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Executive Summary

Todd Schatzki and Robert N. Stavins[1]

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Like many other states, Oregon has begun to pursue climate policies to attempt to fill the gap created by the lack of effective climate policy at the Federal level. After adopting a variety of policies to address climate change and other environmental impacts from energy use, Oregon is now contemplating the adoption of a greenhouse gas (GHG) cap-and-trade system. However, interactions between policies can have important consequences for environmental and economic outcomes. Thus, as Oregon considers taking this step, reconsidering the efficacy of its other current climate policies may better position the state to achieve long-run emission reductions at sustainable economic costs.

1. A Well-Designed GHG Cap-and-Trade Program is a Better Approach to Regulating GHG Emissions Than Alternatives

A GHG cap-and-trade system offers many advantages compared with other approaches to reducing GHG emissions. By capping total emissions, a cap-and-trade system provides a high level of emissions certainty. By comparison, policies that target particular activities through standards do not achieve any particular emission target with certainty.

In addition, cap-and-trade systems achieve emission reductions at a lower cost than other regulatory approaches by creating a uniform incentive that encourages emission reductions through the least-costly approach. Thus, cap-and-trade creates incentives for sources to undertake the least-costly emission reductions, while forgoing more costly options.

Development of a well-designed cap-and-trade system requires careful attention to the details. Prior legislative proposals in Oregon have included elements of a well-designed GHG cap-and-trade system, such as broad economy-wide coverage and flexibility to allow sources to use offsets to cover a portion of compliance obligations. However, many key program design decisions will be left to the program's regulator–the Oregon Environmental Quality Commission. **Thus, decisions made in the rulemaking process will have important implications for the program's eventual performance and possible success.**

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2. Well-Designed Complementary Policies Can Improve Environmental and Economic Outcomes, Although Some Complementary Policies Raise Costs with Little (or No) Environmental Benefit

When developing a climate policy, states may consider pursuing reductions through multiple "complementary" policies that target individual activities that produce emissions. But what appears to be logical, can turn out to be ineffective or worse yet, counter-productive.

Complementary policies *may* provide economic benefits when they target market failures *apart from* the GHG emission externality, or when they target sources that are not covered by the cap-and-trade system. Many other legitimate market failures affect the climate change problem. For example, energy efficiency programs may target distorted incentives to invest in building energy efficiency. Targeting this additional market failure can achieve net cost savings from reduced energy use, while also producing the ancillary benefit of reducing GHG emissions.

On the other hand, overlap between complementary policies and the GHG cap-and-trade program can have perverse consequences. **In general, when state-level policies overlap with cap-and-trade, the complementary policies will fail to create any additional emission reductions.** When a binding cap-and-trade system is in place, aggregate emissions will equal the cap whether or not complementary policies are implemented. Under such conditions, complementary policies produce no incremental emission reductions, but simply shift emissions among sources or sectors covered by the cap (or worse, as we discuss below).

Complementary policies may increase the cost of meeting emission targets when implemented alongside cap-and-trade. If complementary policies require that more costly emission reductions be undertaken, then the shift from lower-cost to higher-cost reduction activities increases the overall cost of achieving emission targets. Thus, unless a policy targets non-GHG market failures or sources not covered by the cap, it likely raises costs (or is not binding).

Further, complementary policies may depress cap-and-trade allowance prices. When complementary policies require emission reductions that are more costly than would be required under cap-and-trade, the requirements reduce the quantity of emission reductions required to meet the cap, thus depressing the price of allowances and reducing incentives for technological change.

In fact, many of Oregon's climate policies would overlap with an economy-wide GHG cap-and-trade program. These policies will likely add little (or no) change in emissions once a cap-and-trade program is adopted. They include the Renewable Portfolio Standard and Clean Fuels Program. However, other policies may provide some economic benefits by targeting non-GHG market failures, such as the Energy Trust of Oregon, which funds residential and commercial energy efficiency programs.

3. Consequences of Policy Overlap: Lessons from California for Oregon

California is several years ahead of Oregon in the adoption of its climate policies, and thus can provide valuable lessons for Oregon. **Analysis of California's Low Carbon Fuel Standard (LCFS) shows that it actually *increases* emissions relative to cap-and-trade alone, while also *increasing* costs.** From 2013 to 2017, estimated costs were over $1 billion. However, as shown in **Figure ES-1**, since the cap-and-trade program was expanded in 2015 to include transportation fuels, emissions outside of California (and outside the state's GHG cap-and-trade system) have *increased* by more than 1.8 million $MTCO_2e$ (through 2017). Moreover, as shown in **Figure ES-2**, the differences in costs between programs

are dramatic; while GHG cap-and-trade allowance prices have been below $16 per $MTCO_2e$, LCFS program credit prices have risen to nearly $180 per $MTCO_2e$, more than a 11-fold difference.

Figure ES-1. Aggregate Change in Emissions from California's LCFS

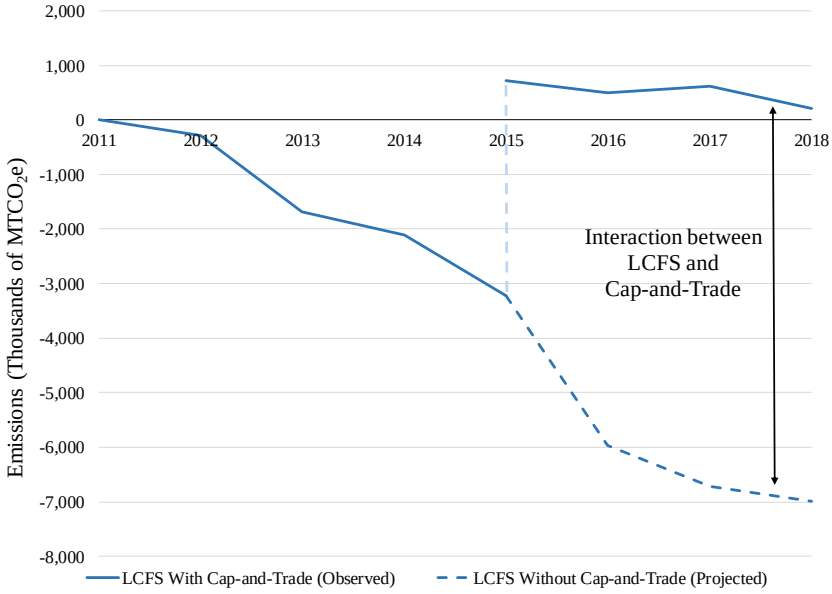
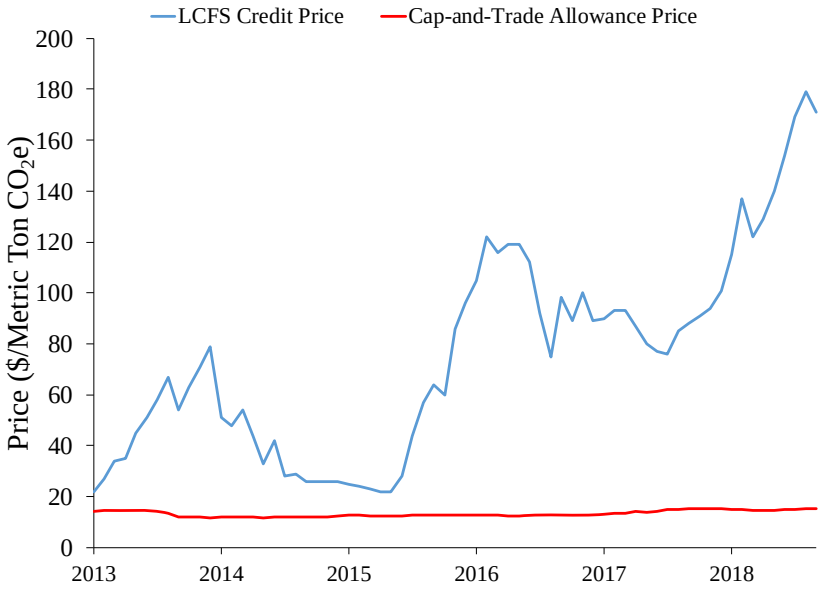


Figure ES-2. California's LCFS Credit Prices vs. Cap-and-Trade Allowance Prices



Some have tried to justify these high costs and negligible environmental impacts by claiming that the LCFS is a "technology" policy aimed at "spurring innovation." While measuring innovation is complex, it should be noted that compliance with the LCFS has largely been achieved through pre-existing technologies. It is unclear to what degree, if any, improved efficiencies ("learning by doing")

have been achieved through the demand for renewable fuels created by the LCFS. Moreover, LCFS costs are comparable to all federal spending on renewable energy, raising the question of whether the LCFS is the best use of society's resources from the standpoint of investment in promoting energy technology innovation.

4. Next Steps for Oregon Climate Policy

As Oregon contemplates the adoption of cap-and-trade, it has several options for its suite of climate policies. One approach maintains all policies, as currently designed. Our analysis shows that, due to interactions among overlapping climate policies, retaining certain complementary policies could be very costly without achieving any incremental environmental benefits.

A second option would be to develop a GHG cap-and-trade program of sufficient stringency to achieve targeted emissions or allow prices to rise to the social cost of carbon, and end complementary policies that do not produce incremental benefits by addressing market failures unrelated to the GHG emission externality or regulating sources not covered by the cap. This approach could begin by undertaking a thorough assessment of the likely interactions among overlapping climate policies and the extent to which policies address market failures unrelated to GHG emissions. The feasibility of this approach will depend on how aggressively Oregon can pursue carbon pricing.

A third approach is a hybrid of these approaches. While economic analysis unambiguously shows that policies relying on GHG emission pricing, such as GHG cap-and-trade, are the most cost-effective approach to achieving emission targets, political realities may not support the immediate adoption of climate policies relying largely (if not solely) on carbon pricing. But the costs of pursuing aggressive GHG emission reductions goals through more-costly complementary policies will grow over time, which makes that path not only costly but politically risky. The hybrid option involves a transition to increased reliance on GHG cap-and-trade by diminishing the reliance (i.e., stringency) of some complementary policies and gradually (or even quickly) shifting to the uniform-price incentives created by cap-and-trade.

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Oregon is contemplating the adoption of a greenhouse gas (GHG) cap-and-trade system.[3] For example, Senate Bill 1507, also known as Oregon's Clean Energy Jobs bill, would create a GHG cap-and-trade system for major sources of GHG emissions.[4] The GHG cap-and-trade system would add to the existing policies Oregon has adopted to address climate change and other environmental impacts from energy use. Like many other states, Oregon has begun to pursue climate policies to attempt to fill the gap created by the lack of effective climate policy at the Federal level.

In this paper, we evaluate Oregon's proposed GHG cap-and-trade system and consider its implications for other climate policies Oregon has already adopted. Section I starts by discussing the benefits of cap-and-trade as an approach to addressing climate change. In Section II, we discuss "complementary" policies states are developing to address GHG emissions. Under certain conditions, such additional policies can improve environmental and economic outcomes. However, due to interactions between policies, some complementary policies raise costs and fail to achieve emission reductions. We identify the conditions that lead to these different outcomes, and discuss how the particular policies currently in place in Oregon would interact with the addition of a GHG cap-and-trade system. In Section III, we analyze certain climate policies in California to identify the impacts of interactions that Oregon might expect from its suite of policies. In particular, we examine California's Low Carbon Fuel Standard (LCFS), including its interactions with California's GHG cap-and-trade system.

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[3] Oregon Department of Environmental Quality, "Considerations for Designing a Cap-and-Trade Program in Oregon," February 14, 2017

[4] 79th Oregon Legislative Assembly, Senate Bill 1507, Ordered February 16, 2018.

