, \rho, \delta) - \Gamma(x_i; \gamma_i) + \Phi_i(a_i; \kappa_i, \phi_i). \quad (7)$$

For the sake of brevity, we henceforth denote the vector of parameters in Equation 7 by θ .

4.1.1 Transitions Between States

To close the model it is necessary to specify how transitions occur between states as firms engage in investment, entry, and exit. We assume that changes to the state vector through entry, exit, and investment take one period to occur and are deterministic. The first part is a standard assumption in discrete time models, and is intended to capture the idea that it takes time to make changes to physical infrastructure of a cement plant. The second part abstracts away from depreciation, which does not appear to be a significant concern in the cement industry, and uncertainty in the time to build new capacity.¹⁶

¹⁶It is conceptually straightforward to add uncertainty over time-to-build in the model, but assuming deterministic transitions greatly reduces the computational complexity of solving for the model's equilibrium.

We also assume that the emissions rate of the firm is fixed. We assume that there are three discrete levels of emissions rates, corresponding to the three major types of production technology in the cement industry. Existing incumbents are modeled as having one of the three technologies, while new entrants are always endowed with the frontier technology. As a result, the emissions profile of an industry changes over time in response to firm turnover.

4.1.2 Equilibrium

In each time period, firm i makes entry, exit, production, and investment decisions, collectively denoted by a_i . Since the full set of dynamic Nash equilibria is unbounded and complex, we restrict the firms' strategies to be anonymous, symmetric, and Markovian, meaning firms only condition on the current state vector and their private shocks when making decisions, as in Maskin and Tirole (1988) and Ericson and Pakes (1995).

Each firm's strategy, $\sigma_i(s, \epsilon_i)$, is a mapping from states and shocks to actions:

$$\sigma_i : (s, \epsilon_i) \rightarrow a_i, \quad (8)$$

where ϵ_i represents the firm's private information about the cost of entry, exit, investment, and divestment. In the context of the present model, $\sigma_i(s)$ is a set of policy functions which describes a firm's production, investment, entry, and exit behavior as a function of the present state vector. In a Markovian setting, with an infinite horizon, bounded payoffs, and a discount factor less than unity, the value function for an incumbent at the time of the exit decision is:

$$\begin{aligned} V_i(s; \sigma(s), \theta, \epsilon_i) = & \bar{\pi}_i(s; \theta) + \max \left\{ \phi_i, E_{\epsilon_i} \left\{ \beta \int E_{\epsilon_i} V_i(s'; \sigma(s'), \theta, \epsilon_i) dP(s'; s, \sigma(s)) \right. \right. \\ & + \max_{x_i^* > 0} \left[-\gamma_{i1} - \gamma_2 x_i^* - \gamma_3 x_i^{*2} + \beta \int E_{\epsilon_i} V_i(s'; \sigma(s'), \theta, \epsilon_i) dP(s_i + x^*, s'_{-i}; s, \sigma(s)) \right], \\ & \left. \left. \max_{x_i^* < 0} \left[-\gamma_{i4} - \gamma_5 x_i^* - \gamma_6 x_i^{*2} + \beta \int E_{\epsilon_i} V_i(s'; \sigma(s'), \theta, \epsilon_i) dP(s_i + x^*, s'_{-i}; s, \sigma(s)) \right] \right\} \right\}, \quad (9) \end{aligned}$$

where θ is the vector of payoff-relevant parameters, E_{ϵ_i} is the expectation with respect to the distributions of shocks, and $P(s'; \sigma(s), s)$ is the conditional probability distribution over future state s' , given the current state, s , and the vector of strategies, $\sigma(s)$.

Potential entrants must weigh the benefits of entering at an optimally-chosen level of capacity against their draws of investment and entry costs. Firms only enter when the sum of these draws is sufficiently low. We assume that potential entrants are short-lived; if they

do not enter in this period they disappear and take a payoff of zero forever, never entering in the future.¹⁷ Potential entrants are also restricted to make positive investments; firms cannot “enter” the market at zero capacity and wait for a sufficiently low draw of investment costs before building a plant. The value function for potential entrants is:

$$V_i^e(s; \sigma(s), \theta, \epsilon_i) = \max \left\{ 0, \max_{x_i^* > 0} \left[-\gamma_{1i} - \gamma_2 x_i^* - \gamma_3 x_i^{*2} + \beta \int E_{\epsilon_i} V_i(s'; \sigma(s'), \theta, \epsilon_i) dP(s_i + x^*, s'_{-i}; s, \sigma(s)) \right] - \kappa_i \right\}. \quad (10)$$

Markov perfect Nash equilibrium (MPNE) requires each firm’s strategy profile to be optimal given the strategy profiles of its competitors:

$$V_i(s; \sigma_i^*(s), \sigma_{-i}(s), \theta, \epsilon_i) \geq V_i(s; \tilde{\sigma}_i(s), \sigma_{-i}(s), \theta, \epsilon_i), \quad (11)$$

for all s , ϵ_i , and all possible alternative strategies, $\tilde{\sigma}_i(s)$. As we work with the expected value functions below, we note that the MPNE requirement also holds after integrating out firms’ private information: $E_{\epsilon_i} V_i(s; \sigma_i^*(s), \sigma_{-i}(s), \theta, \epsilon_i) \geq E_{\epsilon_i} V_i(s; \tilde{\sigma}_i(s), \sigma_{-i}(s), \theta, \epsilon_i)$. Doraszelski and Satterthwaite (2010) discuss the existence of pure strategy equilibria in settings similar to the one considered here. The introduction of private information over the discrete actions guarantees that at least one pure strategy equilibrium exists, as the best-response curves are continuous. However, there are no guarantees that the equilibrium is unique, a concern we discuss next in the context of my empirical approach.

4.2 Market based emissions policy designs

We use the model to simulate both static and dynamic industry response to the introduction of both price instruments (emissions taxes) and quantity instruments (cap-and-trade programs). In the tax regimes we consider, all domestic producers must pay τ per unit of emissions. In the emissions trading programs we analyze, an emissions cap limits greenhouse gas emissions across multiple emissions-intensive sectors. To comply with the trading program, producers must hold permits to offset their uncontrolled emissions. We impose no spatial or sectoral restrictions on permit trading; permits can be traded freely among all program participants. To keep the analysis more tractable, we do not allow banking or

¹⁷This assumption is for computational convenience, as otherwise one would have to solve an optimal waiting problem for the potential entrants. See Ryan and Tucker (2011) for an example of such an optimal waiting problem.

borrowing of permits across time.

The carbon price, τ , is an exogenous parameter. In the case of the tax, this simply means that the level of the tax does not depend on the production and/or pollution decisions of the regulated firms. The tax is set by the regulator and does not change over the time horizon we consider (30 years). In the case of an emissions trading program, we assume that the aggregate marginal abatement cost curve is flat in the neighborhood of the constraint imposed by the emissions cap. This will be an appropriate assumption if the domestic cement industry is a relatively small player in the emissions market, such that changes in industry net supply/demand for permits cannot affect the equilibrium market price.¹⁸ The policy designs we analyze can best be classified into one of four categories: standard auction design/ carbon tax; grandfathering (i.e. lump sum transfer); output-based rebating; emissions-based rebating. In the subsections that follows, these policy design alternatives are described in detail.

4.2.1 Standard design: Emissions tax or emissions trading with auctioned permits

In the wake of failed attempts to implement a federal cap-and-trade program for greenhouse gases, some are advocating for a reconsideration of a carbon tax.¹⁹ In the context of an economy-wide greenhouse gas emissions trading program, a cap-and-trade program that incorporates auctioning also has its proponents.²⁰ Given our assumption about the exogeneity of the carbon price, these two market-based policy designs are, within our modeling framework, functionally equivalent.

The first policy regime we analyze is intended to capture the most salient features of an emissions tax or an emissions cap-and-trade program in which all emissions permits are allocated via a uniform price auction. In the tax regime, regulated firms must pay a tax τ for each ton of emissions. In the emissions trading regime, the equilibrium permit price is τ ;

¹⁸This assumption is likely to be approximately true in the context of a federal GHG trading program that permits offsets. Keohane (2009) estimates the slope of the marginal abatement cost curve in the United States (expressed in present-value terms and in 2005 dollars) to be 8.0×10^7 \$/GT CO₂ for the period 2010–2050. Suppose this curve can be used to crudely approximate the permit supply function. If all of the industries deemed to be “presumptively eligible” for allowance rebates reduced their emissions by ten percent for this entire forty year period, the permit price would fall by approximately \$0.25/ ton.

¹⁹Blinder, Alan. January 31, 2011. “The Carbon Tax Miracle Cure”. Wall Street Journal.

²⁰ For example, in 2007, the Congressional Budget Office Director warned that a failure to auction permits in a federal greenhouse gas emissions trading system “would represent the largest corporate welfare program that has even been enacted in the history of the United States” “Approaches to Reducing Carbon Dioxide Emissions: Hearing before the Committee on the Budget U.S. House of Representatives”, November 1, 2007. (testimony of Peter R. Orszag)

a change in the net supply or demand for permits from the domestic cement industry does not affect this price.

The per-period production profit function becomes:

$$\pi_{it} = P \left(q_{it} + \sum_{j \neq i} q_{jt}; \alpha \right) q_{it} - C_i(q_{it}; \delta) - \tau e_i q_{it}, \quad (12)$$

where e_i is the firm's emissions rate and E represents aggregate industry emissions.

4.2.2 Grandfathering

In this policy scenario, tradable emissions permits are allocated for free to incumbent firms that pre-date the carbon trading program. Firm-specific permit allocation schedules (i.e. the number of permits the firm will receive each period) are determined at the beginning of the program and are based on historic emissions.

Several studies have demonstrated that a pure grandfathering regime would grossly over-compensate industry for the compliance costs incurred under proposed Federal climate change legislation. For example, a recent paper finds that grandfathering fewer than 15 percent of the emissions allowances generally suffices to prevent profit losses among industries that would suffer the largest percentage losses of profit absent compensation (Goulder, Hafstead, and Dworsky, 2010). Under the grandfathering regime we consider, we assume that a number of permits equal to 20 percent of annual baseline emissions are grandfathered each year to incumbent cement producers. The per period profit function becomes:

$$\begin{aligned} \pi_{it} &= P \left(q_{it} + \sum_{j \neq i} q_{jt}; \alpha \right) q_{it} - C_i(q_{it}; \delta) - \tau(e_i q_{it} - A_i), \\ \text{with } \sum_i A_i &= \bar{A}. \end{aligned} \quad (13)$$

The number of permits the firm receives for free from the regulator is A_i ; \bar{A} represents the total amount of emissions allocated for free to domestic cement producers. We assume that the share of emissions allowances allocated to firm i (i.e. A_i/\bar{A}) is equal to its share of the installed kiln capacity at the outset of the program.

