Cost, innovation, and emissions leakage from overlapping climate policy in California

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Abstract

Jurisdictions have implemented a variety of policy instruments to mitigate greenhouse gas emissions. However, interactions between overlapping climate policies can lead to unintended impacts. This study examines how interactions between two policies in California, the low-carbon fuel standard and cap-and-trade program, impact emissions, costs, and innovation. Simulations using a computable general equilibrium model suggest that interactions between an LCFS and an emissions cap can result in *higher* emissions and *higher* average abatement costs relative to an emissions cap alone. Emissions increase as a result of the LCFS incentivizing greater production of alternative transportation fuels with upstream production emissions in sectors not covered by the emissions cap. Inter-industry emissions leakage can be mitigated by incorporating elements of a fixed-price instrument (i.e. carbon price floor/ceiling) to improve policy complementarity or requiring an obligation for the lifecycle GHG emissions of fuels under the emissions cap.

Summary

To address climate change, jurisdictions have implemented a variety of policy instruments to mitigate greenhouse gas emissions. However, the use of overlapping climate policies may lead to unintended interactions that can undermine policy objectives. This study examines how interactions between two policies implemented in California – the low-carbon fuel standard (LCFS) and cap-and-trade (CAT) program – impact abatement costs, innovation, and total emissions. Previous research has shown how additional policies, when applied to a sector covered by a cap-and-trade program, are unlikely to create additional emissions abatement. In this study, I demonstrate how the use of an LCFS in combination with a CAT program can lead to an *increase* in net emissions as well as higher average abatement costs, relative to a CAT alone.

These undesirable outcomes occur as a result of two dynamics: (i) using additional policy to force high-cost abatement from a sector that is covered under the emissions cap increases total policy costs without altering total abatement achieved under the cap, and (ii) when the production of some fuels regulated by the LCFS also generate emissions in sectors not covered under the cap, the net emissions impact from the policy combination depends on the change in emissions outside of the cap incentivized by the LCFS.

To estimate the magnitude of these effects, I simulate a range of policy scenarios using a computable general equilibrium model for California developed for this study. Results suggest that the overlapping policy approach can result in 2.4% higher total emissions and 9% higher average abatement costs from 2015 to 2030, relative to a scenario with an emissions cap alone. While induced technological change from the LCFS lowers policy costs, particularly from lower production costs of bio-based fuels, it is unlikely to offset the higher efficiency costs from overlapping regulation.

In practice, California's cap-and-trade program functions as a hybrid policy instrument incorporating a floor price below which emissions allowances will not be sold and a ceiling price above which unlimited allowances may be sold. When the price of emissions allowances is constrained by the floor or ceiling, the emissions-reduction impact of the LCFS becomes additive because the price incentive remains unchanged in the presence of the LCFS. This results in 1.25% greater cumulative abatement from the LCFS, although at a higher average abatement cost. In this case, complementary abatement from the LCFS that occurs when the floor or ceiling is binding is found to more than offset inter-industry leakage that occurs when the emissions cap is binding. However, the price collar also limits overall emissions reductions by allowing unlimited allowances to be purchased at the ceiling price.

Additional policy alternatives capable of mitigating the negative interaction effects are explored. Emissions leakage could be avoided by incorporating lifecycle GHG emissions for alternative fuels under the CAT program. Alternatively, using a purely price-based instrument such as a carbon tax instead of a CAT program could improve the complementarity between instruments in the policy mix. However, policy costs from overlapping policies remain higher than with a carbon price alone.

1. Introduction

Reducing greenhouse gas (GHG) emissions to mitigate the impacts of climate change is a pressing global priority. To do so, jurisdictions have implemented a wide range of policy instruments across sectors to reduce emissions (Nascimento et al., 2021). These policy mixes can include command-and-control regulations, tradeable performance standards, subsidies, carbon pricing (via carbon tax or cap-and-trade), and research and development funding. While economists often highlight carbon pricing as the most cost-effective policy instrument to reduce emissions, cost is not the sole consideration in the choice among many policies designed to mitigate greenhouse gas emissions. Alternative policies may be preferred by policy makers seeking to achieve multiple government objectives in addition to reducing GHG emissions, such as addressing additional market failures, incentivizing technological change, improving distributional equity, and ensuring political acceptability (Goulder & Parry, 2008).

The combinations of climate policy instruments implemented in practice have typically evolved over many years, often through an *ad-hoc* process of layering new policies onto the existing climate policy mix (Howlett & Rayner, 2013). However overlapping policies can lead to unintended interactions that alter the overall cost and sustainability of mitigation efforts (Bennear & Stavins, 2007; Bouma et al., 2019; Goulder & Stavins, 2011; Jenn et al., 2019). As climate policy mixes continue to expand and diversify, it is important to understand how interactions between policies can create trade-offs between objectives (Axsen et al., 2020).

This study examines the impacts and potential unintended consequences of interactions between two of California's most important climate policies: its cap-and-trade (CAT) program and low carbon fuel standard (LCFS). The LCFS is a tradable performance standard that sets a limit on the aggregate emissions intensity of the road transportation fuel supply. A CAT program sets an overall limit on emissions from regulated sectors, and allows emitters to trade emissions allowances in order to capture lower cost abatement opportunities. Specifically, I show that the use of an LCFS on a subset of emissions regulated by an emissions cap not only *increases* abatement costs but can also lead to an *increase* in total emissions. These undesirable outcomes occur as a result of the two following dynamics.

First, using additional policy to force high-cost abatement from one sector whose emissions are also covered under the emissions cap increases total policy costs without altering total abatement achieved under the cap. Intensity standards, such as the LCFS, have been shown less cost effective at reducing emissions than an explicit carbon price, due to the implicit subsidy on output (Holland et al., 2009). Using such a policy to force high-cost abatement from the transportation sector alters the order of abatement from what would occur under an emissions cap alone, resulting in higher overall policy costs. At the same time, the emissions limit set by the cap remains unchanged, so higher cost abatement is being forced in one sector without necessarily achieving additional abatement (Fankhauser et al., 2010; Rosendahl, 2019; Stavins, 2016). This has been referred to as the *water-bed effect* (Rosendahl, 2019).

Second, the CAT program does not account for all lifecycle emissions associated with the production of some alternative transportation fuels. Therefore, when the LCFS alters the portfolio of fuels consumed it alters the emissions in sectors not covered by the emissions cap. The direction and magnitude of this effect depends on which fuel substitutions occur and the quantity of emissions outside of the cap associated with those fuels. For example, if the LCFS drives a shift from gasoline which is largely covered under the cap to corn ethanol which has significant emissions that do not face an obligation under the cap (i.e. from agriculture), net emissions will increase. In contrast, if the LCFS drives substitution from corn ethanol to cellulosic ethanol which has lower emissions outside the cap, net emissions will decrease.

On the other hand, the goal of the LCFS may not only be to reduce the emissions intensity of fuels, but to incentivize innovation in the transportation fuels sector (Lade & Lin Lawell, 2015). This study incorporates endogenous technological change using experience curves to evaluate the extent to which induced innovation may offset the costs from this policy overlap.

To evaluate these interaction effects, I develop a dynamic computable general equilibrium model to simulate how consumers and producers respond to policy-induced changes in relative prices under alternative policy combinations. The central simulation results suggest that using an LCFS in combination with a CAT program leads to a 2.4% increase in total cumulative emissions (more than 125 million metric tons, MMT) between 2015-2030, relative to a scenario using an emissions cap alone. The average abatement cost is found to be more than 9% higher under the both policy scenario than with an emissions cap alone.

Induced technological change helps lower policy costs significantly from a no-innovation counterfactual. For example, under the both policy scenario total policy costs are more than three times higher with no induced innovation compared to when innovation is incorporated into the model. The LCFS does drive greater innovation in alternative transportation fuels relative to scenarios without the policy. However, across the range of innovation rates tested the LCFS is unlikely to offset the increased policy costs from overlapping regulation.

In practice, California's cap-and-trade program functions as a hybrid policy instrument incorporating elements of both a fixed-price instrument (i.e. a carbon tax) and fixed-quantity instrument (i.e. an emissions cap). Price-certainty is introduced in the form of a "price collar", with a floor price below which emissions allowances will not be sold and a ceiling price above which unlimited allowances may be sold (CARB, 2018a). When the price of emissions allowances is constrained by the floor or ceiling, the emissions-reduction impact of the LCFS becomes additive because the price incentive remains unchanged in the presence of the LCFS. This results in greater cumulative emissions reductions, although at a higher average abatement cost.

When this price collar is introduced in the model, complementary abatement from the LCFS that occurs when the price collar is binding is found to more than offset inter-industry leakage that occurs when the emissions cap is binding. This results in a net impact of 95 MMT of additional abatement from the LCFS, representing 1.25% of total emissions, however average abatement costs are nearly 15% higher. The price collar also limits overall emissions reductions by allowing unlimited allowances to be purchased at the ceiling price. This results in 26% higher emissions in 2030 relative to a scenario with both policies and an emissions cap without a price collar.

There are additional practical policy alternatives capable of mitigating the potential negative interaction outcomes. In particular, accounting for the lifecycle emissions of alternative transportation fuels under the emissions cap – a feature incorporated in the LCFS – can reduce incentives to substitute from covered to non-covered pollutants and thereby reduce the inter-industry leakage effect. Another option is to replace the emissions cap with a fixed-price instrument such as a carbon tax to improve complementarity with additional policy.

This study contributes to the growing literature on instrument choice and climate policy mixes to demonstrate and quantify the unanticipated interaction effect between two major climate policy instruments. The findings demonstrate the importance of evaluating policy impacts within the context of the broader policy mix. They also have important implications for policy design in other jurisdictions, particularly Washington State and Canada, which are in the process of implementing similar policy combinations.

The remainder of this article is organized as follows: Section 2 provides background on California's policy context and the two policies being examined. Section 3 describes the mechanisms of the policy interaction that contribute to inefficiency, leakage, and induced technological change. Section 4 describes the interaction dynamics using a simple theoretical model. Section 5 describes the simulation model developed to estimate the magnitude of the interaction effects. Section 6 presents the results. Section 7 discusses the policy implications and Section 8 concludes.

2. Policy context

Climate policy choices in California have broad implications for the climate in terms of both emissions reduction as well as policy diffusion. The most populous U.S. state, California as a sovereign country would represent the 5th largest economy and the 18th largest emitter of greenhouse gases in the world, as of 2019 (BEA, 2022; CARB, 2021). California also has a track record as a climate policy leader in the United States. The state pioneered the zero-emission vehicle sales mandate, implemented the country's first broad-based carbon price, and pre-empted federal vehicle emissions standards (Mazmanian et al., 2020). This policy leadership has helped foster the diffusion of similar policies to other jurisdictions. For example, 17 states have adopted or plan to adopt California's more stringent vehicle emissions standards and 15 other states have followed its zero-emission vehicle (ZEV) sales mandate (CARB, 2019b). The state's low-carbon fuel standard was implemented in 2011 and similar policies have since been adopted in Oregon, British Columbia, and recently Washington State (Rhodes et al., 2021). The cap-and-trade program began in 2013 and was expanded to cover transportation emissions in 2015. The program has also been linked with a similar policy adopted by the Canadian province of Quebec (and briefly, Ontario). Therefore, understanding the impacts of California climate policy has important implications not only for achieving emissions reductions, but also for climate policy design and implementation in other jurisdictions.

2.1. California's Cap and Trade Program

California's cap-and-trade program sets a declining limit on the total emissions from regulated sectors and allows firms to trade emissions allowances in order for the lowest cost emissions to be abated first. Permits are distributed through a combination of direct allocation to certain emitters and auctioned to all regulated entities. The program began in 2013 covering electricity generators and high-emitting facilities and was extended in 2015 to include natural gas and transportation fuels, representing approximately 85% of the state's emissions.