I. BENEFITS OF GHG CAP-AND-TRADE SYSTEMS

A cap-and-trade system limits (caps) the total emissions permitted from a designated set of sources. By reducing the cap over time, emissions are reduced from current levels to meet policy objectives. Cap-and-trade systems have been widely applied to GHG emissions. At present, there are approximately 21 systems covering emissions at the state, provincial, national, or regional level.[5] A cap-and-trade system can cover a large fraction of economy-wide emissions, because the energy sources that account for most emissions can be regulated through a relatively small number of sources. For example, California's GHG cap-and-trade system covers approximately 85% of state-wide GHG emissions by regulating emissions from electric power generators, large industrial facilities, and suppliers of natural gas and other fuels.[6]

By capping total emissions, a cap-and-trade system provides a high level of emission certainty. By comparison, policies that target particular activities through standards do not achieve any particular emission target with certainty. For example, a low carbon fuel standard may reduce fuel carbon-intensity, but it does not affect the number of miles driven or vehicle fuel efficiency. Thus, total emissions may increase even if carbon-intensity is falling.

Cap-and-trade systems achieve emission reductions at a lower cost than other regulatory approaches. By imposing a cost on activities that generate emissions, cap-and-trade creates a uniform incentive that encourages emission reductions through the least-costly approach. Sources that can reduce emissions at a cost less than the cost of emission permits (allowance prices) will take steps to reduce emissions, while sources that can only reduce emissions at a cost greater than allowance prices will not take such action. Because allowances used to comply with the cap-and-trade system are tradeable among regulated sources, allowances can flow to sources as needed to cover emissions.

Legislative proposals in Oregon (e.g., HB 4001, SB 1507) specify many elements of the GHG cap-and-trade design, but also leave many features for the regulator, the Oregon Environmental Quality Commission (EQC), to determine. These proposals include features of a well-designed GHG cap-and-trade system, and, when providing the EQC with rule-making discretion, do not preclude potentially valuable design features. But, as with any complex regulation, the design details that need to be worked out during this rulemaking process would be critical to determining the eventual effectiveness of the policy.

In these proposals, the program would cover all sectors of the economy that are easily regulated through a GHG cap-and-trade system, including large point sources and fuels, such as natural gas, gasoline and diesel. Sources outside the proposed program are generally more difficult to monitor and enforce, thus making regulation through other measures more promising.

Proposed legislation can accommodate key design features to take advantage of "when" and "where" flexibility, although such features must be developed during the rulemaking process. Because GHG emissions are long-lived "stock" pollutants, the timing of emissions is less critical to the damages they create than is the case with many other pollutants (e.g., criteria air pollutants). Thus, well-designed cap-and-trade systems include banking and multi-year compliance periods to allow sources

[5] ICAP. (2018). Emissions Trading Worldwide: Status Report 2018. Berlin: ICAP.

[6] Center for Climate and Energy Solutions, "California Cap and Trade," March 16, 2018.

flexibility over when emission reductions are made.[7] Further, because the impact of GHG emissions is independent of where emissions occur, systems that include linking and offsets can lower the total costs of achieving emission goals. The legislation includes specific provisions that permit the EQC to link Oregon's programs with other systems and allow sources to use offsets to fulfill up to 8% of their compliance obligation.[8]

The proposed system includes an Allowance Price Containment Reserve, designed to help contain the costs of compliance. The Reserve holds a finite quantity of allowances that are released only when prices rise to a predetermined "trigger" price level. **The Reserve can help mitigate costs and allowance price volatility in the event that there is a sudden increase in demand that would lead to a spike in allowance prices.**

However, the proposed cap-and-trade system does not include an explicit price cap that could provide a "safety valve" in the event that demand for allowances suddenly increases. By itself, the Reserve will not limit prices from rising to economically (and politically) unacceptable levels. Because the Reserve holds a finite quantity of allowances, once the Reserve is exhausted, allowance prices can continue to rise unabated.

A price cap has many benefits.[9] A price cap sends a clear signal to the market about the range of prices that could prevail in the future. It also provides market stability, because absent a price cap, there is a risk that a sudden increase in prices undermines political support for the policy. In the past, the failure of policies to include a safety valve has led to the suspension of emission trading programs when prices suddenly rose to high levels, such as occurred in the RECLAIM program in California's South Coast Air Quality Management District.[10]

California recently adopted a price cap. In its draft rulemaking, the California Air Resources Board (CARB) has set the price cap at $65 per $MTCO_2e$ in 2021. The price cap would rise at a rate of 5% plus inflation. It is anticipated that CARB will finalize these rules this year.

In many respects, the GHG cap-and-trade proposals mirror systems already in place in California and Quebec. This builds on experience gained with design of GHG cap-and-trade systems in California. If sufficiently similar, Oregon could link its system to the California system and other systems (e.g., Quebec), if desired. Linkage can lower the total economic cost of achieving emission targets by expanding the geographic scope of emission reductions opportunities.[11]

[7] SB 1507 does not specify the length of compliance periods and does not explicitly allow allowance banking.

[8] SB 1507, Section 17. Offset Projects; SB 1507, Section 19. Linkage with market-based compliance mechanisms in other jurisdictions.

[9] SB 200. *See* Schatzki and Stavins, 2018.

[10] The RECLAIM trading in the South Coast Air Quality District was dismantled after significant price spikes for RECLAIM allowances contributed to a broader crisis in California's electricity markets.

[11] Stavins, Robert N., Judson Jaffe, and Matthew Ranson, "Linking Tradable Permit Systems: A Key Element of Emerging International Climate Policy Architecture." Ecology Law Quarterly 36(2010):789-808.

The compliance instruments -- allowances -- used by sources to comply with a cap-and-trade system have substantial economic value.[12] Thus, a key decision for legislators in developing a cap-and-trade system is determining how these allowances will be allocated. This can affect both the aggregate economic impact of the cap-and-trade program, as well as the distribution of its economic outcomes across businesses and consumers.

Legislators have two basic options: freely allocating allowances to particular entities, or selling allowances through auction. HB 4001 / SB 1507 proposes to allocate allowances through both of these mechanisms. Some allowances would be allocated directly to electric and natural gas utilities and emission-intensive, trade-exposed industries. These direct allocations have two distinct purposes. **Direct allocations to emission-intensive, trade-exposed industries through an updating, output-based allocation can offset the risk that the GHG cap-and-trade program leads to emission leakage.**[13] Emission leakage occurs when economic activity shifts locations due to higher regulatory costs.

When allowances are allocated directly to regulated utilities, the allowance value is used to lower customer bills to offset the impact of cap-and-trade on consumer energy costs. While this approach reduces customer rate impacts, it also reduces energy customers' incentives to reduce energy use. **Thus, an alternative approach that returns allowance value to customers in a lump sum dividend can help offset program costs while preserving the (marginal) incentives for energy consumers to reduce their energy use. This approach may also address distributional concerns, as carbon pricing tends to disproportionately affect lower-income households.**

Under SB 1507, as proposed, allowances that are not allocated directly would be sold through an auction, with the government retaining the auction revenues. **When selling allowances through an auction, the resulting economic gains and their distribution throughout the economy depends on decisions made by the government about how revenues are used.** The cost of a cap-and-trade program is minimized when auction revenues are used to offset pre-existing distortionary taxes, such as income taxes. This path was taken in British Columbia, which lowered several types of pre-existing taxes to offset new revenues from its carbon tax, including personal income taxes, corporate income taxes, and industrial property taxes.[14]

Legislative proposals in Oregon specify particular uses for the auction revenues, including road and highway maintenance, public education (per existing articles in the State Constitution), projects

[12] Schatzki, Todd and Robert N. Stavins, "Using the Value of Allowances from California's GHG Cap-and-Trade System", Regulatory Policy Program, Mossavar-Rahmani Center for Business and Government, Harvard Kennedy School, August 27, 2012.

[13] SB 1507 indicates that direct allocations to emission-intensive, trade-exposed industries should use an "output-based benchmarking methodology". To mitigate leakage, direct allocations must reflect an "updated" estimate of each firm's level of economic activity during the compliance year. Direct allocations that are fixed (even if based on historical output levels) fail to create an incentive for firms to pass-through the value of the direct allocation to their customers, which increases their ability to compete with firms that are not covered by the cap-and-trade system.

[14] For example, corporate income taxes were reduced from 12% prior to the program to 11% in 2008, 10.5% in 2010 and 10% in 2011. In 2008, corporate income taxes to small business were reduced from 4.5% to 2.5%, and the threshold for the small-business tax rate was raised from $400,000 to $500,000.

aimed at achieving the bill's objectives, and transitioning workers in affected communities.[15] Road and educational spending reflects requirements in the Oregon Constitution given the nature of the revenues being collected.

Like Oregon's proposal, many cap-and-trade programs use auction revenues to support projects aimed at reducing GHG emissions. Such spending may seem natural given the goals of climate policy. However, care is needed when selecting projects and activities to achieve environmental and economic benefits. To achieve reductions in GHG emissions, such spending should target sources outside the cap or programs that address market failures unrelated to GHG emissions. Below, we elaborate on these conditions, as they pertain to complementary policies. But the same logic holds for revenue spending: spending to reduce emissions from sources covered by the cap will not reduce total emissions because the cap remains unchanged. Instead, such spending shifts where emissions occur under the cap and subsidizes spending on emission reductions activities that otherwise would be made solely due to the cap-and-trade price signals.

II. STATE CLIMATE POLICIES

In the wake of a lack of Federal leadership on climate policy, some states have sought to develop their own policies, often in coordination with other states (and provinces). These state climate initiatives often take a "belt and suspenders" approach that includes a suite of policies targeting different activities that generate GHG emissions. This approach can aim to address each activity that produces GHG emissions through one or more measures, sometimes regardless of the merits of each policy or the interactions among the policies.

Oregon already has enacted several other policies intended to reduce GHG emissions, including:

- ***Clean Fuels Program (CFP)***. The Clean Fuels Program is a standard designed to lower the carbon intensity of transportation fuels. The CFP requires reductions in the average fuel carbon-intensity below a baseline level. As regulated by the program, carbon-intensity reflects "life-cycle" emissions that include tail pipe emissions, emissions sequestered in the process of growing fuel crops (for renewable fuels), and emissions created during fuel production. Fuel suppliers can comply with the standard by selling a mix of fuels with an average carbon-intensity below the cap (i.e., "over-complying"), or by purchasing credits generated by suppliers that have over-complied with the standard. The program was implemented in 2016.
- ***Renewable Portfolio Standard (RPS)***. Oregon's RPS requires that 50% of electric power used in the state be generated from renewable sources of electricity by 2030. Renewable energy sources include technologies such as wind power, solar power, geothermal power, small hydropower, certain biomass products, and power generated with landfill gas.
- ***Sustainable Transportation Initiative***.[16] This initiative is an integrated statewide effort to reduce GHG emissions from the transportation sector. Efforts include: a Statewide Transportation Strategy; GHG emission reduction targets for metropolitan areas; land use and transportation

[15] SB 1507, Sections 26 and 28.

[16] https://www.oregon.gov/ODOT/Programs/Pages/OSTI.aspx.

scenario planning guidelines; and tools that support local governments in reaching their emissions reduction goals.

- ***Coal-to-Clean Law***. This law requires that the state's electric utilities eliminate coal-fired electricity from their mix of energy generation by 2030.
- ***Energy Trust of Oregon***. The Energy Trust of Oregon provides information, cash incentives and technical assistance to help Oregon utility customers invest in energy-saving or renewable energy projects. Its services and support are available to both residential and commercial customers. The Trust is funded by charges included in electric and natural gas utility customer bills.

Below, we identify the key conceptual issues affecting decisions to develop policies to complement a GHG cap-and-trade program. First, we identify the conditions under which complementary policies can improve environmental and economic outcomes, particularly by addressing problems ("market failures") not addressed by cap-and-trade and by targeting emission sources not targeted by cap-and-trade. Next, we consider when interactions between GHG cap-and-trade systems and other policies are problematic, raising costs and failing to achieve emission reductions. Finally, we consider options policymakers have when political conditions do not support setting carbon prices at sufficiently high (efficient) levels.