Note that the first order conditions associated with static profit maximization under auctioning are identical to those under grandfathering. This highlights the so-called ‘‘independence property’’, which holds that firms’ short run production and abatement decisions

will be unaffected by the choice between auctioning permits or allocating them freely to firms in lump sum (Hahn and Stavins, 2010).

When permits are grandfathered in a cap and trade program, policy makers must decide ex ante how to deal with new entrants and firms who exit. In our simulations, we assume that a firm forfeits its future entitlements to free permits when it exits the market. We assume that new entrants are not entitled to free permits.²¹ In some existing program designs (including the EU ETS), some fraction of the permits to be allocated are set aside for new production capacity entering the market. Future work will explore these alternative policy designs that offer free permit allocations as incentive for new entrants.

4.2.3 Output-based allocation updating/rebating

The third policy regime we analyze incorporates output-based rebating. This scenario can be motivated in two ways. First, under an emissions tax, tax revenues can be rebated to regulated firms based on output. For example, Sweden has refunded revenues from a tax on nitrogen oxide emissions in proportion to output (Sterner and Isaksson, 2006). Second, our modeling approach also captures the essential features of an emissions trading program in which free permit allocations are contingent upon production levels. For example, under proposed state and federal climate change legislation, output-based updating provisions are used to address concerns about near-term competitiveness impacts, job loss, and emissions leakage. Emissions permits are allocated for free to eligible firms using a continuously updated, output-based formula.²²

Following Bushnell and Chen (2009), we adopt a closed-loop approach to modeling of these kinds of rebating regimes. Permits are allocated (or tax revenues are recycled) based on product shares (or emissions shares) in the current period. The per-period profit function

²¹In practice, policies regarding free permit allocations to free entrants and former incumbents vary. In the EU ETS, policies governing the free allocation of permits to entrants vary across member states. Most states require forfeiture of free permit allocations upon closure.

²²Proposed federal climate change legislation included a provision to allocate permits to eligible industries using an output-based formula. These free allocations are intended to compensate both direct compliance costs (i.e. the cost of purchasing permits to offset emissions) and indirect compliance costs (i.e. compliance costs reflected in higher electricity prices). Under California's Assembly Bill 32, implementing agencies have recommended that free allocation to industry will, "to the extent feasible, be based on output-based GHG efficiency "benchmarks" and "update" to reflect changes in production each year for industry with leakage risk" (Greenhouse Gas Cap-and-Trade Regulation Status Update May 17, 2010 California Air Resources Board).

becomes:

$$\pi_{it} = P \left(q_{it} + \sum_{j \neq i} q_{jt}; \alpha \right) q_{it} - C_i(q_{it}; \delta) - \tau(e_i q_{it} - \phi_i(q_{it})\bar{A}), \quad (14)$$

where $\phi_i(q_{it})$ denotes the share of the total rebate allocated to firm i . Emissions allowances are allocated (or tax revenues are rebated) according to market share:

$$\phi_i(q_i) = \frac{q_i}{\sum_j q_j}.$$

We make two simplifying assumptions. First, implicit in Equation 14, the rebate a firm receives in the current period depends on its production level in that same period. Thus, we do not explicitly account for the fact that firms will discount the value of the subsidy conferred by rebating if the rebate is paid in a future period. Second, the size of the implicit subsidy per unit of output is taken to be exogenous to firms' production decisions. More precisely, we assume that firms do not take into account how their production decisions affects the size of the implicit subsidy via the effect on aggregate production levels. Together, these assumptions simplify the dynamic problem considerably, while still allowing us to capture the dynamic implications of the grandfathering mechanism to a significant extent.

4.2.4 Border tax adjustment with auctioned permits

The fourth and final policy design alternative that we consider layers a border tax adjustment (BTA) atop the standard tax/auctioning regime. This BTA mechanism imposes a tax on emissions embodied in cement imports equal to the tax imposed on domestic emissions. This effectively levels the carbon playing field.

The BTA regime is equivalent to the auctioning regime in terms of strategic incentives. However, domestic firms now face a different residual demand, as the import supply is shifted to the left as follows:

$$\ln M_m = \rho_0 + \rho_1 \ln(P_m - \tau e_m), \quad (15)$$

where e_m is the emissions rates on imported cement.

4.3 Modeling Emissions Abatement

Section 3.1 included a discussion of how carbon dioxide emissions reductions can be achieved in the domestic cement industry. If emissions from cement manufacturing were to be subject to a binding cap, it is anticipated that mandated emissions reductions would be achieved

through a combination of factors. Chief among these are the replacement of old production processes with new state-of-the-art technology and the increased substitution of less carbon intensive materials for clinker or cement.

We explicitly model what is expected to be the most important efficiency improvement: the replacement of older kiln technology with current, state-of-the-art technology. Our modeling approach is well suited to modeling the retirement of old process equipment and entry of new firms. We assume all new entrants adopt new, state-of-the-art equipment. This assumption finds empirical support in the data. Our specific assumptions about the emissions intensities of old and new production equipment are described in Appendix A.

The substitution of SCM for clinker is also expected to play an important role in delivering emissions reductions in a carbon constrained cement industry. Supplementary cementitious materials are used widely throughout the U.S. as additives to concrete. Utilization rates have varied due to economic considerations and the availability of materials. Although we do not explicitly model the substitution of SCMs for clinker, this substitution is implicitly captured, to some extent, by our estimated demand elasticity.

Ideally, a model designed to simulate industry response to an emissions regulation would accurately capture all viable carbon abatement strategies. Unfortunately, our econometric approach is not well suited to modeling responses that have yet to be observed in the data. Consequently, fuel switching and carbon sequestration are not represented in our model. Although these options are not expected to play as significant a role as efficiency improvements or substitution, this omission will bias up our estimates of the economic costs imposed of the emissions regulations we analyze.²³

5 Estimation and computation

The econometric estimation is based on the benchmark model, in which the price of emissions is set to zero ($\tau = 0$), i.e. there is no compliance cost due to emissions regulation. Once estimated, this model can be used to simulate the dynamic industry response to market-based emissions regulations that affect firms' production and investment choices primarily through operating costs provided certain assumptions are met. In particular, we will assume that firms' response to a given operating cost change is independent of whether the cost change is caused by emissions regulation or other exogenous factors (such as changes in

²³In future work we plan to compute what would be an upper bound on the cost of fuel switching for it to be observed in equilibrium together with sensitivity analysis on how important such adoption would be for mitigating the adverse effects of carbon regulation.

energy prices or other inputs).

5.1 Estimation

Although our data sources and identification strategy are similar to [Ryan \(2011\)](#), there are some important differences in how the model is specified and estimated. In this section, significant deviations are discussed. The interested reader is referred to [Ryan \(2011\)](#) for additional details regarding the data and estimation.

5.1.1 Regional market definition

The USGS collects establishment-level data from all domestic Portland cement producers and publishes these data in an annual Minerals Yearbook. Regional shipments and shipment values are aggregated to the regional market level to protect the confidentiality of the respondents. In recent years, increased consolidation of asset ownership has required higher levels of data aggregation. Conversations with the experts at USGS indicate that the current regional market definitions group plants that are unlikely to compete with each other (Van Oss, personal communication).

Rather than adopt the USGS protocols, we base our regional market definitions on the industry-accepted limitations of economic transport as well as company-specific SEC 10k filings which include information regarding markets served by specific plants. To merge the USGS cement prices and quantities with our data set, USGS data are weighted by kiln capacity in each region. For example, if kiln capacity in USGS market A is equally divided between regional markets we define to be B and C, production quantities in market A are equally divided between our defined markets B and C. We report some descriptive statistics using USGS data from 2006 for our regional markets in [Table 1](#).

This table helps to highlight inter-regional variation in market size, emissions intensity, and trade exposure. Notably, the degree of import penetration varies significantly across inland and coastal areas. Whereas several inland markets are supplied exclusively by domestic production, imports now account for over half of domestic cement consumption in Seattle. Import penetration rates tend to be highest along the coasts versus inland waterways.

In our analysis, we will be interested in understanding how policy responses differ across regional markets that differ in terms of trade exposure, concentration of ownership, etc. However, data limitations prevent us from estimating *market specific* demand and cost parameters. For example, we will use the same average demand elasticity parameter in all

Table 1: Descriptive Statistics for Regional Markets (based on 2006 data)

Market	Number of Firms	Capacity	Emissions Rate	Import Market Share
Atlanta	6	1285	0.97	0.12
Baltimore/Philadelphia	6	1497	0.99	0.12
Birmingham	5	1288	0.94	0.35
Chicago	5	972	0.98	0.04
Cincinnati	3	875	0.93	0.21
Dallas	5	1766	1.05	0
Denver	4	998	0.95	0
Detroit	3	1749	1.02	0.19
Florida	5	1297	0.93	0.35
Kansas City	4	1661	0.95	0
Los Angeles	6	1733	0.93	0.18
Minneapolis	1	1862	0.93	0.2
New York/Boston	4	1033	1.16	0.45
Phoenix	4	1138	0.93	0.13
Pittsburgh	3	614	1.08	0
Salt Lake City	2	1336	1.01	0
San Antonio	6	1318	0.95	0.3
San Francisco	4	931	0.93	0.18
Seattle	2	607	1.05	0.65
St Louis	4	1358	1.05	0

regional market simulations. These average demand parameter values may overstate the residual demand faced by firms in some markets. This can trigger immediate investment in some markets, particularly those that are spatially sparse. To deal with this problem, we first simulate outcomes in each regional market with no carbon policy in place for one period so that investments can adjust if needed. Once these investments have been made, we introduce the carbon policy. This approach is intended to separate the effects of possible misspecification in the model or demand estimation from the effects of the counterfactual policy on simulated outcomes.²⁴

5.1.2 Import supply and residual demand elasticities

We estimate the following demand equation using two stage least squares (2SLS):

$$\ln Q_{mt} = \gamma_0 + \gamma_1 \ln P_{mt} + \gamma_{2m} + \gamma_3' \ln X_{mt} + \varepsilon_{1mt}. \quad (16)$$

The dependent variable is the natural log of the total market demand in market m in

²⁴Running the model without this adjustment generates similar qualitative results, although the negative welfare effects from cap-and-trade are exacerbated in the non-adjusted model.

year t . The coefficient on market price, γ_1 , is the elasticity of demand, and X_{mt} is a set of demand shifters.

We instrument for the potential endogeneity of cement price using supply-side cost shifters: coal prices, gas prices, electricity rates, and wage rates. Each market has a demand shifter in the intercept, γ_{2m} , using Atlanta as the baseline market. Data sources are summarized in [Ryan \(2011\)](#).