The policy acts as a constraint on the total emissions from regulated sectors, taking the form:

$$\sum_i \hat{\beta}_i q_i \leq Z^* \tag{1}$$

Where Z^* represents the total emissions cap. As will be discussed further in section 3.2, the CAT program uses a different emissions intensity value $\hat{\beta}_i$ than the LCFS. This constraint, when applied to a producer takes the form:

$$\hat{p}_i = c_i(.) + \lambda_{CAT}\hat{\beta}_i \tag{2}$$

The cap declines by about 3% each year from 2015-2030 and includes cost containment mechanisms including banking of allowances, a strategic allowance reserve, and the use of offsets. The policy also includes a price floor, called the Auction Price Reserve, below which CARB will not sell additional allowances. The price floor was set at $10.50 in 2013 and rises 5% per year plus inflation. In the extension of the program to 2030 through Assembly Bill 398 (2017), a hard price ceiling was introduced, the point at which regulated firms can buy an unlimited supply of allowances, starting at $65 per ton in 2021. Current design of the cap-and-trade market has resulted in a structural over supply of compliance permits in the market (Cullenward & Coghlan, 2016). This has led to allowance prices remaining near the price floor during the early years of the program (CARB, 2022d).

2.2. California's Low Carbon Fuel Standard

In addition to regulation by the CAT program, the transportation sector is covered by another overlapping but unlinked credit market: the low carbon fuel standard. California's low carbon fuel standard sets a decreasing volume-weighted average emissions intensity standard for transportation fuels, and allows fuel suppliers the flexibility to trade compliance credits in order to meet the standard (CARB, 2022a). For example, a supplier of gasoline whose emissions intensity exceeds the standard can achieve compliance with the regulation by: (i) blending with lower carbon intensity biofuels to reduce their overall carbon intensity, (ii) purchasing compliance credits from another fuel supplier whose emissions intensity falls below the standard, or (iii) generating LCFS credits by making specified infrastructure investments.

The standard implicitly taxes fuels based on the degree to which their emissions intensity exceeds the standard and proportionally subsidizes fuels by how much their emissions intensity falls below the standard (see equation 4). This design incentivizes emissions intensity reductions from the fuel supply while remaining technology-neutral, encouraging innovation and competition between biofuels, electric vehicle charging, hydrogen, and propane with gasoline and diesel fuels. The LCFS contributes to reducing emissions through substitution effects but creates incentives for output by subsidizing energy use and has been shown to be less cost-effective than an explicit carbon price at reducing emissions (Holland et al., 2009).

The policy was implemented in 2011 with the goal of achieving a 10% reduction in the emissions intensity of fuels by 2020 and was later extended to reach a 20% reduction in emissions intensity by 2030 (CARB, 2018b). The policy acts as a constraint on the emissions intensity of the total supply of fuel, which takes the form:

$$\frac{\sum_i \beta_i q_i}{\sum_i q_i} \leq \beta^* \tag{3}$$

Where β_i represents the carbon intensity of fuel i (in metric tons of carbon dioxide equivalent per gallon of gasoline equivalent, t CO2-eq/gge), q_i represents the quantity of fuel i in gallons of gasoline equivalent, and β^* represents the fuel intensity standard set by the policy (t CO2-eq/gge). Assuming full cost pass through, the price paid by a purchaser of each fuel then becomes:

$$\hat{p}_i = c_i(.) + \lambda_{LCFS}(\beta_i - \beta^*) \tag{4}$$

Where the shadow price of the constraint (λ_{LCFS}) represents the price of LCFS compliance credits and $c_i(.)$ represents the production cost of the fuel. Empirical research supports the assumption that policy costs from fuel standards are largely passed through (Knittel et al., 2017; Lade & Bushnell, 2019).

However, the incentive functions differently for electricity as a transportation fuel, where vehicles most often charge at home and electricity prices are set by regulated utilities. In practice, utilities are mandated to use LCFS credit revenue to "benefit current or future EV customers" and LCFS credits earned by utilities are typically spent in a lump-sum rebate to EV owners (SDG&E) or for post-purchase EV rebates (PG&E and SCE) (Bushnell et al., 2021; CARB, 2022b). Therefore, in the model described in section 5 the LCFS credits earned by the electricity sector are returned to consumers as a lump-sum rebate.

Credits can also be generated from installing infrastructure, for example for hydrogen or public EV fast-charging stations or installing carbon capture and storage on fuel production facilities (CARB, 2022c). However, these infrastructure credits are limited to 2.5% of total credit generation from the previous quarter and are excluded from consideration in this model.

Huseynov and Palma (2018) estimate that the policy is responsible for reducing transportation emissions by approximately 10% between 2011 and 2017 and also improved air quality. However, this may not represent net emissions reductions, if that abatement simply replaced lower-cost abatement that would have otherwise occurred elsewhere under the cap. In 2021 compliance credits for the LCFS traded for an average of $187.50 per ton carbon dioxide equivalent (t CO_2-eq) (CARB, 2022a). This represents expensive marginal abatement compared to the state's CAT program where allowance auction results were in the range of $17-29 per ton CO_2-eq in 2021 (CARB, 2022d).

3. Interaction effects

3.1. Cost-effectiveness

It has been well established that intensity standards are less economically efficient than a carbon price at reducing the same quantity of emissions (Holland et al., 2009). This occurs because the intensity standard functions as the combination of an implicit tax on emissions and an implicit subsidy to output. The result is that emissions abatement results primarily from substitution of low-carbon for higher-carbon inputs rather than through reductions in total fuel demand. However, this implicit output subsidy may also be desirable in terms of maintaining competitiveness of an industry that competes with other unregulated jurisdictions or limiting the price impacts felt by consumers that can create political backlash.

Another way intensity standards tend to be less efficient than a carbon price is through narrower coverage of emissions. Intensity standards are typically placed on a particular sector or energy service (i.e. fuel emissions intensity or electricity generation), in contrast with carbon pricing which can be applied across multiple sectors (Rhodes et al., 2021). By applying to a smaller scope of emissions, the standard limits the total pool of abatement opportunities available at a low cost.

When an intensity standard like the LCFS is combined with a fixed-price instrument, such as a carbon tax, it creates additional emission reductions although generally at a higher cost (Anderson et al., 2016; X. Chen et al., 2014; Christensen & Hobbs, 2016; Huang et al., 2013; Lepitzki & Axsen, 2018; Whistance et al., 2017). This is because the price incentive from the carbon tax is not altered by the presence of the LCFS, so that the incentives of the two policies can have a cumulative effect.

However, when an intensity standard is used for a sector also covered by a quantity-based instrument, such as a CAT system, the interaction occurs differently. This is because the price incentive from the cap depends on the magnitude of abatement required by covered sectors, which may change in the presence of additional policy. Therefore, if a standard forces some of the abatement required from regulated sectors, less abatement is required under the CAT system and the allowance price can be lower. Yet, the total quantity of abatement achieved remains the same.

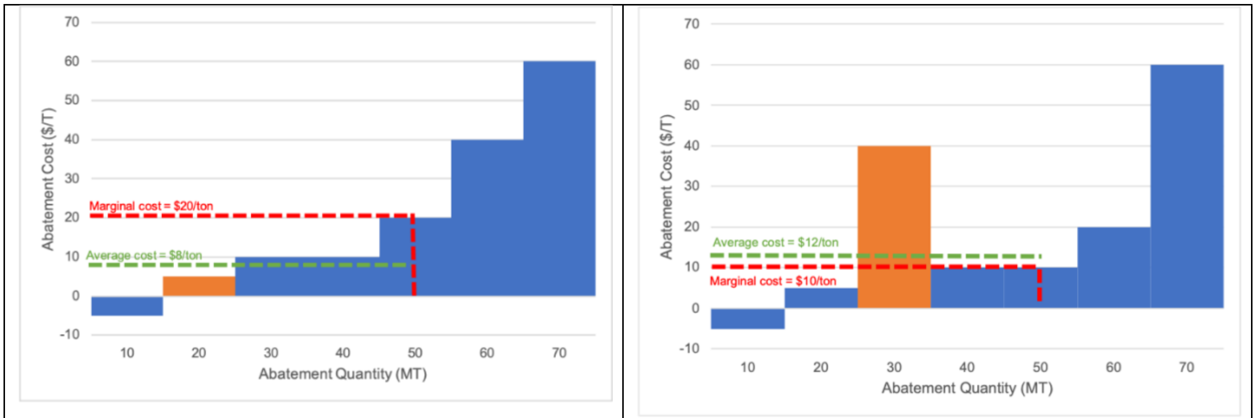
The impact on cost depends on the relative stringency of the two policies. If the cap is sufficiently stringent that compliance with the intensity standard would be achieved even if the standard were not in place, then the intensity standard is redundant and the marginal compliance cost will decrease to zero. This has been referred to as a ‘belt and suspenders’ approach to climate policy, where overlapping policies are redundant but have relatively little impact on costs (Levinson, 2011).

For example, Figure 1a depicts an illustrative marginal abatement cost curve. On its own, the CAT program sets a quantity of abatement to be achieve and allows firms to trade allowances to realize the lowest cost abatement options first. The allowance price is equal to the marginal cost of abatement and the average abatement cost is lower. If a standard forcing a certain abatement activity (marked in orange) to occur is sufficiently low-cost, then it would have occurred under the cap regardless of whether or not the complementary policy existed. Therefore, the complementary policy only adds the cost of policy administration, without changing either total or marginal abatement costs.

Alternatively, when the intensity standard is more stringent than the cap, it forces higher-cost abatement than would occur under the cap alone but does not change total emissions reductions. This is the expected interaction in this study, where LCFS credits traded for more than $180/ton while CAT allowance prices were less than $30/ton in 2021. For example, in Figure 1b, suppose an intensity standard forced a higher cost abatement activity (in orange) to occur. This does not change the total abatement required under the cap, but simply reorders the abatement to be achieved under the cap. As a result of forcing this high-cost abatement, the marginal abatement cost of the CAT program reflected by the allowance price falls, giving the appearance of a less expensive policy. However, the average abatement cost has increased, resulting in higher total compliance costs for the same quantity of emissions reductions. This has been referred to as the 'waterbed effect', where the addition of a policy creates no additional abatement, but simply shifts emissions from one sector to another (Fischer & Preonas, 2010; Fowlie, 2010; Goulder, 2013; Rosendahl, 2019).

Previous literature has examined the waterbed effect, with a particular focus on the interaction between an emissions cap and a renewable portfolio standard (RPS) which mandates a certain proportion of electricity to be generated from renewable resources (Bird et al., 2011; de Jonghe et al., 2009; Dimanchev & Knittel, 2020). Dimanchev & Knittel (2020) show that the magnitude of the efficiency loss depends on the relative stringencies of these instruments and find that increasing the proportion of abatement from a CAT relative to an intensity standard is welfare improving with diminishing returns. However, previous research considers both policies applied to the same total pool of emissions (i.e. a CAT and RPS on electricity), whereas in practice the LCFS applies to a sub-set of emissions covered by the cap that represent relatively high-cost abatement.

Finally, it is also possible that California's CAT program performs more as a fixed-price instrument due to its hybrid design which allows for complementary policies on covered sectors to contribute additional abatement (Perino et al., 2020). The CAT program includes a price floor, below which additional permits will not be sold. It also includes a price ceiling, at which an unlimited supply of credits may be sold. These features incorporate price certainty elements of a carbon tax to the traditional CAT design. Since CAT permit prices cannot fall below the price floor, when complementary policies provide sufficient abatement to make the cap a non-binding constraint, rather than the allowance price falling to zero, the floor price acts as a carbon tax providing further emissions reductions. This could be the case in California where CAT permit prices were at or near the price floor during the early years of the program and is further examined in Section 6.2.



A. *As an illustrative example, consider this marginal abatement cost curve where 50 MT of abatement are required under an emissions cap. CAT incentivizes lowest cost abatement first.*

B. *This chart shows the same MACC, when higher cost emissions abatement is being forced through a complementary policy.*

Figure 1. Illustrative example of forcing high-cost emissions abatement

The idea that such an overlapping approach to climate policy will result in higher cost abatement is fairly well established, but has not yet been evaluated in this context where the complementary policy applies to a sub-set of emissions. There are two additional dynamics that make this policy interaction particularly important to evaluate. First, when the CAT program does not cover all emissions, complementary policies may incentivize emissions that occur outside of the cap, leading to an increase in net emissions. Second, by increasing the market penetration of alternative transportation fuels now, the LCFS may incentivize learning that helps drive down the costs of production and correspondingly the future cost of abatement, potentially offsetting the additional policy cost. These two dynamics are described in the following sections.