A. Economic Principles for Complementary Policies

From an economic perspective, the primary purpose of regulatory interventions is to remedy market failures that prevent markets from arriving at economically efficient outcomes. If a regulation can create positive net benefits (benefits greater than costs) by addressing a market failure, without imposing excessive costs or unintended consequences, economic welfare can be improved.

The key market failure contributing to health and environmental impacts is the failure of households, businesses, and industry to account for these impacts in their energy use decisions. That is, energy prices do not reflect the true social costs of energy use. As a result, energy use and associated impacts are too high from the standpoint of society as a whole.[17] This problem has been well studied, and **there is universal consensus that the most efficient approach to this problem is to set energy prices at their true social costs through environmental prices, such as carbon prices created through a cap-and-trade system.**

For climate change, the economic cost (damages) of additional GHG emissions are measured by the social cost of carbon. Estimates of the social cost of carbon were developed by the United States Government's Interagency Working Group (IWG) on the Social Cost of Greenhouse Gases. Developed to provide United States' regulatory bodies with a consistent estimate of the social cost of carbon for use in regulatory analyses,[18] the IWG's estimates of the social cost of carbon have become a standard

[17] In addition, energy may be underpriced for a variety of other reasons, which could also lead consumers to use excess energy.

[18] *See* Technical Support Document: Social Cost of Carbon for Regulatory Impact Analysis Under Executive Order 12688, Interagency Working Group on Social Cost of Carbon, United States Government, February 2010 ("TSD 2010") at 1-4. There have been a range of estimates produced of the social cost of carbon. *See e.g.* National Research Council, 2009, *Hidden Costs of Energy: Unpriced Consequences of Energy Production and Use*, Washington, DC: The National Academies Press at 216-219 for a brief review.

benchmark, used by many other regulators, including CARB.[19] **The IWG's most recent estimates indicate that the social cost of carbon from emissions occurring in 2030 would range from $25 to $115 per metric ton (in nominal dollars), depending upon the choice of discount rate used to convert the future damages created by those emissions into present value terms.**[20] For example, the damages from 1 metric ton of emissions in 2030 would be $79 in 2030 dollars when the future impact of those emissions are discounted back to 2030 at a 3% discount rate. These social cost of carbon estimates represent the global damages to various sectors, including agriculture and energy dependent sectors, climate driven human health impacts, damages from sea-level rise, and impacts to ecosystem services.[21]

However, the failure of energy prices to reflect true environmental costs is not the only market failure relevant to climate policy. From an economic perspective, the criteria for complementary policies is relatively clear in principle. **Complementary policies *may* provide economic benefits under one of two conditions**:[22]

1. The complementary policy targets market failures *unrelated* to the GHG emission externality; or
2. The complimentary policy targets sources that are not under the cap of the cap-and-trade program.

In each case, particular complementary policies must still be shown to provide positive net benefits and be preferred to other alternatives.

We turn first to the rationale for complementary policies that there are market failures present which are *unrelated* to the GHG emission externality. Several different types of market failures are particularly relevant to the climate change problem:

[19] CARB relies on the IWG's estimate that employs a 3% discount rate. CARB also converts this value to 2015 dollars, accounting for inflation. *See* California Air Resources Board, California's 2017 Climate Change Scoping Plan, November 2017 ("2017 Scoping Plan") at 40 and fn. 97; *see also* CARB, "Preliminary Concepts" February 2018. at Table 5, available at https://www.arb.ca.gov/cc/capandtrade/meetings/20180302/ct_price_concept_paper.pdf.

[20] See TSD 2016 at 4. The IWG also presents a set of higher estimates reflecting more extreme assumptions regarding the underlying modeling inputs. This higher set of estimates places the 2030 social cost of carbon at $240 in 2030 (in $2030). The IWG reports the social cost of carbon in $2007. We convert $2007 to $2030 using historical annual average CPI values for all urban consumers provided by the BLS (https://www.bls.gov/cpi/tables/supplemental-files/home.htm) and forecasted CPI values that we derive from forecasted year to year (specifically Q4 to Q4) percent changes in the CPI presented by the 2018 Economic Report of the President, (https://www.whitehouse.gov/wp-content/uploads/2018/.../ERP_2018_Final-FINAL.pdf, Table 8-1, column 4).

[21] These values derive from three integrated assessment models (IAM) that assess how changes in greenhouse gas driven temperatures impose costs and various impacts. All models also contain some characterization of adaptation, and in various ways capture catastrophic or extreme climate change driven impacts. *See* TSD 2010 § III.A for further detail regarding the models underlying the social cost of carbon estimates. *See also* TSD 2016 § II for further detail regarding updates to these models that underlie the most recent social cost of carbon estimates. For further details regarding the process IWG followed in estimating the social cost of carbon, see TSD 2010 § III, IV.

[22] Bennear, Lori and Robert N. Stavins, "Second-best theory and the use of multiple policy instruments," *Environmental and Resource Economics* 37(1): 111-129, 2007.

- ***Information Problems***. When market participants fail to have accurate information about a product's attributes, they can make decisions that do not account for the true costs and benefits of alternative choices. Two types of information problems are of particular concern.[23] The *principal-agent problem* arises when one party makes decisions with financial implications for another party. For example, building owners may not make investments in energy efficiency if they lease to tenants that pay their own utility bills, since the tenant will keep the cost savings; likewise, renters may not make such investments, because there is a high likelihood they will move and lose out on future energy savings. Informational problems also include *asymmetric information*, which arises when one party to a transaction has more information than others.

- ***Behavioral failures***. Behavioral market failures refer to market outcomes that derive from actions that diverge from what economists have typically defined as rational behavior. For example, consumers seem to require higher compensation for giving up a good than their expressed willingness to pay for the same good. Behavioral failures have been invoked as an explanation for the *apparent* failure of households and businesses to adopt cost-effective energy efficient technologies – that is, technologies that produce energy cost savings that exceed the cost of technology adoption.[24]

- ***Innovation Spillovers***.[25] Achieving ambitious climate goals will require substantial innovation in energy technologies to reduce their GHG emissions, while continuing to provide the many benefits created by use of energy. Such innovation includes both development of new technologies as well as increases in the efficiency and reductions in the cost of existing technologies.

  Innovation leads to positive knowledge spillovers as ideas from research and development (R&D) flow into and enhance other R&D activities. Even if new innovations have legal protections such as patent exclusivity, innovators cannot capture all of these spillovers. Because innovators do not reap all of the rewards created by their innovation, private incentives to invest in R&D are below the socially optimal level.

- ***Congestion Externalities*.** Socially inefficient levels of traffic congestion lead to many costs, including excess fuel use and emissions, and lost time. However, efficient congestion pricing

[23] A third problem is related to the *"public good"* aspect of information: once created, information can be used by many people at little or no additional cost. Because it may be difficult to limit access to information, the incentive for any individual to develop information is reduced. Consequently, general information about energy efficiency may be underprovided. However, this public good attribute does not diminish the incentive for any individual market participant, such as a building owner, to supply information about their own product (or building), since this information can distinguish their products from competitors' offerings. Building labeling has no obvious impact on this potential problem.

[24] Stavins, Robert N., Todd Schatzki, and Jonathan Borck, "An Economic Perspective on Building Labeling Policies," March 2013.

[25] For a related discussion focusing on renewable energy technologies, *see* Gillingham, Kenneth and James Sweeney, "Market Failure and the Structure of Externalities", In: Harnessing Renewable Energy, Boaz Moselle, Jorge Padilla, Richard Schmalensee (eds). RFF Press, 2019.

may be impractical because of technical challenges. As a result, certain public policies may target these externalities, such as subsidies for public transportation.

- ***Network Externalities.*** Many energy systems include distribution networks that deliver fuel to individual consumers. For a given technology or fuel type, the availability and reliability of the network used to delivery energy is an important dimension of consumer technology choices. Network externalities potentially affect these technology choices. Several examples from the transportation sector illustrate network externalities.

  Hydrocarbons, electricity and hydrogen are three important transportation fuel technologies that each require distinct refueling infrastructure. At present, the ubiquity of gasoline service stations creates a positive network externality -- the benefits of owning a traditional gasoline powered vehicle increase with a more-developed refueling network. Due to these positive network externalities, the incentives favor owning a gasoline-powered vehicle relative to, for example, an electric vehicle, which depends on a less-developed network of electric charging stations. While a more developed network of charging stations would increase the benefits of owning an electric vehicle, without sufficient numbers of electric vehicles on the road, incentives to invest in charging stations may be inadequate (Li et al, 2017). The resulting "chicken-and-egg" problem may prevent the efficient market developments.

  Another example of a network externality is hydrocarbon standards. Combustion and diesel engines are designed to accept fuels meeting particular fuel specifications. For example, most gasoline-powered vehicles rely on E10, which includes up to 10% ethanol, but cannot operate on higher fractions of ethanol without creating risks of engine damage and voiding of warranties. As a result, these technical engine standards may create a "blend wall" that limits the ability to blend renewable fuels.

Policies aligned with the underlying market failure will address most efficiently and effectively these market failures. For example, network externalities associated with refilling/recharging station networks suggest subsidization of refilling/recharging networks.[26] By contrast, while some other policies would address transportation technologies, they would not necessarily address network externalities. For example, an LCFS subsidizes all forms of transportation irrespective of whether a particular fuel faces a network externality. Moreover, an LCFS subsidizes the variable costs associated with each fuel system, rather than addressing the fixed costs (or standards) associated with refilling/recharging infrastructure. Likewise, congestion externalities suggests some subsidization of public transportation or congestion zone pricing. But, an LCFS would do nothing to address the congestion market failure, since it does not directly address any component of a consumer's travel decisions.

Similarly, policies to address innovation spillovers should target underinvestment in energy R&D. Many market-based policies, such as GHG cap-and-trade systems, RPS and the LCFS, create financial incentives for private firms (entrepreneurs) to increase investment in energy R&D. However, these policies create uniform incentive for GHG reductions regardless of the state of technology development. Thus, depending on the state of technology development, these policies may promote

[26] Of course, this approach is not without challenges. Uncertainty over which new technology will be the most cost-effective creates the risk that the "wrong" technology is subsidized.

substantial technological innovation or simply lead to widespread deployment of pre-existing technologies.

Despite these issues, regulators, such as CARB, have sometimes argued that these types of market-based policies are "technology" policies aimed at encouraging innovation, as distinct from policies aimed at cost-effectively achieving emission reductions. For example, CARB has argued that its LCFS is a "technology" policy aimed at "spurring innovation" in cleaner fuels.[27] However, this claim not only suggests that the gains in energy innovation outweigh the higher costs of these policies, but that their net gains outweigh those of other policies targeting increased energy R&D.

Other policy approaches may better target R&D incentives. For example, the federal government (and some state governments) undertakes substantial direct R&D investment in energy technologies.[28] To the extent policies aim to subsidize innovative technologies, such subsidies can be gradually reduced as market deployment increases. By contrast, most current policies such as California's LCFS have no mechanism to reduce incentives once a technology becomes mature.