Given our interest in understanding how policy-induced operating cost increases could affect import penetration rates, it will be important to separate the import supply response to changes in domestic operating costs from the domestic market demand response. We estimate the following import supply schedule using 2SLS:

$$\ln M_{mt} = \phi_0 + \phi_1 \ln P_{mt} + \phi_{2m} + \phi_3' \ln Z_{mt} + \varepsilon_{2mt}. \quad (17)$$

This model is estimated using data from those markets exposed to import competition over the period 1993-2007. For inland markets supplied entirely by domestic production, all ϕ coefficients are set to zero. The dependent variable is the log of the quantity of cement shipped to market m in year t . The average customs price of cement is P_{mt} . These data are collected by the U.S. Geological Survey and are published in the annual Minerals Yearbook. These data are reported by Customs districts (i.e. groupings of ports of entry). These districts are matched to the regional markets described in the previous section.

We instrument for the import price using new residential construction building starts, gross state product, value of construction, and population. These state-level data are aggregated for all states included in the regional market area. The matrix Z_{mt} includes other plausibly exogenous factors that affect import supply. To capture transportation costs, we subtract the average customs price from the average C.I.F. price of the cement shipments. This residual price accounts for the transportation cost on a per unit basis, as well as the insurance cost and other shipment-related charges. The Z_{mt} matrix also includes coal and oil prices to capture variation in production costs. Region dummy variables capture regional differences.

To construct the residual demand curve faced by domestic producers in a trade exposed market, the import supply at a given price is subtracted from the aggregate demand at that price. The resulting residual demand does not necessarily feature a constant elasticity and potentially features a kink at the price below which importers do not supply any output at the market. Strictly positive imports are observed in coastal markets across all policy

simulations.²⁵

5.2 Estimation results

Table 2 enumerates the parameter estimates used in our simulations. Overall, these estimates appear reasonable.

- The marginal cost estimate of \$30/ton of clinker falls well within the range that is typically reported for domestic production: \$27-\$44 per ton (Van Oss, 2003 ENV).
- The import supply elasticity point estimate is 2.5. When analyzing the impacts of environmental regulations, the US EPA assumes an import supply elasticity of 3.94 for the cement sector based on Burtraw et al (2010).
- The elasticity of aggregate demand is -2.96. This is higher in absolute value than some other demand elasticities reported in the literature. For example, Jans and Rosenbaum (1996) estimate a domestic demand elasticity of -0.81. On the other hand, using much higher-quality data, Foster, Haltiwanger, and Syverson (2008) estimate several similar high demand elasticities for homogeneous goods industries, such as -5.93 for ready-mixed concrete, cement's downstream industry.
- Investment costs are roughly in line with the accounting costs cited in Salvo (2010), which reports a cost of \$200 per ton of installed capacity. Our numbers are slightly higher, which is in line with the idea that these costs represent economic opportunity costs as opposed to accounting costs. The implied cost of a cement plant is also in line with plant costs reported in newspapers and trade journals. For example, on October 15, 2010, it was reported that the most recent expansion of the Texas Industries New Braunfels cement plant, increasing capacity from 900 thousand tons per year to 2.3 million tons per year, was pegged at a cost of \$350M, which implies a cost of \$250 per ton of installed capacity.²⁶
- The magnitudes of the fixed costs are reasonable at face value, and in conjunction with the estimated variances, are in accord with the observed rates of investment, entry, and exit in the cement industry.

²⁵This is intuitive as the costs of the domestic industry increase in the counterfactuals considered, which weakly raises the market price.

²⁶Source: [KGNB Radio](#), New Braunfels, Texas.

Some of the parameters from the model described above are not reported. Substantial divestment is virtually never observed in the data and thus the estimates of divestment costs to be very large. Fixed costs of production and operation are also not reported, as these are set to zero. The reason is that we do not observe sufficient periods of operation without production (mothballing) which are required to separately identify those parameters from the distribution of exit costs.

5.3 Computation

Once the parameters have been estimated, the model can be computed to compare the market performance under market-based policy designs. In order to compute the equilibrium of the game, we make use of parametric approximation methods. In particular, we interpolate the value function using cubic splines. The reasons behind using parametric methods are twofold. First, the game has a continuous state space, given by the vector of capacities of the firms. By using parametric methods, we can allow firms to deterministically choose their capacity in a continuous space. Second, parametric approximation methods can be useful to improve computational speed. Previous work has already suggested the potential benefits of using parametric approximation methods (Pakes and McGuire, 1994).

Parametric value function methods have been explored in a single agent dynamic programming context.²⁷ However, they have not been widely used in dynamic games, particularly in games in which players take discrete actions, such as entry and exit (Doraszelski and Pakes, 2007). In our application, we find the method to perform well compared to a discrete value function method. In particular, this parametric method allows us to treat capacity as a continuous state, which improves the convergence properties of the game.²⁸

The procedure we use is similar in spirit to the discrete value function iteration approach. In both methods, the value function is evaluated at a finite number of points. At each iteration and for a given guess of the value function, firms' strategies are computed optimally (*policy step*). Then, the value function is updated accordingly (*value function step*). This process is repeated until the value function and the policy functions do not change significantly.

The difference between the discrete value function iteration and our iterative approach is

²⁷For a general treatment of approximation methods used in the context of dynamic programming, see Judd (1998). An assessment of these methods in a single agent model can be found in Benitez-Silva et al. (2000).

²⁸This is mainly driven by the fact that firms take deterministic actions with respect to the continuous state.

Table 2: Simulation Parameters

Parameter	Value
Demand Parameters	
Constant	20.38
Elasticity of Demand	-2.96
Discount Factor	
β	0.9
Production Parameters	
Capacity Cost	1.157E10
Capacity Cost Binding Level	1.896
Marginal Cost	30
Investment Parameters	
Fixed Cost Mean	1,798
Fixed Cost Standard Deviation	420
Marginal Cost	233
Exit Cost	
Scrap Distribution Mean	-67,490
Scrap Distribution Standard Deviation	55,167
Entry Distribution	
Entry Cost Mean	172,680
Entry Cost Standard Variance	41,559

that we approximate the value function with a flexible parametric form. In particular, given a guess for the value function V^k at pre-specified grid points, we interpolate the value function with a multi-dimensional uniform cubic spline, which can be computed very efficiently (Habermann and Kindermann, 2007).²⁹ This interpolation defines an approximation of the value function in a continuous space of dimension equal to the number of active firms. For a given number of firms active N_A in the market, the value function at any capacity vector s is approximated as,

$$\hat{V}_i^k(s) = \sum_{j=1}^{(J+2)^A} \phi_{N_A,j} B_{N_A,j}(s), \quad (18)$$

where J is the number of grid points, $\phi_{N_A,j}$ are the coefficients computed by interpolating the values V^k when there are A active firms, and $B_{N_A,j}(s)$ is the spline weight given to coefficient $\phi_{N_A,j}$ when the capacity state equals s . This coefficient is the product of capacity weights for each of the incumbent firms, so that $B_{N_A,j}(s) = \prod_{i \in A} B_j(s_i)$.

In the *policy step*, optimal strategies are computed over this continuous function. For a given firm, we compute the conditional single-dimensional value function, given the capacity values of the other firms, $\hat{V}_i^k(s_i|s_{-i})$. This formulation allows us to represent the single-dimensional investment problem of the firm. The following expression defines the expected value function of the firm conditional on staying in the market and investing to a new capacity s'_i . Firms maximize,

$$\max_{s'_i} \pi_i(s_i, s'_i|s_{-i}) + \sum_{s'_{-i} \in S_{-i}} Pr^k(s'_{-i}; \sigma^k(s)) \hat{V}_i^k(s'_i|s'_{-i}). \quad (19)$$

We compute the optimal strategy by making use of the differentiability properties of the cubic splines, which allows us to compute the first-order conditions with respect to investment. Given that the cubic spline does not restrict the value function to be concave, we check all local optima in order to determine the optimal strategy of the firm.³⁰ Conditional on optimal investment strategies, we then compute the new policy function with respect to the entry, investment and exit probabilities, which gives us an updated optimal policy σ^{k+1} . This allows us to compute a new guess for the value function V^{k+1} in the *value function step*.

The process is iterated until the strategies for each of the firms and the value function in each of the possible states do not change more than an established convergence criterion,

²⁹For a detailed treatment of splines methods, see de Boor (2001).

³⁰Given that the cubic spline is defined by a cubic polynomial at each of the grid intervals, this implies that at most there will be $2(J-1) + 2$ candidate local optima, where J is the number of grid points.

such that $\|\sigma^{k+1} - \sigma^k\| < \epsilon_\sigma$ and $\|V^{k+1} - V^k\| < \epsilon_V$.

6 Welfare measures and metrics

In this section, we first describe the analytical framework we will use to interpret the results. We then discuss an important parameter in our analysis: the social cost of carbon.

6.1 Analytical framework

We focus exclusively on outcomes in the domestic cement industry. Within a regional cement market, static, per period welfare is defined as follows:

$$w(s, a; \alpha, \delta_i, \gamma, \tau, e) = \int_0^Q P(x; \alpha) dx - \sum_i \int_0^{q_i} C(x; \delta_i) dx - P(Q; \alpha) M(P; \gamma) - \tau \sum_i e_i q_i - \tau e^M M(P; \gamma). \quad (20)$$

The vector e includes the emissions intensity measures of both domestic producers and foreign imports. The parameter e^M denotes the emissions intensity of imports. This value is estimated using an import volume weighted average of estimated foreign cement producers' emissions intensities (Worrell et al., 2001).

This welfare measure ignores any surplus captured by the producers of domestic imports; domestic policy makers presumably ignore the economic costs and benefits accruing to producers and consumers outside their jurisdiction. In specifying this welfare function, we assume that marginal damages from carbon dioxide emissions are constant and equal to the assumed equilibrium permit price τ . Because damages from greenhouse gases are independent of where in the world the emissions occur, we penalize both domestic and foreign emissions at a rate of τ per unit. We also assume that the cement sector is small relative to the larger emissions trading program, such changes in cement industry emissions do not affect the equilibrium permit price.³¹

Each market-based policy regime we consider affects Equation 20 through its effect on firm-level production choices. Our static, single period, aggregate welfare measure sums (20) across regional markets. Welfare analysis in the dynamic simulations sums Equation 20 across markets and time periods, subtracting any entry, exit and investment costs accruing

³¹If net permit demand from the cement sector can affect the equilibrium permit price, our estimates of the costs of allocation updating, vis a vis auctioning or grandfathering, will be too low.

over the time horizon we consider. This measure of dynamic efficiency is somewhat unconventional insofar as it rules out innovation and technological change. For our purposes, a dynamically efficient outcome maximizes social welfare subject to the constraints imposed by existing and proven production technologies.