3.2. Inter-industry emissions leakage

As indicated above, the use of a cap-and-trade program in combination with a low-carbon fuel standard may not only increase policy costs, but could also increase net emissions. This occurs because the CAT program does not cover all emissions associated with the production of low-carbon fuels. Therefore, if the LCFS incentivizes substitution from fuels whose lifecycle emissions are entirely covered under the cap to fuels that generate emissions not covered by the cap, net emissions will increase. This dynamic has been referred to as “inter-industry leakage” in the context of the European emissions trading system interacting with feed-in tariffs (Jarke & Perino, 2017). Conversely, if the LCFS leads to substitution from fuels with significant emissions outside of the cap to an alternative fuel with fewer uncovered emissions, net emissions may decrease. The potential for this interaction to drive emissions leakage was first identified by Schatzki & Stavins (2012) and has not yet been rigorously analyzed.

Table 1. Emissions intensity accounting by each policy			
Fuel	**Average[1] Carbon Intensity** (g CO_2eq/MJ)		**Difference**
	CAT	**LCFS**	
Gasoline	98.9	98.9	--
Corn Ethanol	9.32	73.5	+ 64.18
Cellulosic Ethanol	9.32	25[2]	+ 15.68
Diesel	98.8	98.8	--
Biodiesel (cooking oil)	7	27	+ 20
Renewable Diesel	7	34	+ 27
Electricity	24	24	--

The LCFS and CAT account for the emissions intensity of fuels differently. The LCFS accounts for the life-cycle emissions of transportation fuels, including upstream production emissions (i.e. agriculture emissions for the production of biofuels) and from in-direct land-use change, one of the first major climate policies to do so (Sperling & Yeh, 2010; Yeh et al., 2016).[3] In contrast, the CAT program only considers the end-use combustion emissions of these fuels. This means that upstream emissions from agriculture, land-use change, and biorefineries (that are outside of California or that emit less than 25,000 MT CO_2 per year) are not counted under the cap (see Table 1 for the average carbon intensity values of each policy).

Both policies consider the combustion of biofuels to be carbon-neutral, meaning that any CO_2 emitted from combustion was equivalently sequestered during the growing process and only account for methane and nitrous oxide emissions from combustion. This corresponds to emissions accounting practices established by the United Nations Framework Convention on Climate Change (Pulles et al., 2022).

As detailed in Section 2.2, the LCFS subsidizes any fuel whose life-cycle emissions intensity falls below the standard. Therefore, to the extent that the LCFS leads to an increase in the production of fuels with emissions not covered by the CAT program, it will increase emissions outside of the cap. The water-bed effect described in the previous section suggests that any emission reductions achieved under the cap by the LCFS will be offset by depressing the CAT allowance price which allows for an increase in emissions from other sectors under the cap. On the other hand, if the LCFS incentivizes substitution from fuels with significant emissions outside of the cap to fuels with fewer emissions outside of the cap, net emissions will decrease. This could be the case if the LCFS incentivizes a shift from corn ethanol to cellulosic ethanol or electricity for transportation.

Consequently, the emissions impact of using an LCFS in combination with a CAT program is ambiguous. The direction and magnitude of this leakage effect will depend on the quantity and type of fuels used to comply with the standard, as well as the fuels they displace.

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[1] The emissions intensity values displayed here and applied in the baseline model scenarios represent volume-weighted average carbon intensity values reported by the California Air Resources Board.

[2] Accounting by CARB for the LCFS distinguishes specific carbon intensity rates by fuel and production process but does not publish average emissions for residue-based vs. crop-based ethanol. The value used here is from Murphy & Kendall (2015).

[3] Accurately accounting for indirect land-use change is a longstanding and actively debated area of research (Breetz, 2017; Broch et al., 2013; Khanna et al., 2017; Lark et al., 2022; Searchinger et al., 2008). This study does not seek to evaluate the merits of that literature, but takes estimates of the lifecycle emissions of alternative transportation fuels used by the California Air Resources Board for policymaking as given.

3.3. Innovation

The main objective of such an overlapping policy approach may not be to reduce emissions at lowest cost, but rather to encourage innovation in the low-carbon fuels sector to reduce future costs. Accounting for the dynamic costs of a climate policy over time may differ substantially from static costs when considering the role of induced innovation (Gillingham & Stock, 2018). By incentivizing the increased market penetration of low emission fuels, the LCFS can help decrease the production costs of those fuels over time, lowering future abatement costs, which may offset additional policy costs.

The impacts of innovation under a low carbon fuel standard have received relatively little attention in the literature, with much of the focus on *ex ante* evaluation of the potential of cellulosic ethanol production (X. Chen et al., 2012; Y. Chen et al., 2017; Holland et al., 2015). In practice, cellulosic ethanol has failed to realize sufficient cost declines in order to gain market share and electric vehicle charging and renewable diesel have emerged as the marginal compliance fuels (Bushnell et al., 2019).

To evaluate the potential benefits from induced innovation under alternative policy scenarios, induced technological change is incorporated endogenously in the model using experience curves (described in Section 5.4) An experience curve represents the change in cost associated with a given change in cumulative production (Arrow, 1962). The experience curve approach has been widely used to model innovation in energy technologies (McDonald & Schrattenholzer, 2001; Rubin et al., 2015). It has also been demonstrated empirically in the case of biofuels for US corn ethanol production and Brazilian sugarcane ethanol (see Table 2) (van den Wall Bake et al., 2009).

4. Theoretical Model

Proposition: *Applying an LCFS in the presence of an emissions cap that undervalues the emissions intensity of alternative transportation fuels can lead to a net increase in emissions as well as an increase in policy cost.*

Proof.

To illustrate these dynamics, consider a simple theoretical model developed by Holland et al. (2009) and extended by Dimanchev & Knittel (2020) depicted in Figure 2a. In this model there are two fuels, a high-carbon fuel and low-carbon fuel such that:

$$\beta_L < \beta^* < \beta_H \tag{5}$$

Where β^* represents the intensity standard and β_L and β_H represent the emissions intensities of the low- and high-carbon fuel, respectively. Figure 2 represents the quantities of a high carbon (q_H) fuel and low carbon fuel (q_L). Point X represents an unconstrained equilibrium in the production and consumption of both fuels.

The LCFS constraint can be represented as a minimum proportion of low carbon fuel per unit of high carbon fuel to comply with the maximum emissions intensity of the total fuel supply, depicted as the upward sloping line. This constraint shifts the equilibrium to point L, which imposes some welfare cost represented by the distance from the unconstrained equilibrium point.

The CAT constraint can be represented as the maximum sum of quantities of the two fuels, accounting for their relative emissions intensities (as in equation 1). Therefore, any equilibrium with the CAT must fall below this downward sloping line. With only a CAT in place, the new equilibrium is reached at point C, which also induces some welfare cost represented by the distance from X.

When both policy constraints are in place, equilibrium is attained at point B. Since the CAT constraint theoretically represents the total quantity of emissions, it could be considered an emissions isoquant, where any point along the constraint represents the same quantity of total emissions. As can be seen, the new equilibrium under both policy constraints represents the same total quantity of emissions as the CAT alone, but at a higher welfare cost. It represents a slight decrease in emissions from the LCFS-only equilibrium with a slightly larger policy cost. This illustrates how using an LCFS in combination with a CAT program can increase welfare costs without decreasing total emissions.

However, suppose that a CAT policy does not account for upstream emissions from a low-carbon fuel (β_L^{CAP}), as described in Section 3.2. Therefore, the emissions intensity parameter used is lower than the total life-cycle emissions intensity of the low carbon fuel (β_L^{LC}):

$$\beta_L^{CAP} < \beta_L^{LC} \tag{6}$$

Figure 2b adapts Figure 2a to represent the undercounting of emissions under the CAT policy. The CAT constraint line is defined as:

$$\beta_L^{CAP} \mathrm{q_L} + \beta_\mathrm{H} \mathrm{q}_H \leq Cap \tag{7}$$

However, this cap constraint undercounts emissions from low carbon fuels and does not represent a true emissions isoquant. When accounting for the total emissions of the low carbon fuel, an emissions isoquant, denoted by the dotted lines, is calculated as:

$$\beta_L^{LC} \mathrm{q_L} + \beta_\mathrm{H} \mathrm{q}_H = Total\ emissions \tag{8}$$

Under the CAT-only constraint the equilibrium is again reached at point C. When the LCFS constraint is also in place, the equilibrium again shifts to point B. However, this new equilibrium is on a higher emissions isoquant (dotted line) – representing an increase in total emissions. Therefore, by shifting the equilibrium upward along the CAT constraint, the use of the LCFS in combination with the CAT results in an increase in total emissions.

In contrast, if a CAT constraint were imposed alone using the life-cycle emissions intensity of the low-carbon fuel, it is possible to achieve a new equilibrium at point LC-C which achieves greater emissions reductions at a lower cost than the both-policy equilibrium.

However, as previously mentioned, the magnitude and direction of this effect are ambiguous. If the high-carbon fuel has greater emissions outside of the cap than the low-carbon fuel being incentivized by the LCFS, net emissions can decrease (i.e. if cellulosic ethanol is incentivized to replace corn ethanol). Therefore, to estimate this effect numerical simulation modelling with alternative fuels is required.

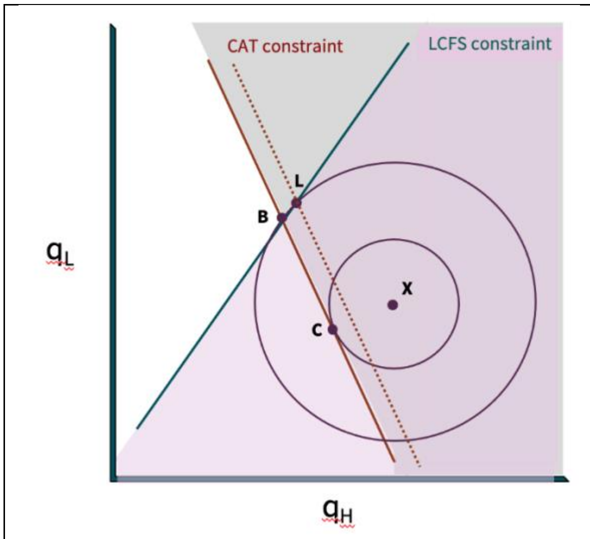


Figure 2a. Inefficiency of an intensity standard relative to a cap-and-trade. *From Dimanchev and Knittel (2019-WP) based on Holland et al. (2009)*

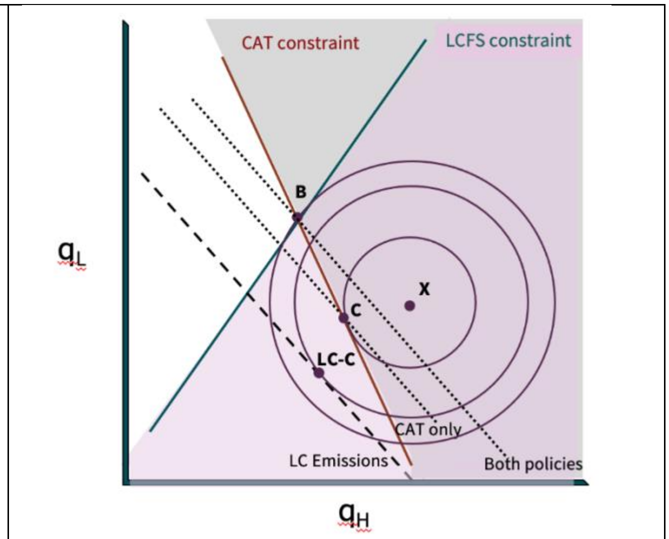


Figure 2b. Applying an intensity standard with a CAT that undercounts emissions.

5. Numerical simulation model

To numerically evaluate the magnitude of these effects, I have developed a dynamic computable general equilibrium (CGE) model to simulate how consumers and producers respond to policy-induced changes in relative prices under alternative policy scenarios. CGE models have been widely used to evaluate climate policy alternatives (Bergman, 2005; Böhringer, 1998; Goulder et al., 2016; Wing, 2011). This type of model is particularly well suited to evaluate the policy interaction of interest because a broad-based carbon price, such as California's CAT program, alters the relative prices of goods and factors across multiple markets and the price imposed by the CAT is endogenous to how multiple sectors respond to the imposed policies. This section provides a summary of the model structure and policy scenarios. A more detailed description of the model can be found in Appendix A.

The model includes representation of five production sectors: transportation, electricity, agriculture, heavy industry, and "other commercial goods," as well as a representative consumer. This sectoral division was chosen based on the major emitting sectors in California, which together represent more than 90% of state emissions (CARB, 2021). Greater technological detail is included in the production structure of the transportation fuel and electricity sectors to reflect the sources of emissions and abatement potential of particular interest to this study. Preferences and production inputs are represented through nested constant elasticity of substitution (CES) functions.