B. Interactions between Cap-and-Trade and So-Called Complementary Policies

For state-level climate policies, interactions can occur between individual elements of a state policy or between state policy and federal policy. In either case, interaction between policies has potential implications for the cost-effectiveness of actions taken to reduce GHG emissions, and can have implications for aggregate emission reductions as well.[29] Interactions between policies are most problematic when two conditions occur:[30]

1. When a state policy creates more stringent requirements that overlap with a "broader" state or federal policy ("overlap criteria"); and

[27] For example, "Since 2011, the LCFS has been a cornerstone of California's effort to reduce greenhouse gas (GHGs) emissions and has spurred innovation in low-carbon transportation fuels such as hydrogen, electricity and biodiesel." CARB, "CARB amends Low Carbon Fuel Standard for wider impact," September 27, 2018.; "The LCFS is an important tool in California's efforts to reduce the impacts of climate change by spurring innovation in an array of cleaner fuels." CARB, "Air Resources Board readopts Low Carbon Fuel Standard," July 19, 2017.; A group of UC Davis researchers concluded similarly, noting that "[t]he LCFS will clearly induce technological innovation and investment in new technologies, but perhaps with some delay." Farrell, Alexander E. and Daniel Sperling, "A Low-Carbon Fuel Standard for California Part 1: Technical Analysis", Institute of Transportation Studies, UC Berkeley, May 29, 2007; See also Parson et al., 2018.

[28] Clark, Corine, "Renewable Energy R&D Funding History: A Comparison with Funding for Nuclear Energy, Fossil Energy, Energy Efficiency, and Electric Systems R&D," Congressional Research Service, June 18, 2018. https://fas.org/sgp/crs/misc/RS22858.pdf.

[29] Goulder, Lawrence and Robert Stavins, "Challenges from State-Federal Interactions in U.S. Climate Change Policy," *American Economic Review Papers and Proceedings*, volume 101, number 3, May 2011, pages 253-257. Goulder, Lawrence and Robert Stavins, "Interactions Between State and Federal Climate Change Policies," *The Design and Implementation of U.S. Climate Policy*, eds. Don Fullerton and Catherine Wolfram. Cambridge: National Bureau of Economic Research, 2012.

[30] Goulder and Stavins identify these conditions for interactions between state and federal climate policies. Goulder, Lawrence and Robert Stavins, 2012.

2. The broader federal or state policy provides flexibility to meet requirements through adjustments across sectors or states, i.e. averaging ("flexible policy criteria".)

Not all policies meet these conditions. For example, broader state or federal policies using command and control or price-based instruments have limited interaction with state-level policies. By contrast, policies that trade in quantities (for example, cap-and-trade) and policies that average performance (for example, renewable portfolio standards and fleet vehicle efficiency standards) provide flexibility that creates perverse interactions between policies.

In the context of Oregon's climate policies, the interaction of greatest concern is between the GHG cap-and-trade program and other climate policies that regulate sources covered by the cap-and-trade program.[31] For example, emissions from transportation fuel combustion are regulated by both the cap-and-trade program and by the CFP, which mandates reductions in the GHG-intensity of transportation fuels.

In general, when state-level policies overlap with cap-and-trade, the complementary policies will fail to create any additional emission reductions. With a binding cap-and-trade system in place, aggregate emissions will equal the cap whether or not complementary policies are implemented.[32] While complementary policies may shift emissions among sources or sectors covered by the cap, aggregate emissions will remain *unchanged*. Under these conditions, the complementary policy produces no incremental emission reductions; it simply relocates the emissions (or worse, as we discuss below).

In addition, complementary policies may increase the cost of meeting emission targets when implemented alongside cap-and-trade. If complementary policies require that more costly emission reductions be undertaken, then the shift from lower-cost to higher-cost reduction activities increases the cost of achieving emission targets. If the complementary policy requires reductions that are cost-effective under cap-and-trade then the reductions occur whether or not the complementary policy is implemented; consequently, costs do not rise, but the policy is irrelevant. A complementary policy can shift emission reductions to *lower*-cost emission reduction activities only if it targets non-GHG market failures, such as information problems or behavioral biases regarding household energy use, or targets sectors not covered by the cap-and-trade system.

Complementary policies may depress cap-and-trade allowance prices. Because complementary policies may require emission reductions that are more costly than would be required under cap-and-trade, these requirements displace emission reductions that would otherwise be required by the cap-and-trade system. As a result, the quantity of emission reductions required to meet the cap are reduced, which depresses the price of cap-and-trade allowances. These low prices are problematic for induced technological change.

[31] Schatzki, Todd and Robert N. Stavins, "Implications of Policy Interactions for California's Climate Policy", Regulatory Policy Program, Mossavar-Rahmani Center for Business and Government, Harvard Kennedy School, August 27, 2012.

[32] If the cap is not binding, then complementary policies can reduce emissions. For example, for several quarters in recent years, California's cap was not binding because the auction reserve prices limited allowance allocations. However, the low demand for allowances was primarily a result of the substantial emission reductions achieved by the complementary policies. Thus, the actual reduction in emissions achieved by the complementary policies was likely limited to the allowances unsold at auction.

In fact, several of Oregon's existing climate policies regulate emissions that would be covered by a GHG cap-and-trade system. Oregon's RPS regulates the generation of electricity, which is largely produced by large stationary sources and electricity imports, both of which would be covered by the proposed GHG cap-and-trade system.[33] Because the RPS regulates sources covered by the GHG cap-and-trade proposals, once the GHG cap-and-trade system is in place, the RPS will not achieve any incremental emission reductions. Worse yet, because the RPS would raise costs if it required the adoption of renewable energy technologies that reduced GHG emissions at a higher cost than other options.

Interactions between the Oregon's CFP and its GHG cap-and-trade system would be more complicated because portions of the transportation fuel lifecycle would be covered by the GHG cap-and-trade system, while other portions of the fuel lifecycle would be outside the cap. In section III, we provide a more detailed description of these interactions and analyze the actual change in emissions from the California's LCFS.

III. EXPERIENCE WITH INTERACTIONS BETWEEN CALIFORNIA'S LCFS AND ITS GHG CAP-AND-TRADE SYSTEM

California's AB 32 Scoping Plan includes multiple elements aimed at achieving climate targets specified in California's Global Warming Solutions Act of 2008. Implementation of these policies has been staggered, but generally started several years in advance of policies in Oregon. Because many of these policies have been in place for multiple years, California's experience can provide a valuable lens into the interactions among policies, which other states, such as Oregon, can expect from their suite of climate policies.

We focus on interactions between California's LCFS and its GHG cap-and-trade system. The LCFS has been in place since 2013. While the GHG cap-and-trade system has been in place since 2013 as well, the cap initially covered only large stationary point sources, and was not expanded to cover fuels until 2015. As a result, California's experience allows market impacts to be analyzed before and after the interactions between the programs first occurred.

A. Changes in Costs Due to the LCFS

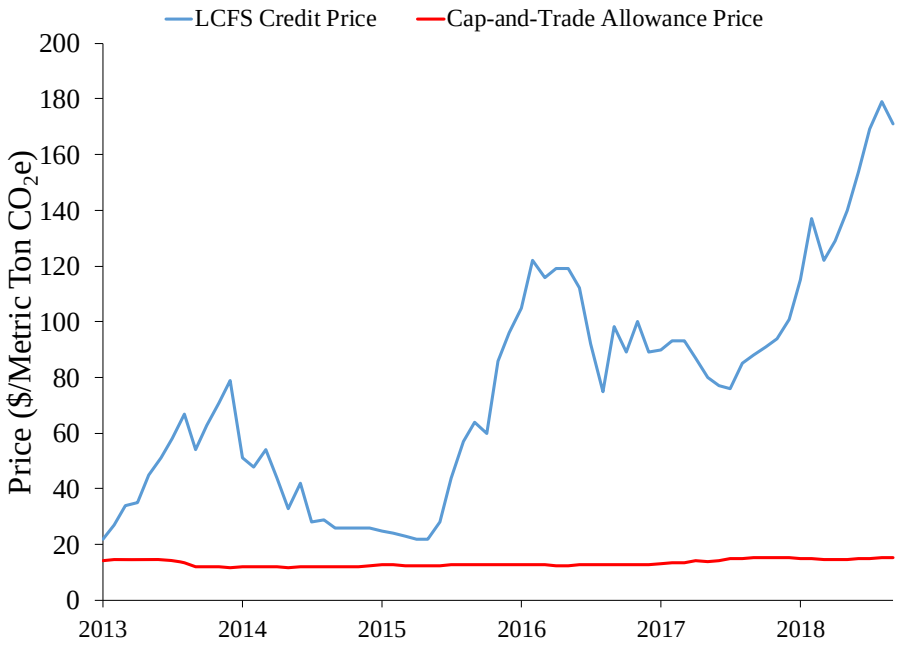
Both the LCFS and GHG cap-and-trade system create price signals that reflect the marginal costs of achieving emission reductions. The LCFS has its own trading program, and LCFS credit prices reflect the (marginal) cost of reducing CO_2 emissions by switching from high-carbon fuels to low-carbon fuels given their differences in carbon intensity. Likewise, GHG allowance prices reflect the tradeoff between taking actions to reduce emissions and the market value of those emission reductions.

There is a large difference between GHG cap-and-trade allowance prices and LCFS credit prices in California. Figure 1 compares LCFS credit prices and GHG allowance prices from 2013 to present. For the program's first few years, LCFS credit prices were often at relatively low levels, comparable to GHG cap-and-trade allowance prices. However, after legal uncertainty about the program

[33] SB 1507, Section 13.(2).

was resolved and the carbon-intensity standard was reduced, prices increased appreciably, and have since remained at levels above $80 per $MTCO_2e$. In recent months, prices have been closer to $180 per $MTCO_2e$. By contrast, GHG cap-and-trade allowances prices are approximately $14 per $MTCO_2e$ and, for a period, were at the administratively set auction reserve price (i.e. the price floor).

Figure 1. California's LCFS Credit Prices vs. GHG Cap-and-Trade Allowance Prices



Notes: [1] The monthly cap-and-trade allowance price is calculated as the monthly average of the California carbon allowance future contract price from SNL. [2] The monthly LCFS credit price is equal to the CARB monthly average credit price.
Sources: [1] CARB. [2] SNL Financial.

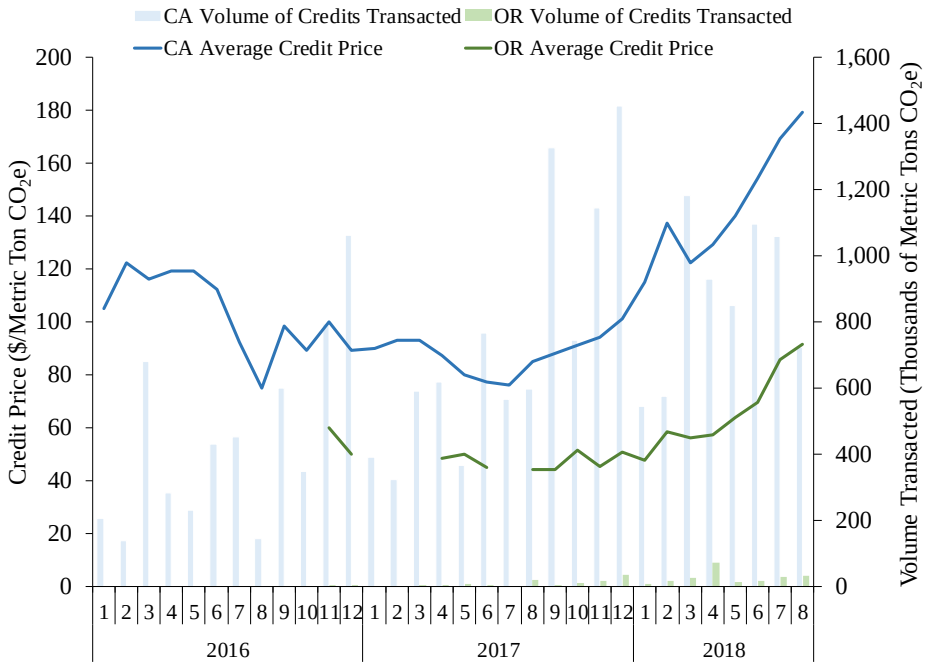
The large difference between LCFS credit prices and GHG cap-and-trade allowance prices in California indicates that, at the margin, the emission reductions being achieved by the LCFS are substantially more costly than reductions achieved through the GHG cap-and-trade system. For example, in August 2018, LCFS credit prices averaged $179/MT, while GHG cap-and-trade allowance prices averaged just $15/MT, indicating that marginal CO_2e abatement costs are more than eleven times greater in the LCFS program than in the GHG cap-and-trade system.