Expositionally, it is useful to decompose the net welfare impact of a policy intervention into the three components introduced in Section 2. We define three welfare measures:

- $W1$ captures changes in the private economic surplus accruing from domestic cement consumption (i.e. the first three terms in Equation 20).
- $W2$ accounts for both economic surplus changes ($W1$) plus the benefits of emissions reductions.
- $W3$ accounts for emissions leakage across policy designs by penalizing foreign increases in emissions.

In this preliminary draft, we report results from simulating outcomes in twelve out of twenty regional markets: Cincinnati, Denver, Detroit, Kansas City, Minneapolis, New York, Phoenix, Pittsburgh, Salt Lake City, San Francisco, Seattle, and St. Louis. Across regional markets we analyze, there is significant variation in market size, plant technology, and import presence. For example, the Salt Lake City market is not accessible by water and demand is met entirely by domestic suppliers, who have heterogeneity in their emissions rates. In contrast, a market like San Francisco is trade-exposed and incumbent producers are homogeneous with respect to emissions rates.

In the future, the scope of the analysis will be expanded to include all domestic cement markets. This will allow us to provide a more comprehensive assessment of the industry-wide effects of the policies we consider, and to assess the extent to which a “one-size-fits-all” policy regime can result in differential outcomes across heterogeneous regional markets.

6.2 The Social Cost of Carbon

The τ value we use to penalize each ton of CO_2 emissions is intended to capture the monetized damages associated with an incremental (one ton) increase in carbon emissions. Given the uncertainty inherent in the estimation of damages from carbon emissions, it is important to consider a range of values of τ . The range of values we choose to consider, \$5 to \$65 per ton of CO_2 , is informed by a landmark interagency process which produced estimates of the social cost of carbon (SCC) for use in policy analysis (Greenstone et al., 2011).

Table 3: Estimated social cost of carbon
(\$ per metric ton of carbon dioxide in \$2007)

	(1)	(2)	(3)	(4)
Discount rate	5%	3%	2.5%	3%
2010	4.70	21.40	35.10	64.90
2020	6.80	26.30	41.70	80.70
2030	9.70	32.80	50.00	100.00

Source: U.S. Department of Energy (2010), “Final Rule Technical Support Document (TSD): Energy Efficiency Program for Commercial and Industrial Equipment: Small Electric Motors,” Appendix 15A (by the Interagency Working Group on Social Cost of Carbon): “Social Cost of Carbon for Regulatory Impact Analysis Under Executive Order 12866”.

Table 3 summarizes the four SCC schedules that were selected in this process. In light of disagreements about the appropriate choice of interest rate, three different discount rates are used (corresponding to the first three schedules). The final schedule (fourth column) corresponds to a scenario with higher than expected economic costs from climate change. The SCC increases over time because future emissions are expected to produce larger incremental damages as physical and economic systems become more stressed.³²

For expositional ease, we will assume that the carbon price reflects the true social cost of carbon. Thus, the carbon tax or permit price and the social cost of carbon are assumed to be one and the same. In section 7, we conduct auxiliary analysis in which we hold the assumed SCC value constant across scenarios associated with different permit prices/tax levels.

7 Simulation results

Before turning to the simulation results, it is important to highlight some underlying assumptions and to issue some caveats with respect to interpretation.

One assumption underpinning our counterfactual simulations is that cement producers will respond to economic changes induced by a market-based emissions regulation in the same way that they have responded historically to similar changes induced by other exogenous

³²In our analysis, we assume the carbon price does not change over the time horizon we consider.

market forces. This assumption seems quite plausible given that the policy designs we consider operate solely through altering production costs and revenues.

Another important assumption is that our structural assumptions, including assumptions about how firms respond to changes in policy incentives, will hold out of sample. We observe significant variation in plausibly exogenous supply and demand shifters across regional markets and across time; this variation is essential to identification. However, our inferences at high carbon prices are quite far from historical experience. To put this in context, consider that a carbon price of \$60/ton would approximately triple the estimated marginal operating costs of the average cement producer. A higher-level concern is that plausible general equilibrium effects of high carbon prices could lead to unforeseen structural shifts in the supply and demand curves characterizing cement outcomes.

In what follows, we have elected to report simulation results for the range of SCC values that have been deemed policy relevant. However, it is important to keep in mind that the higher the carbon price we consider, the farther out counterfactual is from the data we observe, and the more sensitive our simulation results will be to our modeling assumptions. That said, we believe that our results both help illustrate the general forces shaping the interaction of market structure and carbon regulation and provide the best possible estimates of efficiency and distributional welfare effects under a range of policies.

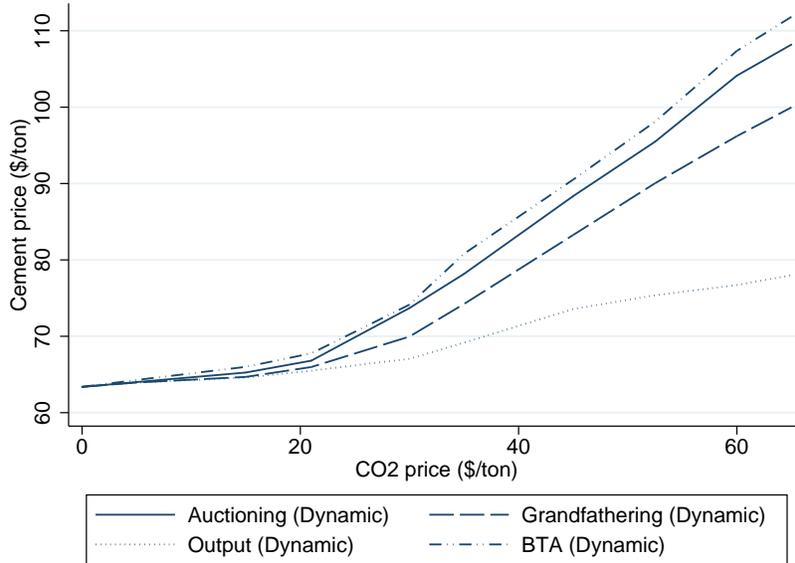
This section begins with a summary of the simulated static and dynamic responses of the domestic cement industry to the range of counterfactual policy interventions we consider. We begin by emphasizing the results using the model that accounts for industry dynamics. We then contrast our welfare measures across the static and dynamic simulation exercises. We also include a section that highlights the heterogeneous impacts of the policy across markets and a section discussing the implications for optimal carbon prices. We conclude with several robustness checks.

7.1 Market outcomes and aggregate welfare measures

7.1.1 Cement prices

Figure 3 plots quantity-weighted average cement prices as a function of the exogenous permit price (or emissions tax) τ . The introduction of market-based emissions regulation increases equilibrium cement prices across the range of τ values we consider. Cement price increases are most pronounced under the BTA and standard auction/tax regime. Under this policy design, firms must bear the complete cost of compliance; no compensation in the form of

Figure 3: Counterfactual: Cement Prices

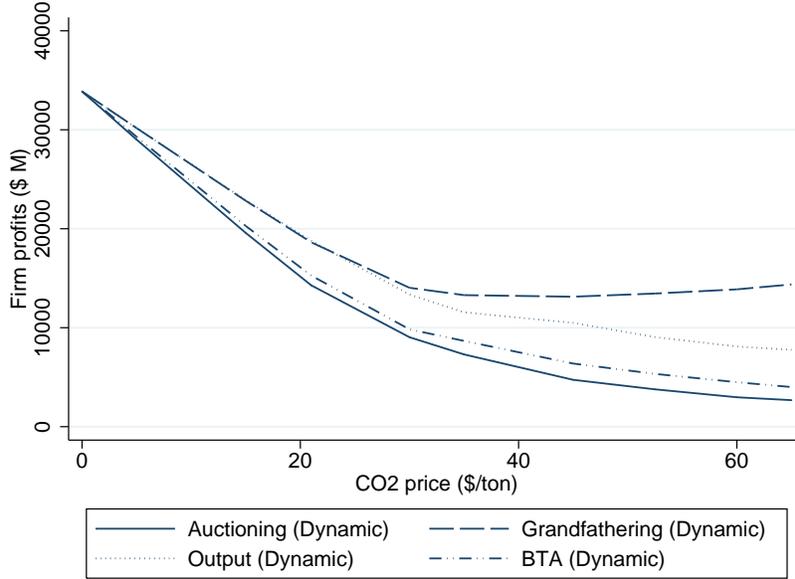


contingent rebates or lump sum transfers is offered. The prices are highest in the case of BTA, given that both domestic and international firms face the full cost of the emissions.

Note that equilibrium cement prices vary across grandfathering and auctioning regimes. Thus, the so-called independence property fails to hold when industry dynamics are incorporated into the model. Under the grandfathering regime, an incumbent firm receives a lump sum transfer each period in the form of free permit allocation. An incumbent firm forfeits this entitlement when it chooses to exit. This lowers the exit threshold for incumbents such that exit rates are lower under grandfathering as compared to auctioning. As a consequence, cement markets are less concentrated at higher permit prices, and equilibrium cement prices are lower compared to the standard auctioning/emissions tax case.

A striking feature of Figure 3 is that, for a given value of τ , cement prices are much lower under the policy regime that incorporates dynamic rebating. The production subsidy that is implicitly conferred by rebating partially mitigates the impact of the emissions regulation on cement prices.

Figure 4: Counterfactual: Cement Profits



7.1.2 Industry profits

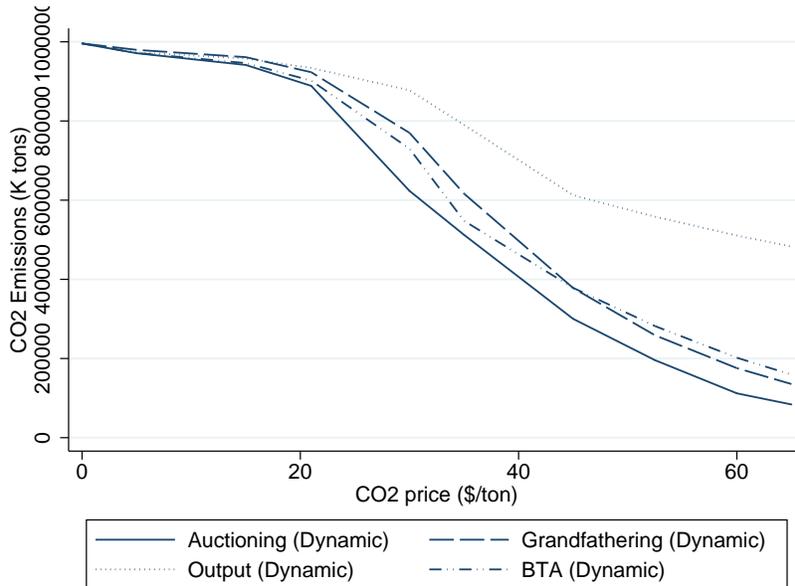
Figure 4 plots the present discounted profits earned by the regulated domestic cement producers over the range of carbon values we consider. For any given value of τ , profits are most significantly impacted by the auctioning regime because firms must pay the tax (or hold permits) to offset emissions, but receive no rebate or compensation for incurring these costs. The BTA mitigates these impacts on domestic producer profits.