The model runs in annual time-steps from 2010 to 2030, with producers and consumers myopically maximizing profits and utility, respectively. In each period, the model solves for a vector of prices that equate supply and demand for all goods and factors while also achieving compliance with the active policy constraints.

5.1. Production

Primary factors of production include labor and capital, as well as the natural resource inputs land, oil, and natural gas resources. These factors, combined with their intermediate products, are used to produce five consumption goods.

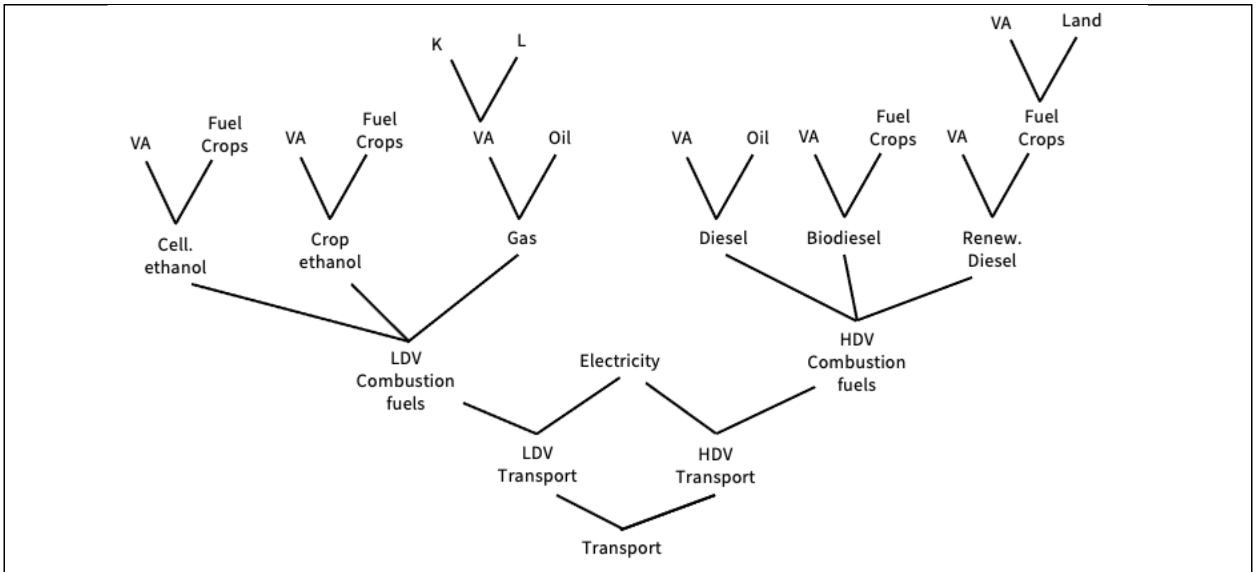
Factors
A labor endowment is defined exogenously for the baseline period (2010) and growth in labor endowment is expressed as effective labor units to reflect both population growth and increases in productivity in subsequent periods. To do this, the growth in labor endowment is indexed to observed gross state product (GSP) growth in California from 2010-2021 using data from the Bureau of Economic Analysis and to GSP forecasts for 2022-2030 from California's Department of Finance (BEA, 2022; California, 2022). Labor is modelled as perfectly mobile across sectors. The supply and transformation of capital follows a putty-clay structure, detailed in Section 5.3.

There are three natural resources represented in the model: oil, natural gas, and land. The supply of oil and natural gas are assumed to be perfectly elastic at an exogenously defined price. This is to reflect the way that commodity prices are set in global markets and demand changes in California from the policies under consideration are unlikely to significantly alter global prices. Alternative high, medium, and low-price trajectories over the model period are examined in the sensitivity analysis, with the medium price path representing the base case. Land is represented as an exogenously defined endowment of a fixed quantity, with agricultural productivity improving exogenously over time. The price of land is solved endogenously in the model based on the relative demand for the goods it is used to produce.

The model includes six main goods demanded by consumers: transportation, electricity, agriculture, home energy, industrial goods, and other commercial goods.

Transportation
Consumers demand transportation as a service which includes both light-duty and heavy-duty transportation. There are seven fuels used to produce transportation: gasoline, crop ethanol, cellulosic ethanol, and electricity for light-duty transportation and diesel, biodiesel, renewable diesel, and electricity for heavy-duty transportation. The two main petroleum fuels (gasoline and diesel) are produced using the value-added composite (VA - which is a composite of capital and labor) and oil. The biofuels are produced using a VA composite and fuel crops which are produced from VA and land by the agriculture sector. Biodiesel and renewable diesel also use alternative feedstocks such as tallow and agricultural residues which are reflected in lower inputs of fuel crops and higher elasticities of substitution between production inputs.

Alternative transportation fuels are not perfect substitutes. Technological constraints limit the extent to which certain alternative fuels can be incorporated into a fuel blend and still be used in traditional combustion engine vehicles which dominate the current fleet. Specifically, the technological blend limit for ethanol with gasoline is approximately 15% and the for biodiesel with petroleum diesel is 20% (National Research Council, 2011). To reflect these technological limits on fuel blending, blend constraints are exogenously imposed in the model.



[redacted]

Electricity

The electricity sector includes five generation sources: solar, wind, nuclear, natural gas, and hydropower which together represent approximately 90% of electricity generation in California over the period 2010-2020 (CEC, 2022). These are grouped into two types of electricity generation, variable and firm, which are intended to incorporate the imperfect substitutability of variable sources for baseload generation (Hirth et al., 2016). Nuclear electricity is defined at a fixed quantity and price to reflect the near constant level of annual generation from California's remaining nuclear facility at Diablo Canyon.[4] Supply of hydroelectricity is modelled to vary stochastically over the range of observed generation capacity to reflect the way annual generation depends on variable water availability.

[4] Based on the current policy uncertainty with regard to the prospective closure of Diablo Canyon in 2024, nuclear generation is assumed constant over the model period in the baseline case.

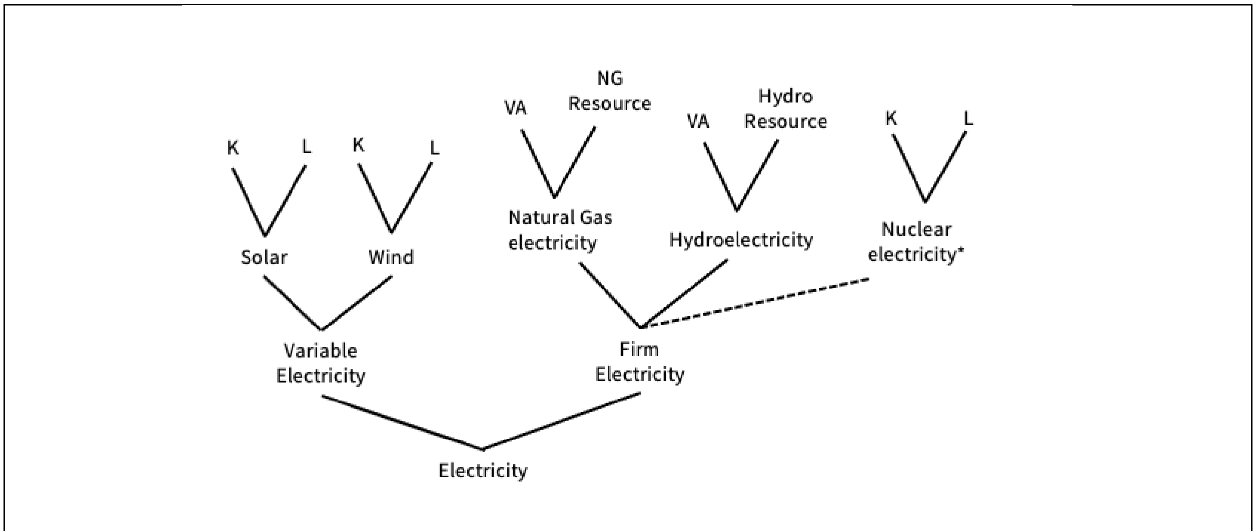


Figure 4. Electricity sector production structure

The agriculture sector produces food and fuel crops. Land is used with capital and labor to produce either food or fuel crops. Food is consumed directly by consumers and fuel crops are demanded by the transportation sector to produce biofuels.

The industrial sector is intended to represent heavy-emitting industries such as steel and cement. The production structure follows a common functional form where capital and labor are combined with an energy composite which includes three major energy sources. In this case the available energy sources include oil, natural gas, and electricity.

Home energy is supplied using electricity and natural gas to represent the main energy sources used for cooking and heating in buildings. The commercial sector is intended to represent other commercial goods produced for consumption. These goods are produced using the CES inputs of labor and capital with an energy composite that includes electricity and natural gas.

5.2. Consumption

A representative consumer seeks to maximize utility through the consumption of five goods (food, transportation, home energy, industrial goods, and other commercial goods) as well as leisure. The consumer receives income from wage labor and rents from capital and natural resource use.

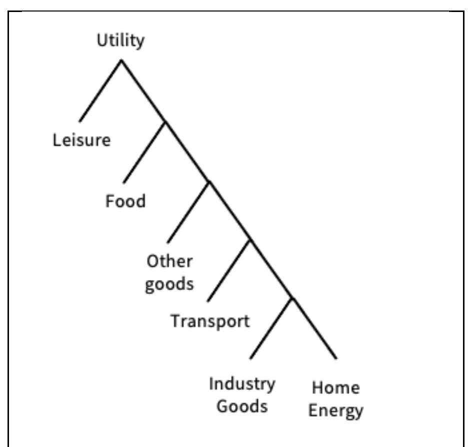


Figure 5. Consumption structure

Consumption follows a nested constant elasticity of substitution structure, as in Figure 5. Each of the sub-utility functions that contribute to overall utility are CES in form. The CES structure allows for the representation of preferences to demonstrate important attributes, namely by being continuous, differentiable, monotonic, and strictly quasi-concave. For example, at the highest-level nest consumers gain utility (U) from the consumption of leisure (L_H) and goods (G), based on the function:

$$U(L_H,\ G) = \left(\alpha_U L_H^{\rho_U} + (1 - \alpha_U) G^{\rho_U}\right)^{\frac{1}{\rho_U}} \tag{9}$$

Each CES function is parameterized with a distribution parameter (α) and substitution parameter (ρ). The distribution parameter is calibrated according to the baseline data (see calibration in Section 5.5). The substitution parameter ρ is calculated from the exogenously defined elasticity of substitution (σ) between leisure and goods, such that $\sigma \equiv 1/(1-\rho)$.

At each nest, consumers seek to maximize their utility, subject to their budget constraint. Taking first order conditions and some manipulation yields conditional demand equations for each good, where demand is a function of the budget constraint and the price of each good in the nest. For example, the conditional demand equation for leisure derived from equation 9 takes the form:

$$L_H^D = \frac{Budget}{\mathrm{w} + p_G \left[\frac{p_G(\alpha_U)}{\mathrm{w}(1 - \alpha_U)}\right]^{\frac{1}{\rho_U - 1}}} \tag{10}$$

Where demand for leisure is a function of the available budget, the opportunity cost of leisure (w, the wage rate) and the price of the goods composite (p_G).

5.3. Savings and capital investment

Savings in each period is assumed to be a fixed proportion of total income, based on the estimated average income-savings ratio for the U.S. from 2010-2020 (BEA, 2022). Savings are used as investment to produce new capital available in the subsequent period at the prevailing rental rate of capital.

The model takes a putty-clay approach to modelling capital to reflect imperfect mobility of capital between sectors (Atkeson & Kehoe, 1999). At the end of each period, existing capital in given sector becomes fixed in that sector. In the subsequent period the existing stock of rigid capital depreciates at the defined rate for sector j (δ_j). Capital demand for each sector is met with the existing stock of capital ($K_{j,t}^R$) and demand for new malleable capital from investment (K_t^M). Demand for new capital in a sector is a function of the marginal product of capital for that sector, such that a prevailing rental rate for new capital is computed endogenously in the model to equate demand for new capital with the supply from investment. Therefore, the capital stock in a given sector (j) in a period (t+1) can be computed as:

$$K_{j,t+1} = K_{j,t+1}^M - (1 - \delta_j) K_{j,t}^R \tag{11}$$

The putty-clay capital approach reflects the way policies can induce investment in production capital, such as fuel refineries. This capital is often long-lived and can help to “lock-in” certain technologies and production pathways even if rates of return to capital change which have implications for future emissions trajectories (Acemoglu & Aghion, 2019).