The interactions between the two programs place downward pressure on cap-and-trade allowance prices. In effect, the more costly emission reductions required by the LCFS displace less costly emission reductions that would otherwise be achieved by the cap-and-trade program. But, by reducing emissions from sources that are covered by the cap, the LCFS requirement effectively reduces the GHG cap stringency, thus reducing allowance prices.

Credit prices for Oregon's CFP will differ from prices in California's LCFS. **Figure 2** illustrates this, comparing credit prices and volume transacted from California's LCFS and Oregon's CFP. Differences in credit prices reflect a number of factors. One important factor is the difference in the stringency between Oregon and California. With a more stringent carbon-intensity standard, the use of a low-carbon fuel generates fewer credits, thus raising the cost of generating credits. For example, in 2017,

Oregon's CFP required reductions in carbon intensity of 0.5% (relative to a 2015 baseline), while California's LCFS required reductions of 3.5% (relative to a 2010 baseline). In 2018, required reductions are 1% and 5% for Oregon and California, respectively. In addition, estimated carbon-intensity for individual renewable fuels tends to be lower in Oregon than in California, so substitutions generate more credits in Oregon than in California.[34] Thus, use of a given type of low-carbon fuel will tend to create more credits in Oregon than California, thus lowering the cost.

Figure 2. Comparison of Credit Prices and Volume Transacted, California's LCFS and Oregon's CFP

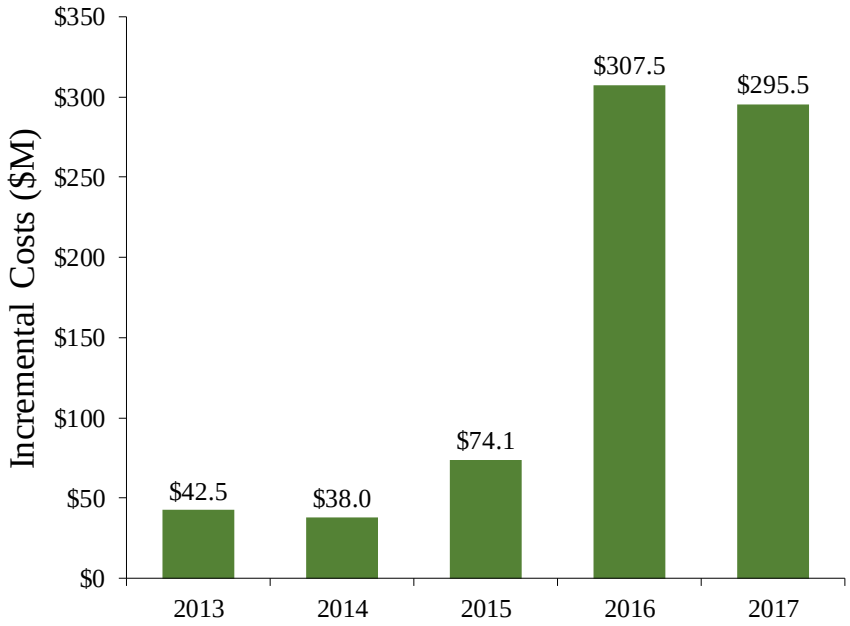


Sources: [1] CARB. [2] Oregon Department of Environmental Quality.

The incremental costs of achieving emission reductions through the LCFS, rather than the GHG cap-and-trade system, have been substantial. Figures 3 provides an estimate of the incremental costs of the LCFS relative to the GHG cap-and-trade system. The observed emission reductions are relatively small, less than 4% of the total annual emissions from the California transportation sector and less than 2% of overall state GHG emissions. In total, estimated incremental LCFS costs were around $300 million in both 2016 and 2017, and over $750 million over the 5-year period from 2013 to 2017. Extrapolating for 2018, estimated costs could exceed $400 million.

[34] A key source of these differences is that California's carbon-intensity estimates account for indirect land use change, whereas estimates in Oregon to not.

Figure 3. Annual Incremental Costs, California's LCFS



Note: We distinguish in our calculations between expenditures by reducing entities and economic cost of emission reductions. Expenditures associated with emission reductions are simply (annual emission reductions [MT]) × (average annual credit price [$/MT]), where the average annual credit price represents the average of the 12 monthly CARB reported average credit prices. Costs of emission reductions can be represented by the area under an emissions reduction supply curve between the origin and market clearing price, here represented by the average annual credit price. If we make the simplifying assumption of a linear supply curve, costs will equal half of the expenditures, since the area of a triangle is one half the area of a rectangle with same base and height.

Source: CARB.

B. Changes in Emissions Due to the LCFS

While the adoption of a GHG cap-and-trade system will increase the certainty of environmental outcomes, interactions between the GHG cap-and-trade system and complementary programs have consequences for the incremental impact of these complementary measures. With the LCFS, these interactions are complicated because LCFS compliance depends on the lifecycle emissions of each type of transportation fuel, not simply emissions from vehicle fuel combustion. While all of the vehicle emissions are covered by the cap-and-trade program, the portion of a fuel's productions emissions that are regulated by the cap-and-trade system varies across fuel types. The change in emissions due to the LCFS will reflect these differences in emissions outside the cap, rather than activity that is covered by the GHG cap-and-trade system.

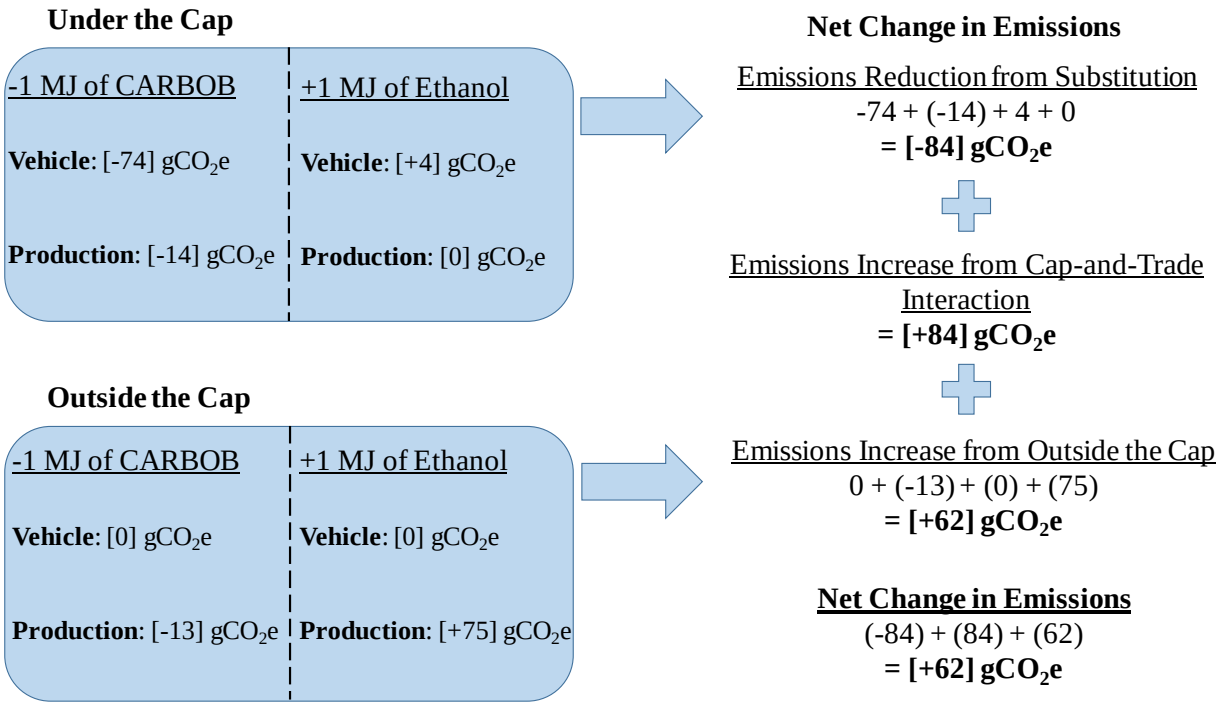
Figure 4 illustrates the impact of the substitution of a quantity of ethanol for an equal quantity of gasoline (CARBOB). The example assumes that the gasoline is refined in California, while ethanol is refined in an out-of-state (non-California) refinery. The gasoline carbon-intensity is 101 gCO_2e, including vehicle emissions (74 gCO_2e), in-state production emissions (14 gCO_2e), and out-of-state production emissions (13 gCO_2e). By contrast, ethanol carbon-intensity is lower, 79 gCO_2e, reflecting only in-state emissions (4 gCO_2e), and out-of-state production emissions (75 gCO_2e). Under the LCFS,

ethanol produces very little net vehicle emissions because carbon sequestered in the process of growing corn to produce ethanol offsets tailpipe emissions.

Absent the GHG cap-and-trade system, switching from gasoline to ethanol results in a carbon reduction of 22 gCO_2e (that is, 101 gCO_2e - 79 gCO_2e). However, with the GHG cap-and-trade system in place, the impact needs to account for the interaction of the switch to ethanol proscribed by the LCFS program with the GHG cap-and-trade system. Accounting for this impact requires a separate analysis of changes in emissions from sources covered by the cap-and-trade system and those outside the cap.

Start with emissions under the cap. For gasoline, 88 gCO_2e of lifecycle emissions are covered by the cap (74 gCO_2e of vehicle emissions + 14 gCO_2e from in-state refining), whereas only 4 gCO_2e of ethanol lifecycle emissions would be covered by the cap. Thus, substituting ethanol for gasoline reduces GHG emissions under the cap by 84 gCO_2e (that is, 88 gCO_2e - 4 gCO_2e). However, because total emissions under the cap is fixed, there is actually no change in emissions under the cap; instead, other sources under the cap will increase their emissions by 84 gCO_2e given the slack in emission created by substitution.

Figure 4. Illustration of Change in Emissions due to Substituting Ethanol for Gasoline under the LCFS



Note: [1] We assume that the cap binds; thus any reduction in emissions covered by the cap will be replaced by emissions from another sector or source covered by the cap. This accounts for the increase in emissions due to interaction with the GHG cap-and-trade system. [2] Assumptions regarding what is under the cap and outside the cap are made for illustrative purposes.
Source: [1] CARB.

Outside the cap, production of 1 MJ of ethanol increases GHG emissions by 75 gCO_2e, while production of 1 MJ less of gasoline decreases GHG emissions by 13 gCO_2e. As a result, substitution of ethanol for gasoline *increases* emissions outside the cap by 62 gCO_2e (that is, 75 gCO_2e - 13 gCO_2e).

Thus, while the substitution produces no change in emissions under the cap, emissions outside the cap increase by 62 gCO_2e.

The change in emissions from the substitution of low-carbon fuels for traditional fossil fuels -- gasoline and diesel -- depends on the specific substitution made and the difference in lifecycle emissions that are outside California's GHG cap-and-trade system. **Table 1** illustrates these differences for several types of substitutions. For each fuel, we break down lifecycle emissions into production and vehicle emissions under the cap, and emissions outside the cap. Without cap-and-trade, the impact of any given fuel substitution reflects the change in "Total" carbon intensity. However, with cap-and-trade, the impact reflects the change in "Outside the Cap" carbon intensity.