Note that discounted industry profits are increasing with τ over the range of higher carbon values in the grandfathering regime. As the carbon price increases, so does the value of the lump sum transfer (in the form of free permits) allocated to incumbent firms. At very high permit prices, some firms will have an incentive to sell permits versus using them to offset their own emissions. This revenue from selling unused permits explains the non-monotonic and increasing (in τ) discrepancy in profits across rebating and grandfathering regimes.

7.1.3 Domestic emissions

Policy makers are very concerned about how industry emissions will be impacted by alternative forms of market-based emissions regulation. Figure 5 shows how emissions from domestic cement producers, summed across all markets and time periods, decreases with τ .

Figure 5: Counterfactual: Domestic Emissions



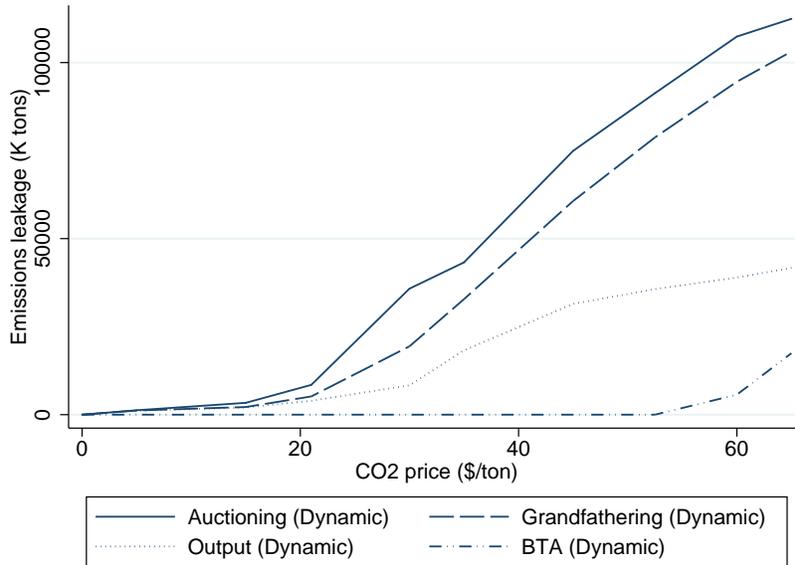
For a given carbon price, the net cost of emitting carbon dioxide (as perceived by firms), is highest under the auctioning regime and lowest under contingent rebating. Consequently, industry emissions are lowest under the auctioning regime and highest under the rebating regime. Whereas firms perceive the same costs of emissions under BTA and auctioning, domestic emissions are higher under BTA because domestic firms are better able to defend their market share under the emissions regulation.

7.1.4 Emissions leakage

In the case of trade-exposed emissions-intensive industries, the potential for emissions leakage is a serious issue. Figure 6 plots the simulated leakage under each policy scenario. Our results suggest that there is potential for significant leakage in the US cement industry. Intuitively, auctioning leads to the highest amount of leakage because it places the highest cost burden on domestic producers. In the long-run, increased costs influence both the intensive margin through reduced production and the extensive margin as the rates of exit are highest under auctioning.

In line with the earlier discussion, grandfathering slows the rate of exit vis a vis auctioning, thus slowing the rate of leakage. In contrast, the BTA reduces leakage completely over a

Figure 6: Counterfactual: Emissions Leakage



large range of carbon values. At very high values, the policy does induce an increase in emissions from imports due to high rates of exit in the domestic sector.

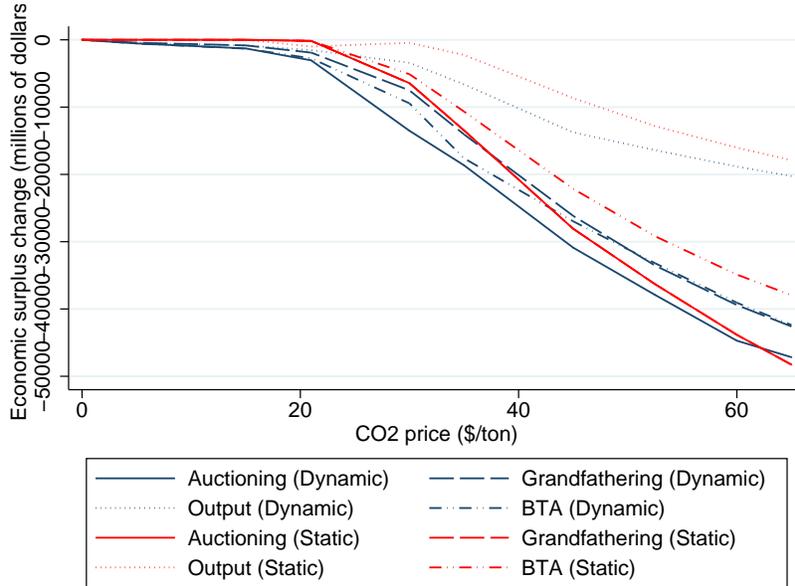
The results also demonstrate that output-based rebating significantly mitigates emissions leakage. The rebate incentivizes relatively high levels of domestic production, thus limiting the extent to which imports outcompete domestic production in trade exposed markets. At the extensive margin, incumbents are more valuable under dynamic updating in comparison to auctioning, which helps keep them active in the market, further decreasing leakage.

7.1.5 Decomposing Changes in Welfare

Our fundamental objective is to investigate the welfare implications of the alternative policy designs we consider. In what follows, the emissions unconstrained (i.e. unregulated) case serves as a benchmark. We present the three welfare metrics introduced in the previous section, decomposing along conceptually distinct lines: product market welfare consisting of producer profits, consumer surplus, and government revenues; benefits accruing to emissions reductions; and costs due to emissions leakage.

To highlight the importance of accounting for industry dynamics, we contrast the results of our dynamic simulations with a simulation exercise that holds fixed industry structure and

Figure 7: Counterfactual: W1



technology characteristics. A common practice in ex ante policy analysis involves simulating regulatory effects in a static setting, using a representative year as the basis for estimating annual regulatory impacts, and then using that test year to extrapolate outcomes over a longer time horizon (OAQPS, 1999). We adopt this approach here. To generate our “static” results, we simulate a single period market outcome in the unregulated baseline case and under the range of counterfactual policy designs we consider. To facilitate comparisons with our dynamic simulations, these results are expressed as net present values using a social discount rate of three percent. We assume the simulated annual outcomes would be observed each year of the 30 year time horizon we consider.

W1: Product Market Surplus Changes in the first welfare metric, W1, capture differences in producer and consumer surplus while also accounting for revenues raised by the government through taxation or permit sales. This is a measure of how the local market changes in response to the regulation, and is a major component of understanding welfare changes in concentrated industries.

Figure 7 shows how economic surplus is impacted across policy designs and assumed carbon prices. Given that this W1 measure captures none of the benefits from emissions abatement, these changes are all negative.

An interesting result to emerge from the static simulations (in red) is the lack of industry response to carbon prices at or below \$20. In the benchmark (unregulated) case, many firms are capacity constrained, producing at a corner solution, and thus are earning scarcity rents. When firms are required to internalize a relatively low emissions cost at a per unit cost of \$20 or less, scarcity rents are reduced, but output decisions are essentially unaffected in the short run.

In the static simulations, it is also the case that outcomes under the auctioning and grandfathering regimes exhibit the independence property: for any given carbon price, impacts on W1 are identical. Intuitively, this is because the short run incentives in production are identical across these two regimes.

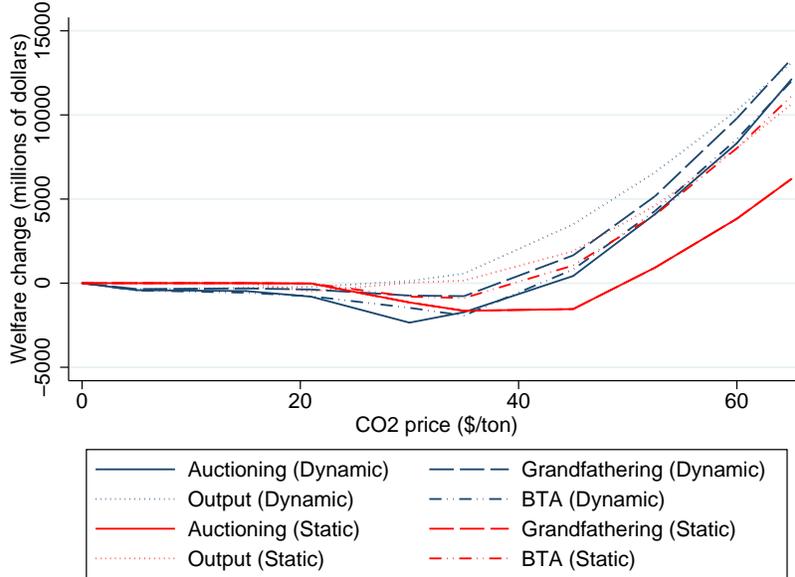
Comparisons across static and dynamic simulations highlight how the evolution of industry structure can affect policy outcomes. First, policy impacts equilibrium prices and quantities at much lower carbon prices in the dynamic simulations. This is due to the reduction in cement production capacity that the emissions regulation induces. The rate of exit is most accelerated under auctioning; firms are more likely to exit when they do not have a steady stream of freely allocated permits to look forward to. Consequently, welfare impacts are most negative under auctioning.

Second, for most of the carbon values we consider, incorporating industry dynamics leads to more pronounced negative welfare impacts. This can again be explained by the capacity/disinvestment response which is shut off in static case.

Finally, whereas outcomes under grandfathering and auctioning are indistinguishable in the static case, negative welfare impacts are mitigated somewhat under grandfathering in the dynamic simulations. This divergence is the results of two countervailing forces. On one hand, high carbon prices incentivize firms to reduce their production, which harms both consumer and producer surplus. On the other hand, grandfathered firms hold an increasingly valuable resource as carbon prices go up. This creates an incentive for firms to remain in the market (versus exiting) because they would otherwise forfeit their entitlements to free permits in the future.