5.4. Technological change

The rate of innovation is endogenous in the model and derives from an experience curve approach for seven low-carbon technologies (see Table 2). The typical formulation of an experience curve is that the unit cost, c(Q), is a decreasing function of cumulative production (Q), initial cost (α), and the innovation parameter (ψ). This is often given the functional form of the power rule, where each doubling in cumulative production results in a proportional decrease in price (Thompson, 2012).

$$c(Q) = \alpha Q^{-\psi} \tag{12}$$

The innovation parameter ψ can be used to define the progress ratio (LR) which represents the per cent cost decline for each doubling in cumulative production, expressed as:

$$LR = 1 - 2^{-\psi} \tag{13}$$

Table 2. Learning rates for low carbon technologies		
Technology	**Learning Rate** *(range)*	**Comments**
Corn ethanol	18% *(13-32)*	Literature estimates of learning rates for corn ethanol range from 13-32%, the current value used is from Hettinga et al. (2009)
Cellulosic ethanol	4% (2-20)	Limited empirical evidence suggests learning rates have been low.
Biodiesel	5% (2-25)	Learning rates used in previous research range from 5% to 19.6% (Berghout, 2008; Chen et al. 2012)
Renewable diesel	10% (2-25)	Empirical evidence is not available, so a range of values based on empirical estimates for biodiesel are tested with the value that most closely replicates observed consumption quantities chosen as the preferred value.
Electric transport	9% *(6-23)*	Wide range of learning rate estimates for electric vehicles and battery technology (Nykvist & Nilsson, 2015; Safari, 2018; Weiss et al., 2019)
Solar electricity	23% *(10-47)*	Estimates from review by Rubin et al. (2015) range from 10-47% with a mean of 23%.
Wind electricity	12% *(5-32)*	Estimates from review by Rubin et al. (2015) range from 5-32% with a mean of 12%.

In the model, technological change involves both endogenous and exogenous elements. Endogenous innovation is incorporated into the model by increasing the production efficiency for each technology in a given period based on the cumulative production from prior periods (as well as prior to the model start year in 2010) following the experience curve approach.

Exogenous innovation is also incorporated to represent efficiency improvements in transportation (fuel efficiency), electricity (energy efficiency), and both food and fuel crop productivity. Exogenous fuel efficiency improvement is set to improve 2.3% annually to reflect the fleet-wide average improvement from California's fuel economy regulations. Energy efficiency is set to improve 1.3% annually based on data reported by the International Energy Agency (IEA, 2021). Agricultural land productivity in is set to increase by 1.2% annually based on statistics from the California Department of Food and Agriculture (CDFA, 2010).

5.5. Data and parameterization

The model is calibrated to a constructed baseline social accounting matrix (SAM) for the California economy in 2010, before either policy was in place. A social accounting matrix tracks input and output transactions between all agents in the model. Industry input and output flows were obtained from the US Department of Commerce Bureau of Economic Analysis and downscaled according to California's share of GDP in 2010 (12.6%). Data on electricity, fossil fuels, and transportation fuels are from the EIA and Alternative Fuels Data Center and data on food and crop production are from the US Department of Agriculture. Data on emissions are from California Air Resources Board. In many cases the "bottom-up" energy consumption data do not perfectly match downscaled "top-down" data from the BEA and a number of assumptions are required to generate a consistent baseline SAM (Böhringer, 1998; Sue Wing, 2008). Additional detail on the data sources and the calibration protocol can be found in Appendix A.

To the extent possible, empirically estimated parameters for elasticities of substitution are drawn from the literature. However, parameters remain uncertain and a number of elasticities have not been empirically estimated. In order to evaluate the robustness of the simulation results, alternative high and low values for elasticities are tested in the sensitivity analysis (see Appendix C). Additionally, by simulating the policy period beginning in 2010 I am able to calibrate uncertain parameter values in order to replicate observed prices and quantities between 2010 and 2022 (see Figure 6). For households, the elasticity of substitution between goods and leisure is calibrated to reflect a compensated elasticity of labor supply equal to 0.4. Elasticities of substitution between production inputs are defined exogenously and detailed in Appendix A.

Emissions intensity parameters used under the LCFS are based on the volume-weighted average life-cycle values approved by CARB. Although there has been extensive literature debating the modelling and assumptions used to generate emissions intensity estimates, particularly for the life-cycle emissions intensity of fuels including indirect land-use change, for this study the values used by CARB for the purposes of policymaking are considered accurate.

Emissions intensity values under the CAT program reflect methane and nitrous oxide emissions from end-use combustion and the portion of production emissions that come from California facilities which hold a compliance obligation under the cap. Production emissions from bio-based fuels only incur a compliance obligation if they are produced in California at a facility with annual non-biogenic emissions greater than 25,000 MT CO_2eq. The vast majority of biofuels consumed in California are produced outside of the state (88%) (CARB, 2022a). Facility level data on biofuel production capacity from the EIA and emissions data from the California Air Resources Board suggests that no producers of renewable diesel and biodiesel have faced a CAT compliance obligation to date. However, there are several ethanol production facilities which do incur an obligation under the CAT program, the C.I. level for ethanol under the CAT program is adjusted based on the production-weighted average emissions covered. Additional detail on these calculations can be found in Appendix B.

Validation

To validate the parameterization of the model, I use hindcasting to compare simulation results from the both policy scenario with observed values between 2010-2022 (see Figure 6). Simulated LCFS permit prices track observed prices reasonably well with a slight overestimation. The higher prices in the model may reflect the absence of banking credits across time, alternative credit generation mechanisms such as infrastructure investment, or additional compliance pathways such as biogas. In the early years of the CAT program, simulated allowance prices are near zero whereas observed values are constrained at the low end by the auction price floor. This may contribute to overestimating CAT allowance prices after year 2018, since the model does not account for allowance banking behavior or the existing stock of emissions allowances prior to incorporating transportation in 2015. Reasonably approximating the observed allowance prices is an important attribute to reflect the relative stringency and economic incentives imposed by the policies. Simulated consumption levels of alternative fuels and electricity generation by alternative sources also reasonably match observed values.

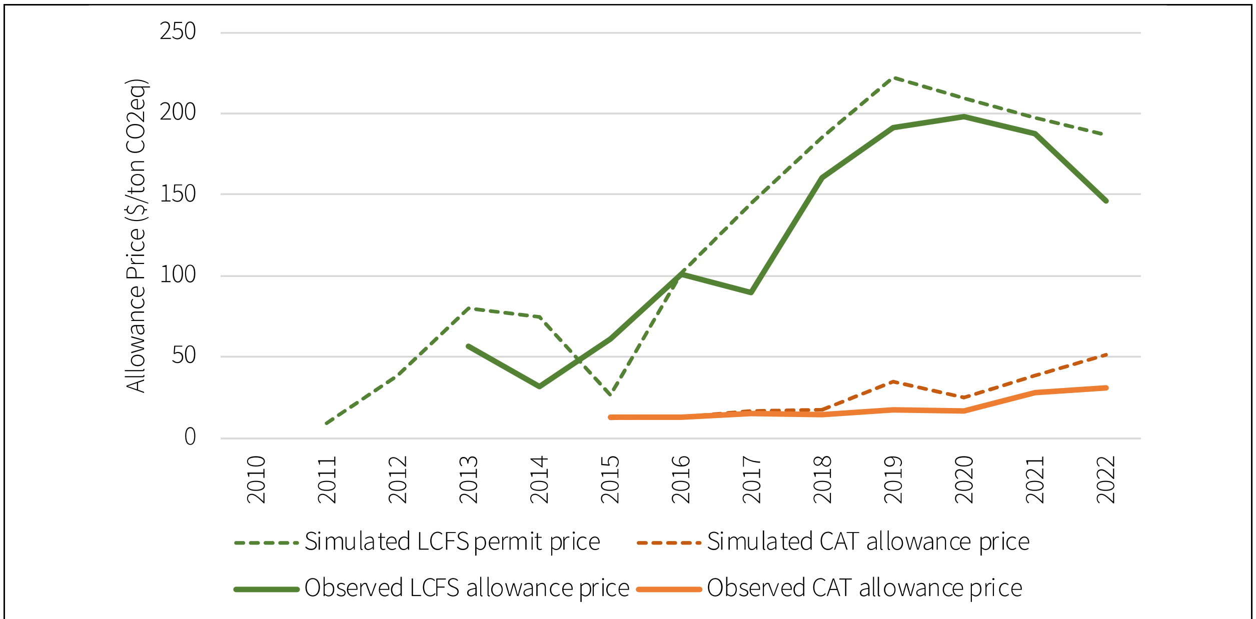


Figure 6. Observed and simulated policy allowance prices under the “both policy” scenario.

5.6. Policy scenarios

The simulations include six main policy scenarios:

1. **No policy** – No policy constraints are applied.
2. **LCFS-only -** The LCFS is implemented starting in 2011 with stringency calibrated to the specified percent reduction in emissions intensity of the fuel supply by the California Air Resources Board (CARB, 2018b). This stringency trajectory is designed to achieve a 10% reduction in emissions intensity of the fuel supply by 2020 and a 20% reduction by 2030.
3. **CAT-only -** The cap-and-trade program is implemented starting in 2015 and calibrated to match the decline rate of the cap.

4. **Both policies** – The LCFS is implemented starting in 2011 and the emissions cap, including transportation, is added starting in 2015.
5. **CAT-only + price collar** – This scenario incorporates the hybrid design of the cap-and-trade program by incorporating an allowance price floor and ceiling.
6. **Both policies + price collar**– This scenario is intended to reflect policy implementation as it has occurred in practice: the LCFS is implemented starting in 2011 and the CAT program including a price floor and ceiling is added starting in 2015.

In particular, the comparison between the both policy scenario and CAT-only scenario, both with and without the price collar, are the main comparisons of interest in this study. Additionally, I examined three alternative policy scenarios to address some of the interaction effects from policy overlap.

7. **Life-cycle emissions intensity: both policies**

If the CAT program modified the compliance obligation for biofuels to account for the up-stream emissions from the production process, the leakage effect of the policy interaction could be mitigated. Therefore, I test a scenario where both policies are implemented at the same stringency level as the both policy scenario but the CAT program uses the life-cycle values for emissions intensity used by the LCFS.

8. **Life-cycle emissions intensity: CAT only**

This scenario takes the same approach as the previous but implements only the CAT program to evaluate the efficiency cost of the LCFS when the leakage effect has been mitigated.

9. **Carbon tax: both policies**

Unlike a cap-and-trade program, combining an LCFS with a carbon tax would likely create additional emissions reductions. This scenario evaluates how changing California's CAT to a carbon tax could improve complementarity with additional policies. Similar to how the policies are likely to interact in British Columbia which has both a carbon tax and LCFS. The carbon tax stringency tested is the same price level computed under CAT-only scenario to provide a direct comparison with the CAT-LCFS interaction in the both policy scenario.

6. Results

6.1. Emissions cap and LCFS

The model simulates alternative policy scenarios from 2010 to 2030, representing the period over which the policies of interest have been defined. In particular, the results presented in this section focus on comparing the both policy scenario with a scenario where no LCFS was applied and only an emissions cap without a price collar was used.

The central simulation results suggest that using an LCFS in combination with an emissions cap leads to increase of more than 125 million metric tons of emissions between 2015-2030, representing a nearly 2.5% increase in total cumulative emissions relative to a scenario using the CAT alone. This suggests that an LCFS can drive meaningful inter-industry emissions leakage, with emissions from alternative fuels not covered by the CAT program increasing as a result of the LCFS. Figure 7 depicts total annual emissions under each main policy scenario.