As shown in **Table 1**, due to interactions with the cap-and-trade program, substitutions can either increase or decrease GHG emissions. For example, while substitution of ethanol for gasoline *increases* emissions by 62 gCO_2e (that is, 75 gCO_2e - 13 gCO_2e), substitution of electricity (EVs) for gasoline *reduces* emissions by 66 gCO_2e (that is, -53 gCO_2e - 13 gCO_2e). Likewise, substitution of biodiesel for diesel fuel (ULSD) increases emissions by 8 gCO_2e (that is, 23 gCO_2e - 15 gCO_2e), while a substitution from gasoline to compressed natural gas (CNG) from landfills decreases emissions by 54 gCO_2e (that is, -39 gCO_2e - 15 gCO_2e).

Table 1. Lifecycle Emissions of Gasoline, Diesel, and Substitutes (gCO_2e per MJ)

	Lifecycle Component	CARBOB	Corn Ethanol	Electricity
Under the Cap	Production	14	-	84
	Vehicle Use	74	4	-
Outside the Cap	Production	13	75	21
	Vehicle Use	-	-	-74
Total		**101**	**79**	**31**
Outside the Cap		**13**	**75**	**-53**

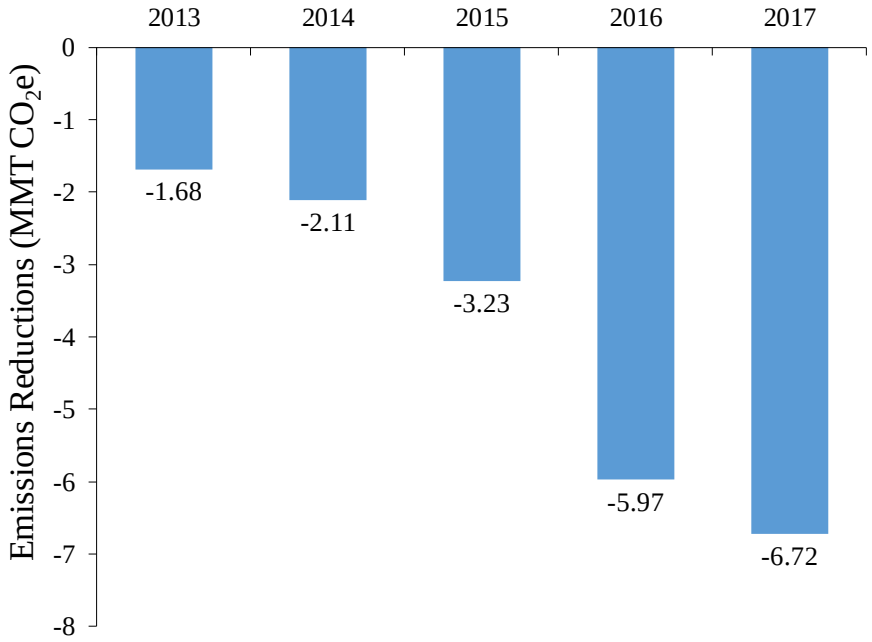
	Lifecycle Component	ULSD	Cooking Oil Biodiesel	Landfill Gas to CNG
Under the Cap	Production	13	-	-
	Vehicle Use	75	3	64
Outside the Cap	Production	15	23	-39
	Vehicle Use	-	-	-
Total		**103**	**26**	**25**
Outside the Cap		**15**	**23**	**-39**

Source: CARB.

Because individual substitution between traditional fossil fuels and renewable fuels could increase or decrease emissions, the aggregate impact of the LCFS will depend on the mix of substitutions used by the market to comply with the LCFS. We estimate the aggregate impact of the LCFS in California from 2012 to present. The analysis reflects the particular mix of fuel used to comply with the LCFS, the carbon-intensity of each type of fuel, and the portion of each type of fuel produced within California (and thus subject to the cap). Further details on the analysis are provided in the appendix.

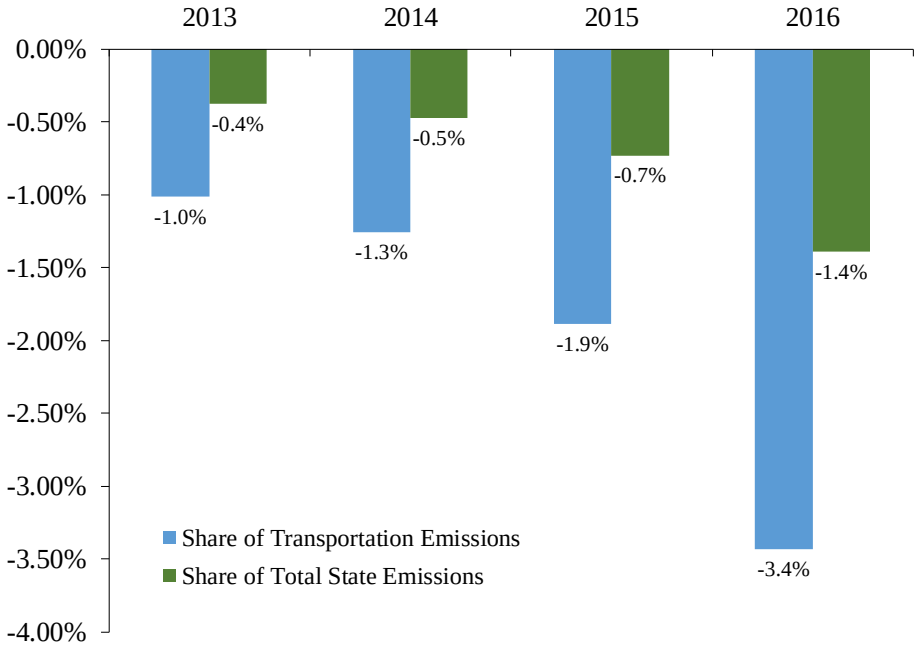
In aggregate, the direct impact of the LCFS leads to modest reductions in emissions. **Figure 6** shows the direct reduction in emissions from the LCFS, before accounting for the interaction with the GHG cap-and-trade system. These estimates reflect reductions achieved by changing the mix of fuels consumed, but do not reflect aggregate reductions. In fact, total GHG emissions from transportation continue to grow under the LCFS, as policies have failed to stem the growth in vehicle miles travelled. **Figure 7** shows these changes relative to California's total emissions and total transportation emissions. These changes are modest. For example, the direct change in emissions from the LCFS in 2017 was about 3.5% of total emissions, although this change required credit prices of more than $80 per $MTCO_2e$.

Figure 6. Direct Change in Carbon Emissions by Year from California's LCFS



Source: California Greenhouse Gas Emission Inventory.

Figure 7. LCFS Direct Emission Change Relative to California Aggregate Emissions



Source: California Greenhouse Gas Emission Inventory.

On average, fuel substitution required to comply with the LCFS has led to an increase in emissions from fuel production outside California not covered by the state's GHG cap-and-trade system. Thus, in aggregate, the LCFS has *increased* total GHG emissions. Figure 8 shows estimated changes in total GHG emissions from the LCFS over the period 2012 to 2015. Starting in 2015 when the GHG cap-and-trade system was expanded to include fuels, Figure 8 shows the actual change in emissions given the interaction between the LCFS and the GHG cap-and-trade system (the solid blue line) and the emissions reductions the LCFS would have achieved absent the GHG cap-and-trade system interaction (the dashed blue line).[35] **Table 2** provides detailed estimates of the changes in emissions associated with the increased (or decreased) use of each type of renewable fuel.[36]

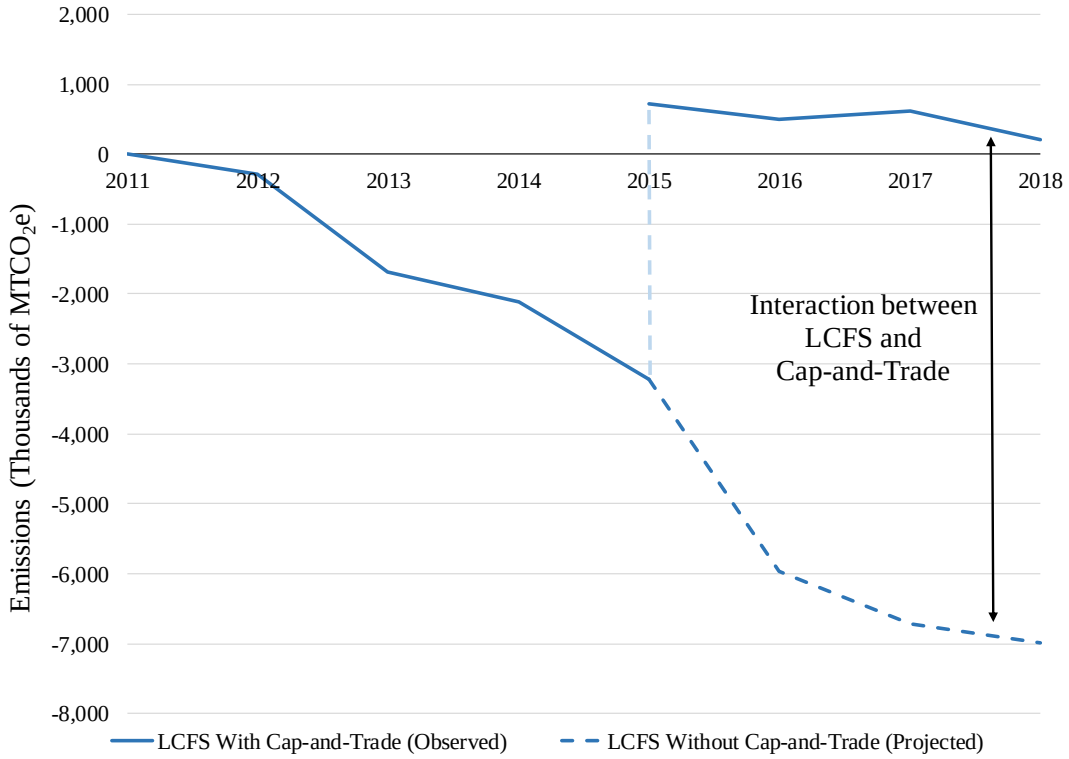
Prior to 2015 the LCFS achieved reductions ranging from 287,406 to 2,105,268 $MTCO_2e$ annually, as low-GHG fuels were substituted for fossil fuels. However, since the GHG cap-and-trade system was expanded to include transport fuels in 2015, emissions outside California (i.e., outside the

[35] We cannot measure changes in emissions relative to 2010, the year prior to the adoption of the LCFS, because CARB does not provide detailed data on fuel consumption in 2010. Thus, we measure changes in emissions relative to actual emissions in 2011, the first year the program was in effect. Because the LCFS required only a 0.25% reduction in carbon intensity in 2011, a relatively weak requirement, emissions in 2011 provides reasonable benchmark for evaluating program impacts.

[36] Over time, the mix of non-traditional fuels has shifted, resulting in lower consumption for some renewable fuels with comparatively higher carbon-intensity. For example, LCFS incentives have led to "fuel shuffling", with reduced consumption of high-carbon-intensity ethanol (> 75 gCO_2e/MJ) and increased consumption of lower-carbon-intensity ethanol.

state's GHG cap-and-trade system) have increased in each year due to the interaction between the two programs. In 2015, emissions increased by 720,777 $MTCO_2e$, while in 2017, emissions increased by over 600,000 $MTCO_2e$.

Figure 8. Aggregate Change in Emissions from California's LCFS



Note: [1] For 2012-2017, the estimated change in emissions assumes a counterfactual with renewable fuel use equal to 2011 levels. For 2018 (Q1), the estimate change emissions assumes a counterfactual of one fourth of renewable fuel use from 2011. The analysis does not assume any adjustment to renewable fuel use from 2011 levels that might occur under a GHG cap-and-trade system. Estimates also do not account for emissions from in-state production that might be covered by a GHG cap-and-trade system.