W2: Accounting for Domestic Emissions Abatement Figure 7 fails to capture any of the benefits from emissions abatement. Welfare measure W2, as shown in Figure 8, adds the social benefits associated with reducing CO₂ emissions from domestic cement producers to the changes in the product market measured under W1. Recall that the value of the avoided emissions are assumed to be equal to the prevailing permit price or tax. Thus, the

Figure 8: Counterfactual: W2

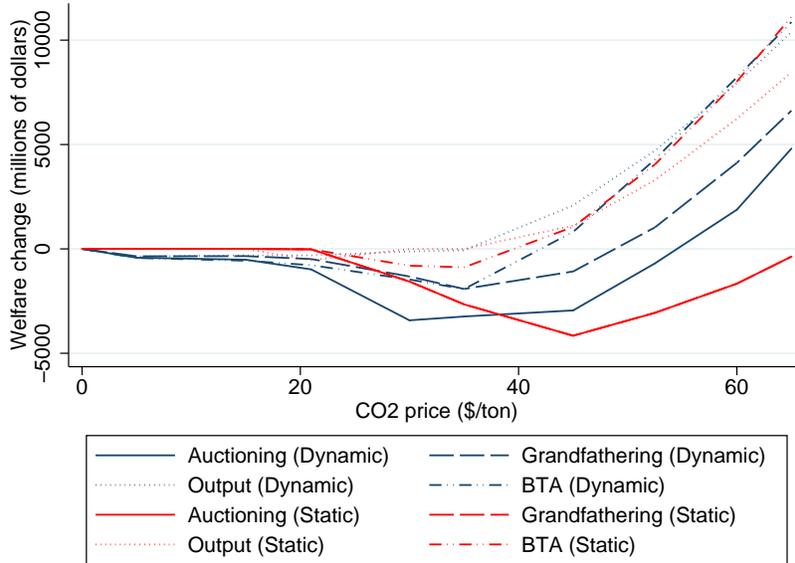


welfare adjustment per unit of emissions abated is increasing along the horizontal axis of Figure 8.

Beginning with the static simulations, the benefits from internalizing the emissions externality more than offset the economic costs under the policy regime that incorporates rebating; net welfare impacts are weakly positive across all carbon values. The same cannot be said for the grandfathering and auctioning regimes. Over the mid-range of the carbon values we consider, the net welfare impacts are negative. This result is driven by Buchanan’s observation that there are two competing distortions in concentrated markets with externalities. As the assumed social cost of carbon increases, the value of avoided damages ultimately overwhelms the value of the lost economic surplus, and the net welfare impacts turn positive.

The dynamic simulations yield somewhat different results. Welfare gains associated with rebating regimes and grandfathering in the dynamic case are larger than those generated using the static model. This is partly due to firms’ ability to invest in cleaner production equipment (which reduces the emissions intensity per unit of cement produced) and partly because the quantity produced is lower, resulting in lower damages from domestic emissions. The factors that made auctioning and grandfathering unattractive under metric W1—namely, output contraction and the accelerated exit of domestic producers—make them attractive under metric W2 once we account for the benefit of carbon emissions reductions.

Figure 9: Counterfactual: W3



W3: Accounting for Emissions Leakage Our final welfare metric, W3, augments W2 by accounting for damages associated with emissions leakage. Emissions occurring in other jurisdictions are penalized at the same rate as domestic emissions.³³

Figure 9 illustrates the welfare impacts of the policy regimes we consider using this more comprehensive welfare measure. In the static simulations, once leakage is accounted for, welfare impacts of the grandfathering and auctioning regimes are negative across the full range of carbon values we consider. In contrast, the benefits from domestic emissions reductions more than offset costs associated with emissions leakage and the excessive withholding of output and investment in the upper range of carbon values when rebates are incorporated in the policy design.

In the dynamic simulations, the net welfare impacts remain at or close to zero for all carbon values below \$35. Net welfare impacts of grandfathering and auctioning are negative for carbon values below \$50. The welfare ordering of policy regimes no longer reverses at higher carbon values. The policy designs that incorporate rebating welfare dominate over most of the range of carbon values we consider.

The benefits from the BTA mechanism stand out in Figure 9. Differences in leakage

³³Ignoring co-pollutants, damages from emissions are independent of location. This contrasts to other emissions that have spatially-varying damages. See, for example, Fowlie and Muller (2010).

Table 4: Dynamic simulation results: Social cost of carbon values \$21 and \$35/ton CO₂

Outcome	SCC Value	Base-line	Auction	Grand-father	Output-updating	BTA
Market size (tons per year)	21 35	58,831.79 58,831.79	52,120.92 37,546.95	54,209.55 52,116.94	54,821.92 47,351.27	52,989.76 39,538.35
Q. weighted price (\$/ton)	21 35	63.69 63.69	65.95 75.17	65.23 72.02	65.00 67.81	66.86 77.45
Domestic emissions (NPV tons CO ₂)	21 35	995,868.41 995,868.41	888,696.27 512,363.08	923,221.73 616,486.29	933,764.95 790,133.57	902,017.44 547,824.04
Emissions rate (tons CO ₂ /ton clinker)	21 35	0.98 0.98	0.98 0.95	0.98 0.97	0.98 0.96	0.97 0.94
Domestic firm profits (\$ M)	21 35	33,876.20 33,876.20	14,283.90 7,299.99	18,628.88 13,292.19	18,782.92 11,557.25	15,269.83 8,664.51
Change in W1 (\$ M)	21 35	0.00 0.00	-3,055.08 -18,647.44	-1,902.80 -14,049.98	-1,533.27 -6,628.99	-2,750.24 -17,606.91
Change in W2 (\$ M)	21 35	0.00 0.00	-804.47 -1,724.75	-377.22 -771.60	-229.10 571.73	-779.37 -1,925.36
Change in W3 (\$ M)	21 35	0.00 0.00	-981.99 -3,238.77	-485.85 -1,919.95	-311.68 -68.22	-779.37 -1,925.36

between the BTA and the auctioning mechanism increase the gap between the two mechanisms, making the BTA mechanism more attractive. For the highest carbon price that we consider (\$65), this is the preferred mechanism.

To help summarize this discussion, Table 4 reports key results from dynamic simulations which assume carbon values of \$21 and \$35, respectively. At these carbon values (and carbon prices), rebating regimes welfare dominate auctioning, grandfathering, and the BTA. Intuitively, the benefits from rebating (mitigation of the exercise of market power and emissions leakage in trade-exposed markets) outweigh the costs (dampened incentives for emissions abatement).

7.2 Heterogeneous impacts of environmental regulation

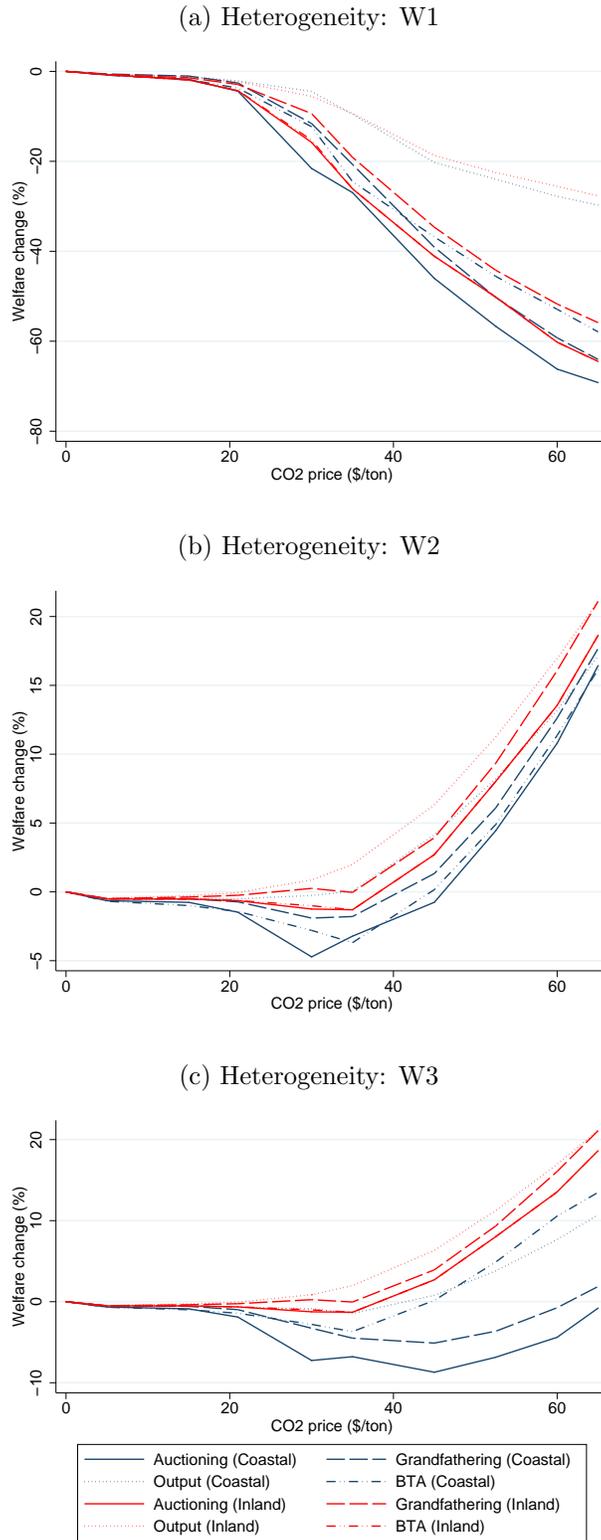
The simulation results allow us to examine the impact of a federal environmental regulation on local markets. In the debates over carbon policy design and implementation, it is typically assumed that different industries will be treated differently (in terms of permit allocations, compliance requirements, etc.). But it is generally assumed that domestic firms within a given sector will be treated symmetrically. For example, in the case of the cement industry, implicit output-based updating mechanisms would be implemented in the same way across all regional cement markets. However, given the differences in the industry composition of local markets, as well as the differences in trade exposure, the costs associated with a seemingly uniform federal policy can differ significantly across regions.

Intuitively, prices increase more rapidly in inland markets, as firms do not face competition from imports and can pass-through more of the costs of the emissions. The effect is particularly striking for higher carbon prices, in which coastal markets reach a threshold price in which all local demand is served by imports from unregulated areas whereas inland markets still experience high price increases. As a consequence, firm profits are more negatively impacted in trade exposed regions. Even though this is a quite extreme finding, it highlights one of the major differences between coastal and inland markets.

Figure 10 represents the welfare differences among coastal and inland markets. In order to re-normalize the measures across coastal and inland markets, the welfare measure represents the percentage change in welfare with respect to the baseline. Economic surplus is more significantly impacted in inland markets. This is due to the fact that demand is inelastic and there are no imports to serve the market. Panel (b) shows that accounting for emissions reductions in the market has similar effects to both inland and coastal markets, relatively favoring grandfathering and auctioning with respect to updating mechanisms due to the full internalization of emissions costs and, thus, lower emissions. For inland markets, the net benefits of the carbon policy in the industry, accounting for the positive externalities of emissions reduction, is much more favorable than for coastal markets.