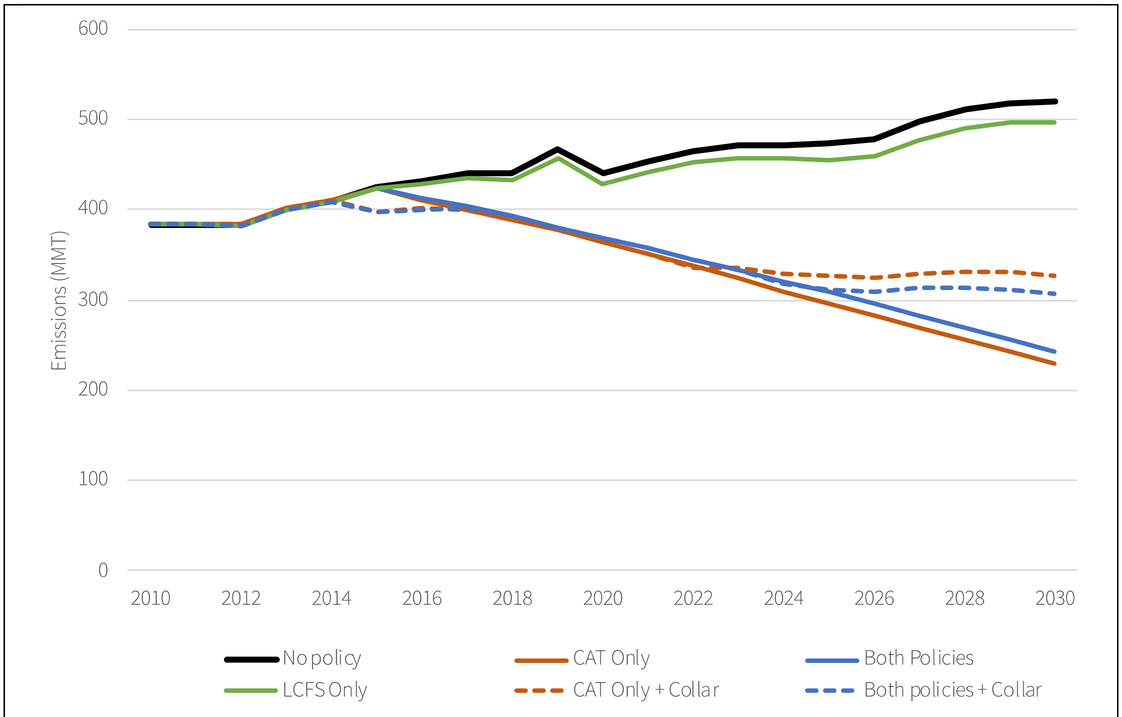


Figure 7. Total annual emissions by scenario

Policy costs are measured via the *equivalent variation*, the change in wealth under a no-policy reference case that would be required to achieve the same level of utility in the policy scenario. Average abatement costs are calculated as the equivalent variation divided by the quantity of emissions abatement achieved under a policy scenario. Central simulation estimates suggest that the average abatement cost is more than 9% higher under the both policy scenario than the CAT-only scenario. This corresponds with the anticipated impact of the waterbed effect, where using an LCFS in combination with the CAT program causes some higher-cost emissions-abatement activities to substitute for lower-cost ones, thereby implying higher aggregate costs per unit of abatement.

The presence of the LCFS reduces the CAT allowance price required achieve compliance with the emissions cap, by effectively doing some of the work required by the cap. However, it increases *average* abatement costs by forcing higher cost abatement. Marginal abatement costs for each policy are revealed as the allowance price, calculated as the shadow cost of the policy constraint. Figure 8 depicts the allowance prices for each policy under the both-policy and single policy scenarios. As expected, the LCFS forcing high-cost abatement lowers the marginal cost of compliance with the CAT program. Conversely, the implementation of the CAT lowers the required LCFS allowance price by reducing the total quantity of fuel demanded, making it easier to attain the required level of low-carbon fuel production to achieve the standard.

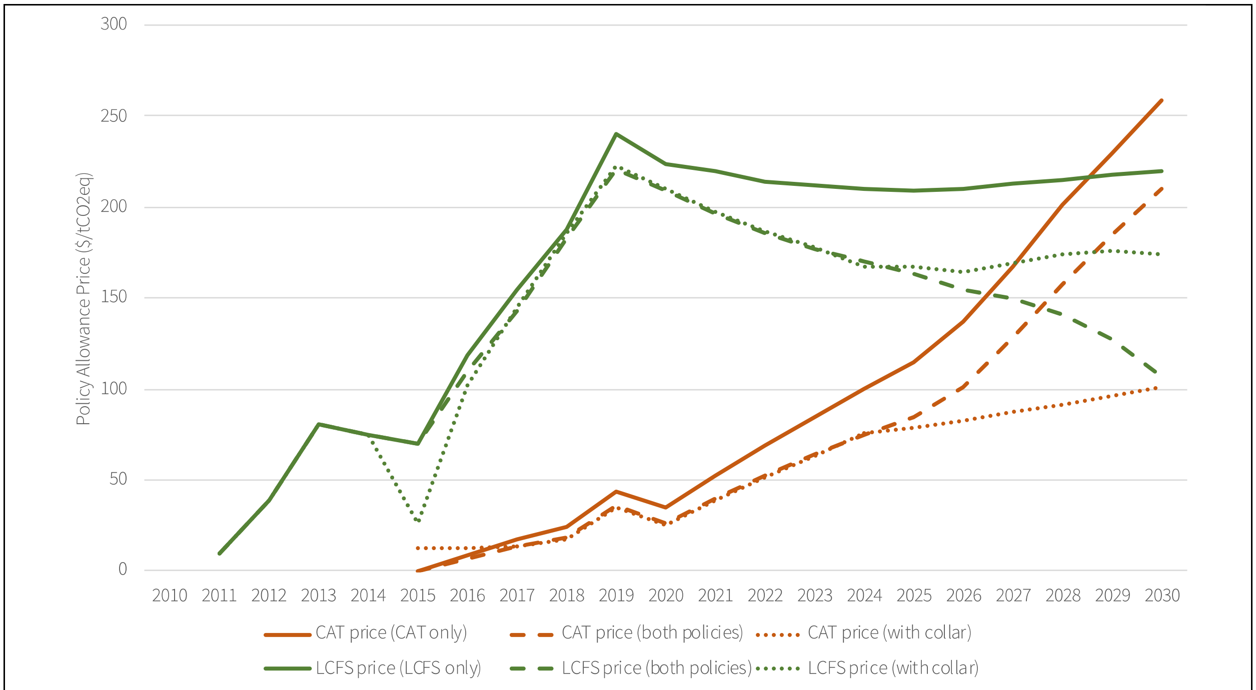


Figure 8. Use of overlapping policies depresses allowance prices

The finding that using an LCFS in combination with an emissions cap leads to an increase in both emissions and cost is robust across a broad range of model specifications. There are numerous uncertainties associated with the specification of the model. Notably, from the utilized elasticities of substitution, the parameterization of natural resource cost and availability in the future, and innovation rates. The results were tested with extreme high and low values for these uncertain parameters. Under all parameterizations, total emissions are found to be between 1.5% and 3.5% higher under the both policy scenario relative to the CAT-only scenario. Likewise, average abatement costs are between 5% and 15% higher under both policies than the CAT alone in all but one specification, which occurs with a high learning rate for renewable diesel, discussed below. A more detailed description of the sensitivity analyses is available in Appendix C.

Induced technological change helps lower policy costs significantly from a no-innovation counterfactual. For example, under the both policy scenario total policy costs are more than three times higher with no induced innovation compared to when innovation is incorporated into the model. In all cases, testing the model with higher innovation rates help lower average abatement costs.

The LCFS does increase innovation in alternative transportation fuels relative to scenarios without the policy. Figure 9 presents the percentage cost declines from induced innovation for each of the seven clean technologies across the central scenarios. The LCFS results in greater cost declines for alternative fuels, in particular renewable diesel.

Innovation rates represented endogenously as learning curves are uncertain and in practice innovation can vary stochastically. Therefore, range of innovation rates were tested based on the limited empirical evidence available.

Notably, when the learning rate for renewable diesel was sufficiently high, prices declined so much that the LCFS became non-binding and average abatement costs were lower under both policies than an emissions cap alone. This likely occurred since little renewable diesel was consumed prior to 2010, and no knowledge spillovers are captured by this model (i.e. knowledge from biodiesel innovation reducing the cost of renewable diesel production). For example, with a learning rate representing a 10% cost decline for each doubling in cumulative production, unit costs for renewable diesel fell by more than 65% by 2030. In this scenario, renewable diesel captured more than 85% of the heavy-duty fuel market in 2030 and the price of LCFS allowances fell to zero starting in 2023. However, in this case emissions leakage was also high. Emissions under the both policy scenario were more than 180 MMT higher under the both policy scenario than the emissions cap alone, representing an increase of more than 3.4% of total cumulative emissions from 2015-2030. This suggests that if gains from innovation are sufficiently high, it may be possible to offset the anticipated higher costs of less efficient policies that target certain technologies or abatement pathways.

The magnitude of emissions leakage depends largely on the relative market share of electric vehicles versus bio-based fuels. Since bio-based fuels have upstream emissions not covered under the emissions cap, while electricity is largely covered by the cap. Therefore, under alternative scenarios with high biofuel innovation or low EV innovation, emissions leakage is found to be greater than 2.5% of cumulative emissions. However, when EV innovation is high and/or biofuel innovation is low, the magnitude of leakage is less than 2.1%. Previous research has noted that bio-based diesel and electric vehicles have emerged as the marginal compliance fuels for the increasingly stringent LCFS and the blend rate of bio-based diesel required will depend heavily on the speed of transition to electric vehicles (Bushnell et al., 2019). This finding emphasizes the importance of EV adoption in achieving compliance with the LCFS and limiting emissions leakage. However, it has been noted that the LCFS is not a particularly well suited policy instrument to encourage vehicle electrification (Bushnell et al., 2021).

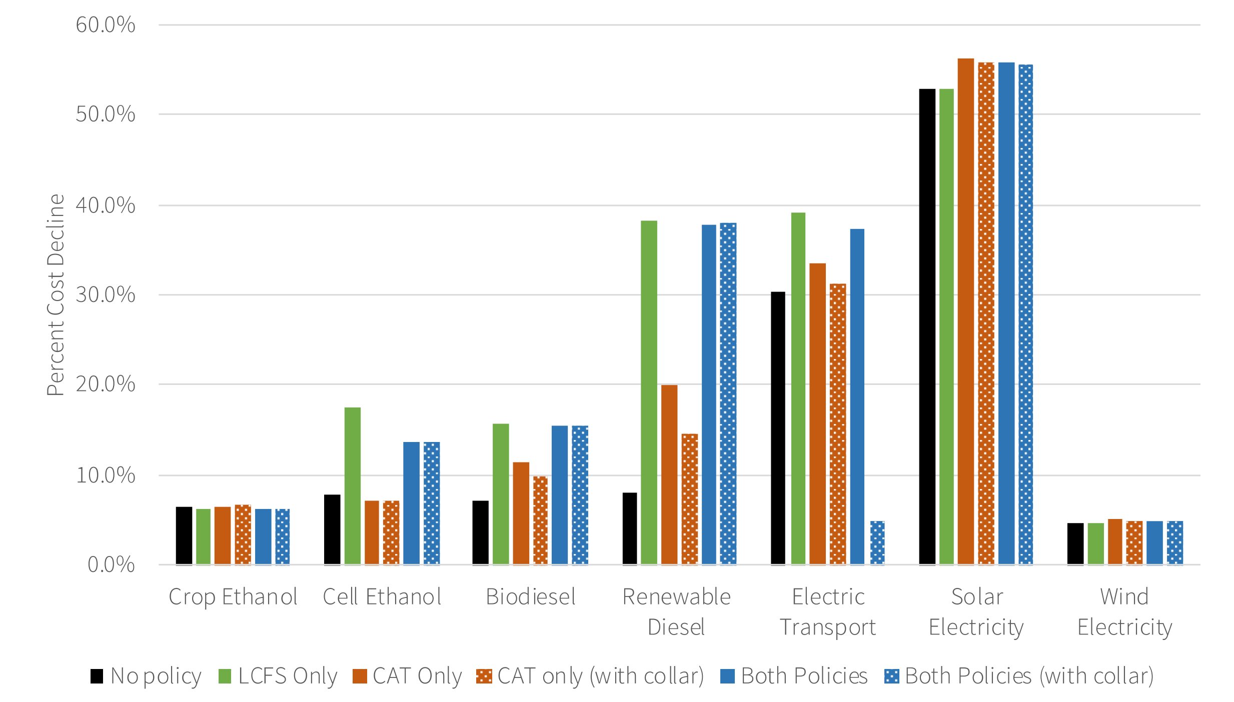


Figure 9. Cost declines from induced innovation

6.2. CAT with a price collar

The findings in Section 6.1 demonstrate how implementing an LCFS in combination with an emissions cap can lead to an increase in both costs and emissions. However, in practice California's CAT program functions as a hybrid price/quantity instrument where a "price collar" limits the possible price range of CAT allowance permits. This includes a price floor, below which allowances will not be sold, and a price ceiling starting in 2021 above which allowances in excess of the cap level may be sold, as discussed in Section 2.2.