Table 2. California Overall Net Change in GHG Emissions ($MTCO_2e$)

	LCFS Without Cap-And-Trade - No Leakage				LCFS With Cap-and-Trade - Leakage			
	2011	2012	2013	2014	2015	2016	2017	2018 (Q1)
Bio-CNG	0	0	-61,040	-157,016	-43,219	-63,121	-91,938	-25,255
Bio-LNG	0	-196	-29,732	-92,082	-28,857	-31,467	-19,961	-3,453
Fossil CNG	0	-18,702	-31,100	-84,899	-11,726	-691	2,118	314
Fossil LNG	0	-15,693	-18,134	1,018	16,252	20,079	21,051	5,292
Hydrogen	0	0	0	0	-3	-3	-246	-104
Electricity - Onroad	0	-6,244	-25,736	-64,663	-11,552	-19,893	-28,823	-9,373
Electricity - Offroad	0	0	0	0	0	-35,068	-40,236	-9,997
Ethanol <65	0	-114,225	-189,306	-386,832	528,455	787,250	360,397	58,593
Ethanol 65-75	0	-94,366	-212,113	14,938	-119,866	4,538,552	5,578,020	1,417,485
Ethanol >75	0	90,885	199,980	156,361	-553,054	-5,601,373	-6,338,801	-1,683,664
Biodiesel	0	-63,272	-465,040	-578,330	78,772	15,568	197,440	37,252
Renewable Diesel	0	-65,593	-851,453	-913,762	865,575	881,223	972,919	262,765
Total	0	-287,406	-1,683,674	-2,105,268	720,777	491,055	611,939	49,856

Note: [1] For 2012-2017, the estimated change in emissions assumes a counterfactual with renewable fuel use equal to 2011 levels. For 2018 (Q1), the estimate change emissions assumes a counterfactual of one fourth of renewable fuel use from 2011. The analysis does not assume any adjustment to renewable fuel use from 2011 levels that might occur under a GHG cap-and-trade system. Estimates also do not account for emissions from in-state production that might be covered by cap-and-trade.

The increase in emissions reflects several factors. First, nearly all gasoline and diesel used in California is produced in refineries located in California. Thus, reductions in emissions from this refining activity are offset by increases in emissions from other activities. Second, fuel production and refining emissions tend to be larger for low-carbon fuels than for traditional fossil fuels. Thus, the substitution to low-carbon fuels produced out of state often leads to large emission increases.

C. Technology Innovation

CARB has justified the high cost of the LCFS by claiming that it is a "technology" policy aimed at "spurring innovation." A complete analysis of incremental innovation and R&D fostered by the LCFS is beyond the scope of this paper. Such an analysis would need to determine whether technology innovation outcomes (e.g., patents, R&D spending) have increased with LCFS credits prices (after controlling for all other factors affecting investments in innovation). Several points can be made about LCFS outcomes.

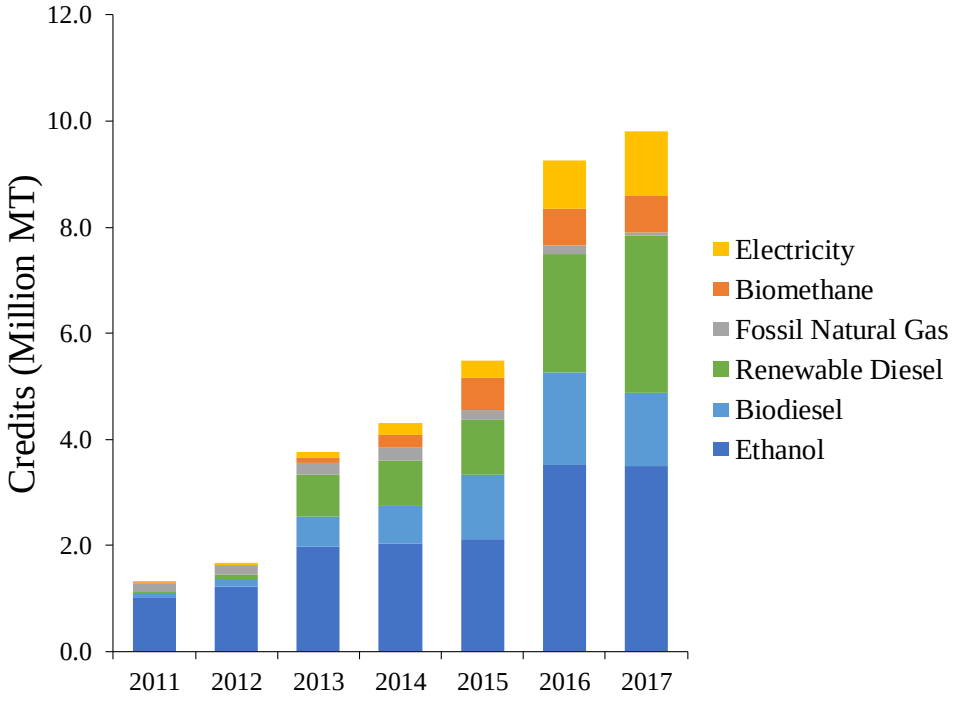
First, **the costs of the LCFS have been large relative to the all government spending on clean energy R&D**. As discussed above, the LCFS has imposed costs of over $700 million in both 2016 and 2017. This spending in on par with *all* Federal spending on renewable energy R&D. From 2009 to 2018, Federal research and development spending on renewable energy averaged $937 million annually.[37] For example, one important energy R&D program is the Advanced Research Projects Agency-Energy's (ARPA-E), established in 2007 to help advance high-potential, high-impact energy technologies that are

[37] Clark, Corine, "Renewable Energy R&D Funding History: A Comparison with Funding for Nuclear Energy, Fossil Energy, Energy Efficiency, and Electric Systems R&D," Congressional Research Service, June 18, 2018. https://fas.org/sgp/crs/misc/RS22858.pdf.

too early for private sector investment. In 2016, the budget for ARPA-E was $294 million, less than half of the LCFS's incremental cost in the same year.

Second, **compliance with the LCFS has been achieved through fuel technologies which have been commercially available prior to the LCFS, but have generally been too costly compared with alternatives without the LCFS subsidy. Figure 9 and 10** illustrate the mix of fuels used to comply with the LCFS in terms of number of credits (Figure 9) and percentage of credits (Figure 10). To date, LCFS compliance has been achieved primarily through ethanol, biodiesel, and renewable diesel, accounting for over 80% of credits each year. These fuels were commercially available prior to the LCFS. **Thus, to date, LCFS compliance has largely been achieved through the deployment of existing, rather than innovative technology.** The LCFS program has expanded the market for these fuels, potentially providing producers of these fuels or suppliers of the underlying feedstock with windfalls (economic rents).[38]

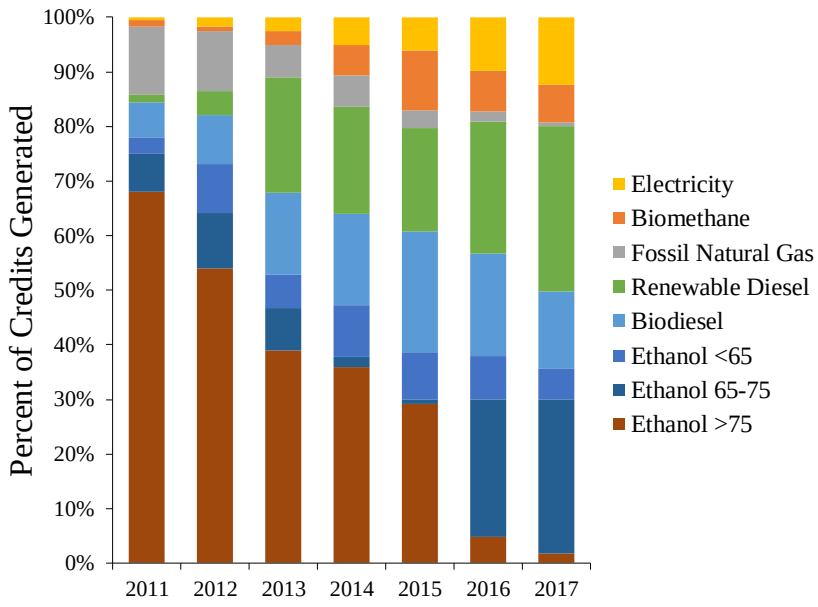
Figure 9. Mix of Fuels Used to Comply with the LCFS, MMT Credits



Source: CARB.

[38] The increase in LCFS credit prices increases the value of the underlying feedstock and means of production. In some cases, some production may be held by companies with proprietary technologies although, as we describe below, the fundamental chemical processes used in current renewable production are fairly well understood scientifically.

Figure 10. Mix of Fuels Used to Comply with the LCFS, Percent of Total Credits



Source: CARB.

Ethanol has been the largest source of credits since the inception of the LCFS, while biodiesel has been the third largest source of credits. Both ethanol and biodiesel have been widely produced in the United States for decades, in part due to subsidies from the Federal Renewable Fuel Standard. Ethanol use to comply with the LCFS also includes sugar cane ethanol produced in Brazil, where the sugar cane industry was well-established prior to the LCFS, having produced significant quantities of fuel for decades.

Some ethanol and biodiesel credits have also been created through "fuel shuffling," which occurs when low-carbon intensity ethanol is directed to California (because of the higher price), while high-carbon intensity ethanol is directed to other parts of the county. Fuel shuffling creates "paper" emission reductions in California without actually creating any change in the ethanol fuel stock.

The second largest source of credits is renewable diesel. Renewable diesel is a "drop in" replacement for diesel that does not require any blending. Use of renewable diesel in California has grown in recent years as credit prices have increased. But, renewable diesel was in production long before the LCFS was established. California's renewable diesel is supplied primarily by two producers, Neste (Singapore) and Diamond Green Diesel (Louisiana).[39] Neste has four plants, all of which have been operational since 2011. Thus, renewable diesel is not a novel technology.

[39] Neste produces renewable diesel at facilities in Finland, Rotterdam and Singapore in facilities that were operational in 2007/2009, 2010 and 2011, respectively.

The share of credits from electric powered vehicles (EVs) has grown in each year. In 2017, EVs accounted for over 10% of credits. Electric vehicles have been growing slowly in share, and face significant technical hurdles to broad commercial acceptance (including battery life and cost, and necessary recharging infrastructure). EVs also benefit from multiple state and federal subsidies, including federal tax deductions, rebates and incentives and requirements related to EV charging stations. The extent to which the LCFS materially increases these incentives is unclear.

D. Implications for Oregon

California's experience with its LCFS has important implications for Oregon.

First, the GHG cap-and-trade system will achieve emission reductions at a lower cost than other (complementary) policies that Oregon has already adopted to address climate change and other environmental impacts. At present, credit prices for the CFP program are approximately $80 per $MTCO_2e$, which is significantly above likely GHG cap-and-trade allowance prices. At present, emission reduction costs from the RPS appear comparable (but subject to uncertainty due to limited information).[40] These costs may rise as the stringency of Oregon's CFP standard increases.

Second, Oregon should expect the adoption of a GHG cap-and-trade system will have consequences for the effectiveness of the CFP in producing incremental emission reductions. Like California's LCFS, the CFP will lead to no (or little) emission reductions, and potentially even increase emissions as has been the experience in California. As with California, actual emission outcomes will depend on the particular fuel substitutions used to comply with the CFP. However, differences between the state's programs and markets will lead to differences in emission outcomes. While nearly all of California's fossil fuel refining occurs in-state and is thus under the cap, none of Oregon's fuel is refined in-state, and so all reductions in refining emissions are outside the cap. All else equal, this will increase the emission reductions achieved by the CFP (compared to California's LCFS) because reduced gasoline and diesel consumption will reduce out-of-state refinery emissions. In addition, details of the policies, notably the carbon-intensities, differ between the states.