Finally, W3 shows that the net benefits of the policies we consider are smaller-or more negative- in coastal markets as compared to inland markets. Intuitively, the existence of import competition limits the extent to which domestic producers can pass-through compliance costs. Reductions in domestic producer profits and emissions leakage works against the gains (in terms of emissions reductions) in these coastal markets.

Figure 10: Counterfactual: Inland versus Coastal Welfare



7.3 Carbon Prices and the Social Costs of Carbon

One important assumption that we have maintained throughout our analysis is that the permit price reflects the social cost of carbon. There are two reasons why one may wish to relax this assumption. First, our results suggest that the negative welfare effects of imposing the full social cost on the cement industry may outweigh the benefits from mitigation; as a result, it may be optimal to set a permit price that is somewhat less than the true social cost. This insight helps explain why the updating mechanism is particularly attractive relative to the alternative mechanisms—it offers an implicit production subsidy to the industry which lowers the effective permit price below the social marginal cost. Our model allows us to solve for the optimal permit price for any social cost of carbon. Second, the political process which sets the cap for emissions may result in a permit price that is below the actual cost of carbon. It is therefore an interesting and relevant question to ask what the welfare consequences of having a permit price below the social cost of carbon would be for a range of damages.

Table 5 reports the results for two cases. In the first scenario, we compute changes in welfare as we move carbon prices while holding the true social cost of carbon at \$21 per ton of CO₂. The striking outcome is that the optimal permit price is zero for all mechanisms. This suggests that it would be inefficient to abate carbon in the cement industry at this cost range.

The picture is different if we set the social cost of carbon to be \$35 per ton. In this case, it can be welfare improving to induce abatement in the cement industry, although still not at the full social cost of carbon. The optimal permit price ranges from a low of \$15 per ton for auctioning to a high of \$22.5 for both output-based updating and border tax adjustments. The gains in total welfare are much more substantial at a price of \$35 per ton than the essentially zero gains found at \$20 per ton; however, the welfare differences across mechanisms are not as large. In order of performance, border tax adjustments perform best, followed by the updating, auctioning, and finally grandfathering.

Comparing the relative magnitudes of welfare losses under the full social cost of carbon is enlightening: the border tax adjustment does so poorly at the true social cost in large part because it restricts product market output too stringently. Under an optimal permit price, however, the border tax adjustment is able to successfully balance the concerns of domestic market power, leakage, and emissions mitigation. These results further highlight the complex interplay of forces at work in the market.

Table 5: Optimal carbon prices for different mechanisms

	Optimal Carbon Price	Welfare if Optimal Carbon Price	Welfare if Carbon Price = SCC
SCC = \$21			
Auctioning	0.0	0.0	-8919.6
Grandfather	0.0	0.0	-6454.8
Output	0.0	0.0	-2384.1
BTA	0.0	0.0	-7783.6
SCC = \$35			
Auctioning	15.0	471.6	-3133.5
Grandfather	20.0	423.8	-1851.7
Output	22.5	474.0	67.8
BTA	22.5	516.9	-1864.3

Notes: Table for a subset of regional markets with four or less firms. Optimal carbon prices computed on a grid of \$2.5 increments.

7.4 Robustness checks

7.4.1 Demand and import elasticities

Demand and import supply elasticities are important parameters that affect the computation of welfare measures. Unfortunately, publicly available data on producer prices, production quantities, import quantities, etc. are quite noisy. This results in elasticity estimates which are imprecise. In order to assess the robustness of our results, we simulate a subset of markets using a range of elasticity values.

The demand elasticity is a key determinant of the consumer gross surplus in the model and therefore an important element in our welfare measures. The qualitative results are robust to a wide range of demand elasticity values. Table 6 in Appendix C presents W3 welfare differences for a range of carbon prices and demand elasticities. As one would expect, we find that the negative effects are more striking when demand is more inelastic, due to the Buchanan effect. Negative values are attenuated, or turn positive, for more elastic demand curves.

We also consider a wide range of import supply elasticities. Our import supply function is assumed to be exogenous. This fails to capture the possibility that importing firms could respond to increasing domestic cement prices by investing in more efficient production or transport practices. By allowing for a more or less responsive supply curve, we capture in a

crude way the possibility of different short-run import responses. Overall, we find that the qualitative results are unaffected by changing the import supply elasticity.

7.4.2 Differential Changes in Cement Substitutes

Table 6 also helps us address questions about how the outside options for cement consumption change as the carbon price increases. A potential concern with our approach is that our model effectively holds constant demand shifters. A carbon tax or cap and trade program would affect the supply markets for substitutes to cement in different ways. For example, asphalt is a substitute for cement in paving applications, and an important issue is whether or not that product will become differentially more or less expensive than cement as the price of carbon increases.

Explicitly modeling these intermarket interactions would involve the specification and estimation of a more general equilibrium model. This is well outside the scope of this paper. However, one can use Table 6 to get a rough idea of how our results may change. If it is believed that cement will become differentially more expensive as carbon prices rise, one can simply start the the baseline elasticity at the zero carbon price and trace down the table, letting the elasticity increase with the carbon price. While this is not as satisfying as an exercise that explicitly models interactions between climate policy and markets for cement substitutes, it is at least a simple way of representing the degree of sensitivity of our results to our partial equilibrium modeling assumptions. In general, if cement demand becomes more elastic with carbon prices, the negative welfare results tend to increasingly hold. On the other hand, if cement becomes more inelastic, then the benefits of implementing carbon schemes in the cement industry become increasingly welfare-improving, primarily as the Buchanan welfare effects are relatively smaller.

7.4.3 Output- vs. Emissions-based updating

As a robustness check, we consider emissions-based rebating. This works in precisely the same way as output-based rebating, except that rebates (in the form of recycled tax revenues or free emissions permits) are allocated based on emissions. The more emissions intensive a firm, the larger the rebate (per unit of output) it receives. This design has been proposed in cases where firms owning older, less efficient kilns insist that they should be entitled to a larger allowance allocation so as to compensate them for their higher compliance costs.

Outcomes differ only slightly across the output-based and emissions-based updating regimes. The reason is two-fold: first, new firms enter at a fixed frontier emissions rate,

so as the industry turns over we asymptote towards having identical outcomes under both policies. Second, among incumbents, the only margin for differences between the two is differential reorganization of production across units with different emissions intensity. This margin is relatively small compared to the overall contraction in market quantity; as such, differences between the two policies are masked by the overall market changes.

8 Conclusion

We present a dynamic model to evaluate the welfare impacts of market-based regulation of carbon dioxide emissions in the US cement industry. We assess the implications of several alternative policy designs, including those that incorporate both an emissions disincentive (a tax or an obligation to hold an emissions permit) and a production incentive. Simulation results reported in this working paper pertain to only a subset of regional markets. The analysis will ultimately include the entire domestic cement sector.

We find that both the magnitude and the sign of the welfare impacts we estimate depend significantly on how the policy is implemented and what we assume for the social cost of carbon. At low to moderate carbon values, our results echo [Buchanan \(1969\)](#). Market-based emissions regulation that internalizes the full emissions externality leads to small social losses. These losses are exacerbated by emissions leakage in trade exposed regional markets. At higher carbon values, our results are more in line with [Oates and Strassman \(1984\)](#) who argue that the welfare gains from pollution control will be large relative to losses associated with output contraction and the exercise of market power. Notably, we find that policy designs that incorporate both an emissions penalty and a production incentive in the form of a rebate welfare dominate more conventional policy designs. Intuitively, the production incentive works to mitigate leakage in trade exposed cement markets and the distortion associated with the exercise of market power.

Of course, these simulation results condition on the structural assumptions that define the underlying model. The higher the carbon price we consider, the farther out counterfactual is from the data we observe, and the more sensitive our simulation results will be to our modeling assumptions.

Policy makers are very interested in understanding how proposed climate change policies would impact strategic, emissions intensive sectors such as the cement industry. The scale and scope of these policy interventions are unprecedented, making it difficult to anticipate how industry will respond and what that response will imply for social welfare. This paper

illustrates important forces that shape the interaction of industry structure, trade flows, and proposed carbon regulations. Simulation results provide important insights into the efficiency and distributional properties of leading policy design alternatives.

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A Construction of Emissions Rates

Over half of the emissions from clinker production come from the chemical reaction that occurs when the calcium carbonate in limestone is converted into lime and carbon dioxide. To measure carbon dioxide emissions from calcination accurately, emissions factors can be determined based on the volume of the clinker produced and the measured CaO and MgO contents of the clinker. In the absence of this detailed plant-level information, we assume a default rate of 0.525 metric tons of carbon dioxide/metric ton of clinker (WBC, 2005).

The other major source of carbon dioxide emissions from clinker production is fossil fuel combustion. The preferred approach to estimating CO₂ emissions from fuel combustion requires data on fuel consumption, heating values, and fuel specific carbon dioxide emission factors. Although the Portland Cement Association (PCA) does collect plant level data regarding fuel inputs and fuel efficiency (i.e. BTUs per ton of cement), these data are disaggregated data are not publicly available. We do have data aggregated by kiln type and vintage. We use these data (reported in 2006), together with average carbon dioxide emissions factors, provided by the U.S. Department of Energy, to estimate kiln technology specific emissions intensities.

We consider three classes of kilns in particular: wet process kilns (i.e. older, less efficient technology), dry process kilns with preheater/precalciner, and a best practice energy intensity benchmark (Coito et al., 2005)³⁴ Because of the dominant role played by coal/pet coke, our benchmark emissions calculations are based on coal/petcoke emissions factors. We assume an emissions factor of 210 lbs carbon dioxide/mmbtu.³⁵

Our technology-specific emissions rate calculations are explained below. To put these numbers in perspective, the national weighted average emissions rate was estimated to be 0.97 tons carbon dioxide/ton cement in 2001 (Hanle et al, 2005).

Wet process In 2006, there were 47 wet process kilns in operation. On average, wet kilns produced 300,000 tons of clinker (per kiln) per year. The PCA 2006 Survey reports an average fuel efficiency of 6.5 mmbtu/metric ton of clinker equivalent among wet process kilns.

³⁴The industry has slowly been shifting away from wet process kilns towards more fuel-efficient dry process kilns. On average, wet process operations use 34 percent more energy per ton of production than dry process operations. No new wet kilns have been built in the United States since 1975, and approximately 85 percent of U.S. cement production capacity now relies on the dry process technology.

³⁵Fuel-specific emissions factors are listed in the Power Technologies Energy Data Book, published by the US Department of Energy (2006). The emissions factors (in terms of lbs CO₂ per MMBTU) for petroleum coke and bituminous coal are 225 and 205, respectively. Here we use a factor of 210 lbs CO₂/MMBTU. This is likely an overestimate for those units using waste fuels and/or natural gas.