Simulations across a wide range of parameterizations suggest that the CAT allowance price would fall below the price floor early in the program and exceed the price sometime between 2020 and 2030 (see Figure 10). Findings suggest a relatively short period during which the emissions cap is binding as a quantity instrument with an endogenously determined price. Rather, when the price collar is binding the CAT program functions as a price-based instrument with the price determined by the floor/ceiling. This corresponds with previous research by Borenstein et al. (2019) who demonstrate that the use of multiple "complementary" policies with the CAT program result in a high likelihood that the CAT allowance price is at either the floor or ceiling in equilibrium.

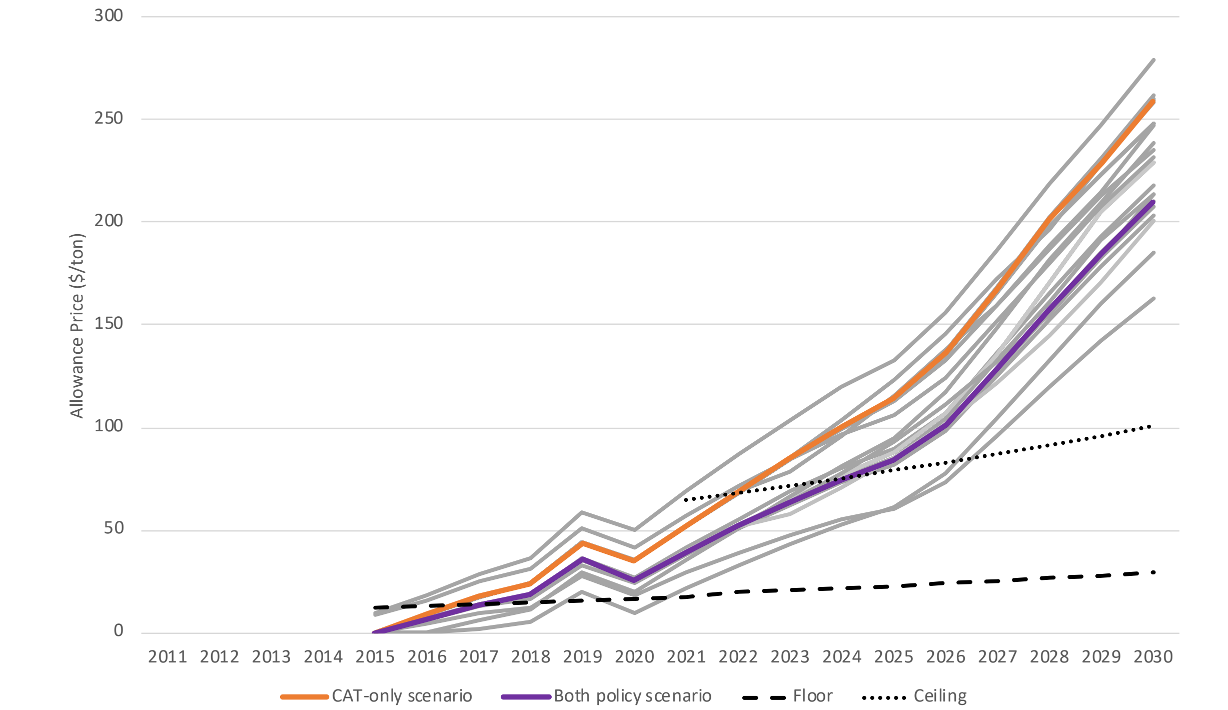


Figure 10. CAT allowance prices relative to the price collar

When the price collar is imposed on the model, the both policy scenario achieves a 26% reduction in total emissions between 2015-2030 relative to no policy; this is slightly larger than the reduction (25%) under CAT with no price collar. The larger reduction reflects the fact that the floor maintains a slightly higher allowance price for a while and thereby induces more emissions abatement. The price collar does not completely eliminate the leakage effect but rather offsets the leakage that occurs when the emissions cap is binding with complementary abatement between the instruments when the floor or ceiling are binding (see Figure 11).

Average abatement costs are 15% higher and total costs are 20% higher under the both policy scenario compared to the CAT alone with the price collar in place. However, compared to no price collar in place, total policy costs of the both policy scenario are 9.4% lower (and more than 26% lower with CAT only).

Table 4. Emissions and cost impacts of the CAT price collar relative to no policy

	Change in Total Emissions		Avg. Abatement Cost ($/ton)	
	Both policies	*CAT-only*	*Both policies*	*CAT-only*
No price collar	-28%	-30%	109.10	98.95
Price collar	-26%	-25%	107.41	91.63

This highlights an important trade-off between total emissions reductions, policy complementarity, and limiting cost impacts. While the price ceiling can effectively limit the costs of the policy and improve complementarity between instruments when binding, it removes the quantity certainty and results in higher emissions by allowing emissions to exceed the level set by the cap.

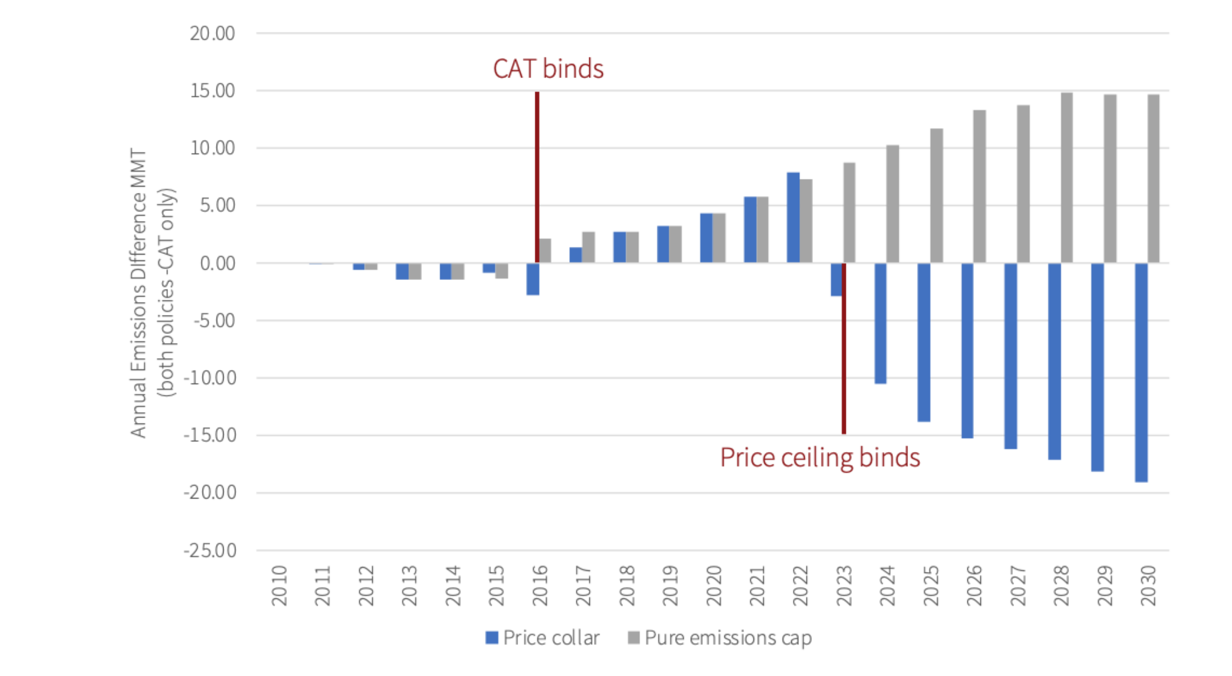


Figure 11. Annual GHG emissions impact of the LCFS with and without the price collar

Recent evaluations suggest that current policies are insufficient to achieve California's legislated emission reduction target of 40% below 1990 levels by 2030 (LAO, 2023). If the current price ceiling becomes binding, as this modeling suggests, then total costs will be limited but emissions abatement will be undermined. Therefore, the price ceiling level can have important implications for whether and how California is able to achieve its emissions reduction goals. For example, increasing the price ceiling to increase by 10% annually (rather than 5%, as currently scheduled) to reach over $140 per ton in 2030 results in an additional 130 MMT of abatement while average abatement costs increase only $0.50 from the current price collar. Notably, this represents 15% lower annual emissions in 2030 (an additional 35 MMT). Although this also extends the period during which the CAT is binding and the LCFS is driving emissions leakage.

Yet, in practice the wide array of additional policies not included in this model and existing bank of allowances may contribute to depress the CAT allowance price, delaying or preventing it from reaching the price ceiling.

6.3. Policy Alternatives

As described in Section 5.6, three alternative policy scenarios to mitigate the negative interaction effects are tested. Alternative scenario results are summarized in Table 3.

A) Life-cycle emission intensity: both policies

Using lifecycle carbon intensity value for fuels under the CAT program can prevent inter-industry leakage by effectively expanding the scope of the CAT to include upstream emissions incentivized by the LCFS. Additionally, this adjustment could help address the structural oversupply of allowances currently in the market. In this scenario, total emissions fell 7.7% below the both policy scenario and average abatement costs were 0.3% lower.

B) Life-cycle emission intensity: CAT only

Although adopting the life-cycle emissions intensity can effectively address the emissions leakage caused by the interaction, the increased costs from the policy overlap are still present. Therefore, it is possible to further reduce policy costs from the LC-Both policy scenario by removing the LCFS. In this case, total emissions fall 7.7% below levels under the both policy scenario and average abatement costs are 7.6% lower. This emphasizes the cost impact of using the LCFS, where even if emissions leakage is avoided the LCFS increases policy costs by more than 7%.

C) Carbon tax + LCFS

As discussed in Section 3, policy interactions are fundamentally different with a price-based policy, such as a carbon tax, than a quantity-based policy such as a CAT program. When used in isolation, the CAT and carbon tax are functionally equivalent when set to achieve the same quantity of emissions reductions. To quantify the effect of this distinct interaction, I implement a carbon tax set at the price level reached by the CAT when used in isolation alongside the LCFS. In this scenario, emissions are 4.9% lower than the both-policy scenario. However, average abatement costs are 1.3% higher because the cumulative incentive forces all abatement lower than the tax level as well as the higher-cost abatement from the LCFS.

6.4. Welfare analysis

To provide a comparison between the alternative policy scenarios discussed, this section evaluates the simulated policy alternatives in terms of policy costs, avoided climate damages, and avoided air pollution damages. As discussed in Section 6.1, policy costs are calculated as the equivalent variation, the change in wealth under a no-policy reference case that would be required to achieve the same level of utility in the policy scenario.

Using an estimate of the social cost of carbon (SCC), we can assess the economic benefit of a policy's GHG emissions reduction in terms of avoided climate damages. The social cost of carbon represents the marginal cost to society of an additional ton of carbon dioxide emitted. The value of social cost of carbon remains uncertain and an active area of research; however, several recent studies suggest that previous values employed by governments in policy decisions represent a significant underestimate (Carleton & Greenstone, 2022). The calculation for climate damages avoided presented in Table 3 employs an SCC value US$185 in 2020, corresponding to recent research by Rennert et al. (2022). A discount rate of 2% is applied to future costs and benefits and alternative values for the discount rate and SCC are tested in the sensitivity analysis.

In addition to reducing emissions of greenhouse gases, climate policy will also impact criteria air pollutants such as nitrous oxides (NO_x), sulfur dioxide (SO_2), fine particulate matter ($PM_{2.5}$), and volatile organic compounds (VOCs). These air pollutants have been shown to be linked to significant human health impacts and increased mortality which can be computed as economic damages using the value of statistical life. However, unlike damages from climate change, damages from criteria air pollutants are spatially heterogenous and depend on dynamics of mixing, dispersion, and population exposure. Explicitly modelling pollution dispersion and exposure dynamics is beyond the scope of this paper; however, better understanding the distributional impacts of air pollution from alternative policy combinations represents an important avenue for future research.