Oregon's CFP has yet to have a meaningful impact on renewable fuel use, thus making it premature to evaluate potential impacts of the overlap with a GHG cap-and-trade system. In 2017, renewable fuel use increased by only 33 Million MJ compared with 20,360 Million MJ consumed in 2016, an increase of less than 0.2%.[41] Changes in the mix of non-traditional fuels led to reductions in emissions (as measured by Oregon) of 17,751 $MTCO_2e$. This is a very small change in emissions, less than 0.1% of total transportation in emissions in 2016 (24.2 million $MTCO_2e$). This change in fuel mix included a decrease in ethanol consumption of 713 Million MJ and an increase in consumption of other

[40] In 2017, the average reported cost of bundled RECs was $29 per REC for Portland General Electric. (Reported cost per REC for unbundled RECs was substantially lower.) Assuming that the REC displaces natural gas-fired generation with a heat rate of 8 MMBtu per MWh, this results in a cost of $68 per $MTCO_2e$. PacifiCorp's costs were not publicly reported. Portland General Electric, UM 1958 - PGE 2017 Renewable Portfolio Standard Compliance Report, June 1, 2018. https://www.oregon.gov/energy/energy-oregon/Documents/2017-PGE-Compliance-Report.pdf

[41] As with our analysis of California's LCFS, we estimate changes in emissions relative to 2016, the first year of the CFP, because Oregon does not report detailed information on non-traditional fuel use in 2015.

fuels (including biodiesel, renewable diesel and forms of CNG) of 746 Million MJ. This shift in the composition of non-traditional fuels may be the result of CFP incentives, or it may be the result of other market factors.

IV. NEXT STEPS FOR OREGON CLIMATE POLICY

As Oregon contemplates the adoption of a GHG cap-and-trade system, it has several options for its suite of climate policies. **One approach maintains all policies, as currently proposed, with a new GHG cap-and-trade system plus existing complementary policies. Our analysis shows that, due to interactions among overlapping climate policies, retaining many of the existing complementary policies could be very costly without achieving any incremental environmental benefits.**

A second option would develop a GHG cap-and-trade system of sufficient stringency to achieve targeted emissions or allow prices to rise to the social cost of carbon, and end complementary policies that do not produce incremental economic benefits by addressing market failure unrelated to the GHG emission externality or target sources not covered by the cap. This approach could begin by undertaking a thorough assessment of the likely interactions among overlapping climate policies and the extent to which policies address market failures unrelated to GHG emissions or target sources not covered by the GHG cap-and-trade system. **Economic analysis unambiguously shows that this second option, relying largely (if not solely) on GHG emission pricing, is the most cost-effective approach to achieving emission targets.**

However, political realities may not support the immediate adoption of climate policies relying on high (efficient) carbon pricing. Thus, when carbon pricing is adopted, the resulting price levels are often well below the social cost of carbon and price levels that would be needed to achieve the emission targets sought by legislators. Instead, states (and other local governments) often take a "belts and suspenders" approach that pursues reduction through a "suite" of policies targeting many of the activities that lead to GHG emissions (i.e., the first option described above). This approach is often more politically expedient, as it offers the possibility of addressing climate change while hiding the costs. However, this approach may actually be less effective at achieving desired emission reductions. And, as state climate policies become increasing stringent, the costs associated with inefficient complementary policies will become larger.

A third option takes a hybrid approach by enacting carbon pricing that begins at relatively low levels that are politically feasible even if they are not optimal, gradually raising prices over time. As carbon prices are increased over time, reliance on the more costly complementary policies to achieve targeted emission reductions can be diminished, and in some cases eliminated. Those complementary policies that address other legitimate market failures (e.g., energy R&D, energy efficiency programs, etc.) or sources outside the cap would be retained. Thus, the burden of achieving GHG emission reductions can be shifted from complementary policies to GHG pricing.

Along with acclimating politicians and citizens to carbon pricing, decisions about the use of the revenues from GHG pricing may also have an important role in affecting political willingness to adopt ambitious carbon-pricing policies. Much of the opposition stems from the perception that carbon pricing constitutes a new tax. Thus, while decisions about revenue use has important impacts on the costs of achieving climate goals, these decisions may also lower political barriers to GHG cap-and-trade systems if revenues uses can defuse arguments grounded in opposition to new taxes. In particular, making a GHG

cap-and-trade system revenue neutral may address concerns that the policy is a new tax. Making the use of revenues transparent may also reduce political opposition.

In the interim, there are several important considerations for decisions regarding complementary policies. First, policies that meet the criteria identified above, such as addressing market-failures unrelated to the GHG emission externality or targeting emission sources outside the emission cap, will continue to provide economic, and potentially environmental, benefits. Second, complementary policies that achieve emission reductions at a lower cost than alternatives will be more economically efficient. Finally, complementary policies that include mechanisms to reduce their stringency over time may better allow carbon pricing to achieve a growing share of emission reductions. In this regard, subsidies are problematic, as they create a constituency that inevitably lobbies for their preservation.

Technical Appendix

Our analysis assesses the change in lifecycle emissions achieved by the LCFS. In particular, for fuels covered by both the LCFS and GHG cap-and-trade programs—CARBOB gasoline, ULSD diesel, ethanol, electricity, biodiesel, and CNG—we model the change in emissions due to switching from either CARBOB or ULSD diesel to a lower carbon intensity fuel substitute. The model is similar in structure to that used in Schatzki and Stavins (2012), with three key differences: (1) the model relies on actual fuel usage reported by ARB instead of forecasted scenarios; (2) the model considers all alternative fuel types instead of just a few different forms of low-carbon fuels; and (3) the model accounts for differences in the location of production.

The model considers annual emission changes due to the LCFS in two distinct periods of the program. The first period occurs from 2011 through 2014, prior to the GHG cap-and-trade system expanding to cover fuels. The second period occurs from 2015 through the first quarter of 2018 (the latest period for which data exists), during which time the GHG cap-and-trade system and the LCFS both regulated transport fuels.

In the 2011-2014 period, the analysis calculates the emissions changes due to fuel switching caused by the LCFS alone. Specifically, changes in emissions are calculated for each alternative fuel type as the difference between the carbon intensity of the alternative fuel being switched to and the carbon intensity of the fuel being switched away from (CARBOB or ULSD diesel). Because the LCFS considers lifecycle carbon intensity (i.e. throughout the entire production process and point of use), the change in emissions from fuel switching reflects lifecycle emissions changes.

In the 2015-2018 (Q1) period, the analysis calculates the emissions due both to fuel switching as just described *and* the interaction between the LCFS and GHG cap-and-trade system. In estimating the impact of this interaction, changes in emissions depend upon the component of each fuel's production process that occurs within California and outside California. The GHG cap-and-trade system covers emissions from combustion, in-state petroleum refining, and in-state renewable fuel production. Therefore, the model incorporates the following assumptions:

- Crude production occurs in and outside California, reflecting data from CARB, while CARBOB gasoline and ULSD diesel refining occur solely in California.[42]
- Production of ethanol occurs both in and outside California, reflecting data from the Energy Information Administration (EIA).[43]
- Production of renewable diesel occurs outside California, reflecting market information and communications with market participants.

[42] CARB, "Detailed CA-GREET Pathway for California Reformulated Gasoline Blendstock for Oxygenate Blending (CARBOB) from Average Crude Refined in California," December 15, 2014, available at https://www.arb.ca.gov/fuels/lcfs/121514carbob.pdf; CARB, "Detailed California-Modified GREET Pathway for Ultra Low Sulfur Diesel (ULSD) from Average Crude Refined in California," December 15, 2015, available at https://www.arb.ca.gov/fuels/lcfs/121514ulsd.pdf.

[43] For ethanol production see EIA, "California State Energy Profile," October 19, 2017, available at https://www.eia.gov/state/print.php?sid=CA; for biodiesel production see EIA, "Monthly Biodiesel Report," September 2018, available at https://www.eia.gov/biofuels/biodiesel/production/biodiesel.pdf.

- Production of electricity used to power electric vehicles is covered entirely by the GHG cap-and-trade system, consistent with the program design.
- Production of remaining fuels (bio-CNG, bio-LNG, fossil CNG, fossil LNG, and hydrogen) occur within California.

The lifecycle emissions change (in grams of CO_2e per MJ) implied by the model are captured in the table below:

California Change in GHG (gCO_2e/MJ)

	LCFS Without Cap-And-Trade - No Leakage				LCFS With Cap-and-Trade - Leakage			
	2011	2012	2013	2014	2015	2016	2017	2018 (Q1)
Bio-CNG	-	-	-81	-76	-8	-8	-8	-8
Bio-LNG	-78	-78	-72	-59	-8	-8	-8	-8
Fossil CNG	-27	-27	-27	-26	-8	-8	-8	-8
Fossil LNG	-19	-19	-19	-19	-8	-8	-8	-8
Hydrogen	-	-	-	-	-8	-8	-8	-8
Electricity - Onroad	-57	-57	-69	-69	-8	-8	-8	-8
Electricity - Offroad	-	-	-	-	-	-8	-8	-8
Ethanol <65	-41	-48	-41	-40	45	44	39	43
Ethanol 65-75	-30	-30	-30	-28	51	53	53	53
Ethanol >75	-12	-13	-16	-16	64	60	59	59
Biodiesel	-61	-67	-78	-85	5	1	10	9
Renewable Diesel	-81	-72	-57	-63	41	27	22	21

Source: [1] Carbon intensities are calculated based on fuel volumes and credits from ARB's LCFS Quarterly Data spreadsheet as of 7/3/2018.

Annual Direct Emission Reductions and Incremental Costs, California's LCFS

Year	Observed Emission Reductions (MT) [A]	Average Credit Price ($ / MT) [B]	Estimated Incremental Cost of Reducing Emissions [C] = ([A] * [B]) / 2
2013	1,683,674	50.5	$42,512,760
2014	2,105,268	36.1	$37,982,546
2015	3,225,935	45.9	$74,062,084
2016	5,969,891	103.0	$307,548,894
2017	6,723,286	87.9	$295,544,445
2018Q1	1,749,576	124.7	$109,056,884

Note: We distinguish in our calculations between expenditures by reducing entities and economic cost of emission reductions. Expenditures associated with emission reductions are simply (annual emission reductions [MT]) × (average annual credit price [$/MT]), where the average annual credit price represents the average of the 12 monthly CARB reported average credit prices. Costs of emission reductions can be represented by the area under an emissions reduction supply curve between the origin and market clearing price, here represented by the average annual credit price. If we make the simplifying assumption of a linear supply curve, costs will equal half of the expenditures, since the area of a triangle is one half the area of a rectangle with same base and height.

Source: CARB.

California LCFS Emission Reductions Relative to Total Emissions, 2013 - 2016 (MMT of CO_2e)

Year	Emission Reductions [A]	Transportation Sector		All Sources	
		Emissions [B]	Reduction Share [C] = [A] / [B]	Emissions [D]	Reduction Share [E] = [A] / [D]
2013	1.68	165.8	1.02%	447.6	0.38%
2014	2.11	167.14	1.26%	444.1	0.47%
2015	3.23	170.89	1.89%	441.4	0.73%
2016	5.97	174.01	3.43%	429.4	1.39%

Source: California Greenhouse Gas Emission Inventory.

2017 Direct Emission Reductions and Incremental Costs, Oregon's CFP

Observed Emission Reductions (MT) [A]	Average Credit Price ($ / MT) [B]	Estimated Incremental Cost of Reducing Emissions [C] = ([A] * [B]) / 2
17,751	48.1	$426,810

Note: We distinguish in our calculations between expenditures by reducing entities and economic cost of emission reductions. Expenditures associated with emission reductions are simply (annual emission reductions [MT]) × (average annual credit price [$/MT]). Costs of emission reductions can be represented by the area under an emissions reduction supply curve between the origin and market clearing price, here represented by the average annual credit price. If we make the simplifying assumption of a linear supply curve, costs will equal half of the expenditures, since the area of a triangle is one half the area of a rectangle with same base and height.

Source: Oregon Department of Environmental Quality.