The relevant conversion is then $0.095 \text{ metric tons carbon dioxide/mmbtu} * 6.5 \text{ mmbtu/metric ton of clinker equivalent} = 0.62 \text{ tons carbon dioxide/ton clinker}$. When added to process emissions, we obtain our estimate of 1.16 tons carbon dioxide/ton clinker.

Dry process In 2006, there were 54 dry kilns equipped with precalciners with an average annual output of 1,000,000 tons of clinker per year. The PCA 2006 Survey reports an average fuel efficiency of 4.1 mmbtu/metric ton of clinker equivalent among dry process kilns with precalciners. Thus, $0.095 \text{ metric tons carbon dioxide/mmbtu} * 4.1 \text{ mmbtu/metric ton of clinker equivalent} = 0.39 \text{ tons carbon dioxide/ton clinker}$. Adding this to process emissions results in the estimate for dry-process kilns: 0.93 tons carbon dioxide/ton clinker.

Frontier technology To establish estimates for new entrants, a recent study (Coito et al, 2005) establishes a best practice standard of 2.89 mmbtu/ metric ton of clinker (not clinker equivalent). The calculation is then: $0.095 \text{ metric tons carbon dioxide/mmbtu} * 2.89 \text{ mmbtu/metric ton of clinker equivalent} = 0.275 \text{ tons carbon dioxide/ton clinker}$. Adding this to process emissions obtains in 0.81 tons carbon dioxide/ton clinker for new kilns.³⁶

B Abatement response

In the simulation exercise, the state space is modified such that emissions rates vary systematically across plants of different vintages and technology types. Incumbent firms are classified as either wet-process, dry-process, or dry-process with precalciner/preheaters. New kilns are assumed to be state-of-the-art. This modification allows us to crudely capture changes in embodied emissions intensity as the industry evolves.

There are four main strategies for reducing the carbon intensity of domestic cement industry. First, it is anticipated that capital stock turnover will be a major driver of emissions intensity reductions (Worrell, 1999). Replacing old wet-process kilns with state-of-the-art dry kilns could deliver significant reductions in combustion-related emissions.

Second, the carbon intensity of clinker production can also be reduced via fuel switching. Currently, coal and petroleum coke are overwhelmingly the dominant fuel used in pyroprocessing and electricity is used to grind raw materials into kiln feed. Most domestic kilns are

³⁶This is very similar to the CO₂ emissions rate assumed in analyses carried out by California's Air Resources Board in 2008 under a best practice scenario that does not involve fuel switching. If fuel switching is assumed, best practice emissions rates drop as low as 0.69 MT CO₂/ MT cement. See [NRDC Cement GHG Reduction Final Calculations](#).

capable of burning a variety of fuels in principle, although fuel switching can adversely affect plant performance.

Third, concrete manufacturers have the capacity to partially substitute SCMs for clinker inputs. The advantage of this emissions reduction strategy is that, by reducing the use of clinker, carbon emissions from both fuel combustion and calcination are eliminated. Finally, cement manufacturers have some capacity to substitute less carbon intensive raw materials for limestone. A more detailed discussion is relegated to the robustness section.

Data limitations will prevent us from being able to model input and fuel substitution capabilities accurately at the plant level. In our model, these two abatement options are ignored. In the policy simulations, carbon dioxide emissions from the domestic cement industry can be reduced via four channels: accelerated capital turnover (i.e. retirement of older kilns and investment in newer, more efficient operations), a reallocation of production from more to less emissions intensive incumbents, an increased reliance on imports, and a decrease in domestic clinker consumption. To the extent that fuel and input substitution are economically viable and cost effective compliance alternatives, our results will over estimate compliance costs and thus should be interpreted as upper bounds.

C Robustness Tables

Table 6: Differences in welfare with respect to baseline (W3) for different demand elasticities

	5.0	15.0	21.0	30.0	35.0	45.0	52.5	60.0	65.0
$\eta = 1.5$									
Auctioning	-56.2	-656.6	-1807.2	-2917.9	-2993.9	-3918.4	-4403.8	-4308.1	-3929.4
Grandfather	-96.8	-593.7	-1249.5	-2098.5	-2356.0	-2784.3	-2981.3	-3037.9	-2770.2
Output	-41.6	-202.2	-456.0	-1062.1	-937.9	-350.6	515.5	1553.7	2142.0
BTA	-93.5	-733.3	-1427.5	-1901.7	-1862.9	-949.9	13.5	1301.8	2291.2
$\eta = 2.0$									
Auctioning	-43.1	-222.4	-669.0	-2305.3	-2205.0	-3003.3	-3477.4	-2885.3	-2782.3
Grandfather	-35.4	-77.1	-497.3	-1497.3	-1743.3	-2043.5	-2108.4	-1800.2	-1328.1
Output	-32.2	-60.5	-79.3	-583.1	-673.8	128.1	1182.4	2313.7	2966.7
BTA	-77.8	-282.1	-671.3	-1459.2	-1441.2	-408.6	682.5	1983.9	3070.7
$\eta = 2.5$									
Auctioning	-72.8	-204.6	-287.1	-1837.1	-1800.2	-2989.3	-2740.0	-2039.6	-1419.5
Grandfather	-69.3	-101.9	-134.0	-1052.2	-1292.1	-1558.0	-1447.6	-842.7	-187.2
Output	-64.1	-70.5	-73.3	-54.8	-299.3	503.9	1594.7	2870.9	3636.1
BTA	-102.5	-256.6	-304.4	-1051.6	-1144.8	-227.7	950.8	2526.0	3702.2
$\eta = 3.0$									
Auctioning	-337.7	-243.1	-255.3	-1513.0	-1538.3	-2576.3	-1827.4	-943.9	427.1
Grandfather	-290.2	-199.6	-132.8	-619.0	-841.0	-911.0	-619.5	293.7	1177.4
Output	-277.6	-169.6	-78.5	124.7	52.6	979.2	2252.4	3630.1	4602.4
BTA	-317.3	-302.0	-270.2	-586.6	-950.5	210.4	1582.2	3432.8	4465.6
$\eta = 3.5$									
Auctioning	-1243.1	-707.8	-457.0	-1115.1	-853.9	-1183.4	174.5	1806.2	3602.7
Grandfather	-1125.9	-710.9	-341.0	-30.4	-156.5	320.5	1093.3	2613.3	3934.2
Output	-1224.3	-688.7	-288.2	365.1	573.9	2062.9	3754.2	5646.2	6867.9
BTA	-1246.8	-743.1	-408.4	-84.9	-218.6	1500.6	3432.2	5350.4	6991.1
$\eta = 4.0$									
Auctioning	-1179.7	-771.0	-489.0	-909.0	-806.4	-866.1	481.3	2541.1	4444.4
Grandfather	-1185.5	-855.8	-375.0	10.8	-67.3	461.5	1348.5	3043.8	4606.0
Output	-1154.0	-698.0	-327.6	333.4	710.2	2024.8	3809.2	5703.4	6856.0
BTA	-1182.2	-1682.5	-441.6	8.8	-320.7	1625.5	3190.2	5696.2	7130.2

Notes: Table reports average differences in welfare for a subset of regional markets with three or less firms (Cincinnati, Minneapolis, Pittsburgh, Salt Lake City, Seattle).

Table 7: Differences in welfare with respect to baseline (W3) for different **import elasticities**

	5.0	15.0	21.0	30.0	35.0	45.0	52.5	60.0	65.0
$\eta = 1.5$									
Auctioning	-176.3	-117.8	-158.9	-1345.8	-1215.9	-2221.7	-3814.0	-884.4	-51.7
Grandfather	-176.3	-103.4	-59.7	-760.3	-838.7	-1080.2	-850.2	-84.6	630.7
Output	-172.4	-100.7	-61.6	-22.7	-311.2	446.4	1260.3	2217.7	2705.8
BTA	-198.9	-158.5	-191.9	-818.2	-809.5	-19.0	887.0	1682.9	2594.0
$\eta = 2.0$									
Auctioning	-180.8	-119.5	-158.9	-1433.3	-1283.2	-2459.7	-2048.4	-1573.1	-668.3
Grandfather	-180.0	-108.0	-61.8	-489.6	-866.5	-1219.4	-1208.9	-664.1	-87.9
Output	-176.1	-106.0	-65.5	-10.6	-274.5	342.8	1133.9	2055.9	2530.1
BTA	-203.5	-171.6	-215.5	-701.8	-785.5	34.7	951.3	2065.1	2706.9
$\eta = 2.5$									
Auctioning	-236.5	-127.4	-164.8	-1463.4	-1344.6	-2667.5	-2398.6	-2096.4	-1205.3
Grandfather	-195.5	-117.8	-68.7	-622.9	-886.0	-1340.6	-1515.4	-1198.0	-755.6
Output	-188.3	-115.8	-72.0	-19.0	-212.2	251.2	996.0	1808.2	2389.3
BTA	-216.0	-186.0	-185.0	-537.0	-756.9	119.2	1010.9	2280.3	2833.2
$\eta = 3.0$									
Auctioning	-337.8	-168.2	-195.3	-1080.5	-1393.3	-2824.7	-2689.0	-2513.8	-1651.6
Grandfather	-253.5	-160.5	-343.1	-710.8	-907.1	-1427.6	-1786.8	-1671.8	-1332.4
Output	-306.3	-157.9	-101.6	-37.0	-216.6	191.3	940.2	1742.7	2338.2
BTA	-315.5	-246.4	-194.0	-509.3	-723.6	186.1	1088.4	2400.8	3199.9
$\eta = 3.5$									
Auctioning	-385.2	-275.2	-728.2	-1250.6	-1315.2	-2753.2	-2689.4	-2579.5	-1690.9
Grandfather	-343.2	-182.2	-78.6	-360.8	-764.4	-1296.0	-1775.0	-1820.0	-1487.6
Output	-367.8	-191.6	-83.4	61.3	-126.2	323.0	1141.3	1952.8	2639.9
BTA	-383.7	-219.5	-146.7	-319.6	-516.0	454.8	1434.7	2807.6	3667.7
$\eta = 4.0$									
Auctioning	-494.3	-294.5	-249.3	-1430.2	-1379.3	-2882.5	-2931.5	-2904.0	-2000.6
Grandfather	-494.3	-287.7	-155.0	-378.5	-809.8	-1365.8	-1999.0	-2245.8	-1856.0
Output	-488.7	-286.3	-156.5	16.3	-83.5	259.8	1057.6	1966.0	2595.2
BTA	-520.0	-297.3	-195.8	-331.2	-506.4	502.3	1501.3	2897.6	3925.5

Notes: Table reports average differences in welfare for a subset of regional markets with three or less firms (Cincinnati, Minneapolis, Seattle).