While exposure and corresponding damages of air pollutants are highly uncertain in this aggregated analysis, the available evidence suggests that they can be similar in magnitude to climate damages in some cases (Tong & Azevedo, 2020). Therefore, a range of marginal damage values for each pollutant based on estimates from Tschofen et al., (2019) is used to approximate damages avoided. Emissions factors for air pollutants by fuel are based on average values from the CA-GREET3.0 model used by CARB to determine lifecycle emissions intensity under the LCFS (CARB, 2019a). Air pollutant emissions factors from other sources are calibrated based on economic output from the baseline social accounting matrix with emissions reported for California in the 2011 National Emissions Inventory (US EPA, 2015).

Results of the welfare analysis are presented in Table 3. Net benefits are highest in the Life-cycle both policies scenario, where a cap on emissions (without a price ceiling) is used in combination with an LCFS and bio-based fuels face an obligation under the cap for their life-cycle emissions. This policy combination results in the greatest reduction in both GHG emissions and air pollution damages. Notably, although the use of the LCFS does not alter total GHG abatement from the Life-cycle CAT-only scenario, the additional benefits from avoided air pollution damages are large enough to offset the additional cost of the policy. In contrast, when leakage occurs (as in the both policies scenario without accounting for life-cycle emissions) the additional benefits of avoided air pollution damages are nearly exactly offset by additional climate damages from leakage, while also incurring a greater policy cost.

This highlights the significant role climate policy may play in reducing air pollutant emissions and how such damages represent an important consideration climate policy analysis. Additionally, since air pollutant damages have the potential to alter the preferred policy choice, more precise evaluation of their distribution and impacts from the policy combinations explored in this study represents an important avenue for future research.

The magnitude of the calculated benefits are sensitive to assumptions about the social cost of carbon and discount rate. However, the best performing policy combination across the metrics described in Table 3 remains unchanged unless the SCC chosen is so low as to make any policy intervention a net cost.

Table 3. Alternative policy scenario results relative to no-policy scenario (2010-2030)

Policy Scenario	**Total GHG Emissions Change**	**Climate damages avoided** (billion \$)*	**Air pollution damages avoided** (billion \$)	**Avg. GHG abatement cost** (\$/ton CO_2eq)	**Net benefit** (billion \$)	**Benefit-Cost Ratio**
CAT-only	-30%	\$260.5	\$61.1	\$72.67	\$179.1	2.10
Both policies	-28%	\$246.0	\$75.2	\$80.15	\$156.6	2.01
LCFS-only	-3%	\$26.1	\$28.2	\$175.64	\$16.9	1.42
CAT-only + price collar	-25%	\$216.4	\$43.7	**\$68.51**	\$149.3	**2.18**

Both policies + price collar	-26%	$226.8	$69.6	$79.76	$158.5	2.02
Life-Cycle CAT only	**-34%**	**$294.4**	$70.4	$74.47	$199.4	2.06
Life-Cycle Both policies	**-34%**	**$294.4**	**$88.4**	$80.39	**$202.3**	1.99
Carbon tax Both policies	-32%	$276.7	$83.5	$81.37	$188.2	1.97

**All $ figures in 2010*

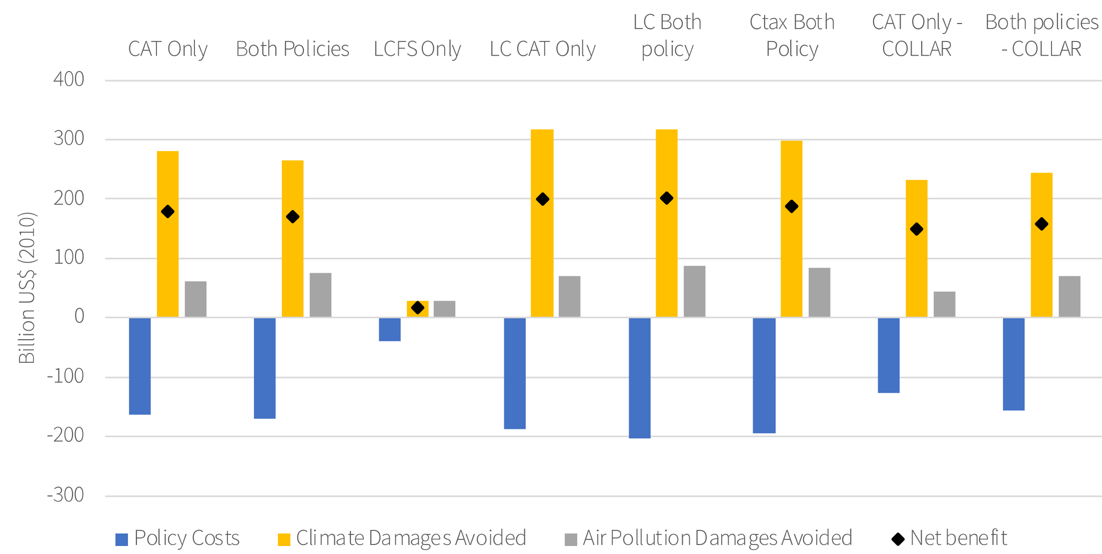


Figure 12. Welfare impacts of alternative policy combinations

7. Discussion and policy implications

California is widely viewed as a climate policy leader with other jurisdictions adopting similar policies. Therefore, it is imperative to critically evaluate the policy instrument choices and designs of California to not only help the state achieve its climate targets at lowest cost, but also so that other jurisdictions may learn from the California experience. The simulations presented here suggest that using an LCFS in combination with an emissions cap can result in both higher emissions and higher policy costs compared to a CAT alone. In practice, California's CAT program functions as a hybrid price-quantity instrument which alters interaction effects. Complementary abatement when the price collar is binding is expected to offset emissions leakage when the emissions cap is binding. However, this results in less total abatement than a pure emissions cap. The policy combination found to maximize net benefits was a pure emissions cap that includes the lifecycle emissions of all covered fuels together with a low-carbon fuel standard to help reduce air pollutants. However, further research on the distribution of costs and air pollution damages is warranted.

The interaction effects explored here have implications for other jurisdictions such as Washington State which is preparing to implement both a "cap-and-invest" program as well as a clean fuel standard. As other jurisdictions seek to enhance their climate policy efforts with a broad-based carbon price, the findings of this study suggest a fixed-price instrument can contribute to greater complementarity with additional climate policy instruments.

This study highlights a potentially important mechanism for emissions leakage from the implementation of overlapping yet incomplete emissions policies. While it could reasonably be argued that the emissions increase from upstream production of alternative transportation fuels predominantly occurs outside the state of California, the California Air Resources has a statutory requirement to minimize leakage under AB32 and has undertaken significant consideration of resource shuffling and emissions leakage from electricity imports under the CAT program (Fowlie & Cullenward, 2018). If the goal of the policy mix is to mitigate emissions and limit the impacts of climate change, then it seems reasonable that the same effort should be made to address this leakage mechanism. Simple solutions such as adjusting the CAT program to require a compliance obligation for fuel suppliers to cover the lifecycle emissions of fuels under the CAT program could close this leakage pathway as well as help reduce the oversupply of emissions allowances in the market.

A potential political economy justification for the use of this overlapping policy approach is that the use of complementary policies is *intended* to depress the CAT allowance price to lower the salience of the costs and maintain the political feasibility of the program. Gasoline represents one of the most salient prices in American society, with prices posted on billboards on many street corners and high fuel prices may undermine support for environmental policy (Aldy & Stavins, 2012). It is therefore worthwhile to consider how alternative policy approaches impact consumer costs such as the price of gasoline. With the price collar in place, simulated gasoline prices are equivalent in the both policy and CAT-only scenario in 2020 and \$0.03 per gallon more expensive with both policies in 2030. In contrast, without price constraints on the CAT program, the gasoline prices faced by consumers are 8% higher in 2020 and more than 50% higher in 2030 under the both policy scenario than when price constraints are imposed. This emphasizes how the choice of policy instruments can alter the policy cost incidence which may have important distributional impacts across societal groups. Better understanding the distributional impacts of such interactions represents an important avenue for future research.

This study also highlights an important issue in the evaluation and attribution of the emissions abatement contribution of a policy: the choice of reference point. If the LCFS is evaluated by comparing the LCFS-only policy scenario to no-policy, the contribution is a 3% reduction in total emissions. However, if we compare the use of the LCFS within the current policy mix relative to the same policy mix without the LCFS, the contribution of the policy nearly halved, representing a 1.6% reduction in emissions. Or when used with an emissions cap without price constraints, even a 2.4% *increase* in emissions.

Some limitations in the analysis deserve acknowledgment. First, the model does not account for all of the complexities of policy design and implementation, particularly with regard to the CAT program. For example, it assumes revenue from CAT is returned to consumers as a lump-sum rebate, when in practice revenue is primarily used to fund other government programs that contribute to emissions reductions and may help depress allowance prices. It also requires annual compliance which fails to account for strategic banking and borrowing of compliance credits as well as the free allocation and excess supply of allowances from earlier periods. In this way, the model may overestimate policy costs, particularly in later years where a bank of low-cost allowances could be used to meet the cap. This could also extend the period over which the CAT is binding rather than the price ceiling and the LCFS is driving emissions leakage.

Second, the model does not explicitly incorporate a number of other policies that may also put downward pressure on the CAT allowance price by forcing certain abatement activities, such as the state's renewable portfolio standard and the federal renewable fuel standard. However, in the simulations the targets of the renewable portfolio standard are met under any policy scenario with some form of carbon price. The model also does not incorporate representation of the ZEV purchase incentives or the recently announced Advanced Clean Cars II regulation which mandates the share of new vehicles sold to be ZEVs to increase to 100% by 2035 (CARB, 2022e). Future research examining the interaction effects of additional policies could further shed light on the role of alternative policy instruments within the climate policy portfolio.

Third, the representation of innovation in the model is simplified and incomplete. Experience curves have been shown to be useful for modelling technological change but do not represent an identified causal relationship. Therefore, attributing cost declines to certain policy adoption may not be justified. Additionally, technological innovation is also occurring elsewhere over the time period modelled. This model does not represent the development of technologies in other jurisdictions and is constrained by the assumptions that knowledge spillovers are limited to the U.S. and the technology market share of California within U.S. consumption remains constant at 2010 levels.

8. Conclusion

This study examines the interaction effect between two overlapping climate policies – a cap-and-trade program and low carbon fuel standard – in terms of emissions abatement, cost effectiveness, and innovation. A low carbon fuel standard is a performance standard that mandates a declining emissions intensity of the transportation fuel supply and allows fuel suppliers flexibility to trade credits to reach compliance. A CAT program sets a limit on the total emissions from electricity, natural gas, transportation, and heavy emitting facilities.

The interaction between these two overlapping policies can lead to unintended consequences for costs and emissions. This occurs because the LCFS forces high-cost abatement under the cap that could otherwise be achieved for lower cost under the CAT alone. Additionally, because the LCFS incentivizes the production fuels which generate emissions not covered by the CAT program, it can lead to a net increase in emissions through inter-industry leakage.

To examine the extent of this interaction, I constructed a CGE model calibrated to the case of California over the period 2010-2030. Simulation results suggest that the use of both an LCFS and an emissions cap can lead to 2.4% higher total emissions and more than 9% higher average abatement costs compared to using the emissions cap alone. When incorporating the price collar on emissions allowances in California's CAT program, the LCFS adds complementary abatement when the price collar is binding and creates inter-industry emissions leakage when the emissions cap is binding. This results in a net emissions reduction of 1.25% from the use of the LCFS, however the price ceiling limits the total emissions abatement achieved. While the LCFS does increase innovation in alternative transportation fuels, lower production costs from innovation are unlikely to compensate for the additional cost of overlapping policy.

Several policy reforms are available that could mitigate the negative interaction effects identified in this study. Specifically, a compliance obligation for the life-cycle emissions intensity of alternative transportation fuels under the CAT program could avoid the inter-industry leakage effect. Alternatively, a more price-based approach rather than a quantity-based instrument could improve the complementarity of the carbon pricing system with additional climate policies. These findings have important implications for California to achieve its emissions target cost effectively as well as for other jurisdictions implementing similar policy combinations.

The climate policies examined in this study also impact emissions of criteria air pollutants and their associated health consequences. Incorporating damages from air pollutants has the potential to alter the preferred policy choice and further research on expected air pollution exposure and impacts is warranted.

Future research can extend this work in a number of additional directions. For example, incorporating additional policies can help quantify the relative contributions and interaction effects of a range policy instruments within a complex climate policy portfolio. It is also important to examine how shifting the distribution of costs through overlapping policy may impact households of different demographic socio-economic status.

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