Prioritizing greenhouse gas mitigation strategies for local governments using marginal abatement cost

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Abstract
Cities and other subnational jurisdictions have emerged as key agents for plans and policies to mitigate global climate change, with many presenting their strategies through climate action plans (CAPs). A critical review of CAPs from local government jurisdictions in California found that many CAPs lacked quantitative information on the economic cost and emissions reductions of proposed strategies, precluding decision-making based on the cost-effectiveness of competing policies. In response, this study develops a framework for comparing strategies based on their life cycle emissions mitigation potential and life cycle costs in a marginal abatement cost curve (MACC) to allow for side-by-side comparison. This framework was piloted with the cooperation of two California counties, Yolo and Los Angeles, to analyze six strategies in the transportation sector: intercity bike lanes, installation of roundabout intersections, alternative pavement rehabilitation, electric bus fleet, alternative fuel city fleet and installation of solar canopies in parking lots. Applying the life cycle approach revealed strategies that had net cost savings over their life cycle, indicating there are opportunities for reducing emissions and costs (roundabout intersections and solar canopies). The life cycle accounting processes also revealed that some emissions reduction strategies in fact increased emissions relative to no action (intercity bike lanes and alternative pavement rehabilitation). Despite the attractiveness of a life cycle-based MACC to support local government decision-making, there is the mismatch between a life cycle perspective and the annual emissions accounting used in most CAPs and associated policies. Future work should explore how to marry the life cycle and annual accounting methods and incorporate other sustainability concerns, such as equity and environmental justice, to support local decision-making.

1. Introduction
Global warming caused by anthropogenic emissions of greenhouse gases (GHGs) and the resulting climate change effects of warming is a defining issue of our time (United Nations 2019). Cities are responsible for the majority of anthropogenic GHG emissions, and as such, cities and other subnational jurisdictions representing urbanized areas have crucial roles to play in mitigating GHG emissions (Fong et al 2014). In fact, tens of thousands of cities have joined international compacts, such as the Global Covenant of Mayors (2020) and C40 (2020), to reduce emissions. Many jurisdictions use climate action plans (CAPs) to guide their approach to GHG mitigation. As described in C40’s climate action planning framework, a CAP serves to set and commit the jurisdiction to emissions reduction targets, present baseline and trajectory emissions, and outline actions that will be implemented to reach the set targets (C40 2020).

California has been a leader in the United States (US) and globally in the development of policies for reducing GHG emissions (California Assembly 2006). In particular, the Sustainable Communities and Cli-
mate Protection Act of 2008, or SB375, requires jurisdictions to develop GHG reduction targets and undertake specific actions to achieve them (California Institute for Local Government 2008). In response, many cities and counties in California have developed CAPs that identify GHG reduction targets and specific actions to achieve them. California local governments thus present a ready opportunity to systematically explore GHG mitigation strategies formulated at local jurisdictional scales. California counties are responsible for county-wide services and all services and land use planning in areas not included in incorporated cities. Counties therefore provide services in rural areas, and in urbanized areas between and around incorporated cities, and pockets within cities.

The transportation sector is a major contributor to GHG emissions in the US, causing 28% of total GHG emissions (EPA 2018). In California, the contributions from transport are even more dominant, comprising 41% of statewide emissions (CARB 2018). Thus, it is not a surprise that transportation is one of the key sectors identified in most California CAPs and targeted for reduction. Reducing motorized travel, measured in terms of vehicle miles traveled (VMT), is a crucial element for most reduction targets. However, the infrastructure required for nearly all travel modes includes hardscapes and may present an additional opportunity for GHG mitigation. Many cities and counties, and other jurisdictions such as port authorities, are responsible for managing a significant portfolio of transportation-related hardscapes including roadways, parking lots, airfields, and bike and pedestrian pathways. In the context of CAPs, these surfaces, and the vehicles and equipment that cities and counties operate on them, provide opportunities for directly and indirectly affecting GHG emissions, through changes in their operations, management, design, material selection, and others.

Unfortunately, the actual quantitative analysis of the mitigation potentials, and costs of mitigation for these strategies, have not previously been evaluated. This research examines the extent to which what data have been included in CAPs, then proposes a framework to inform implementation, and implements it in two case studies. The framework uses a lifecycle marginal abatement cost curve (MACC) to present the expected costs and emissions reduction potential of proposed CAP strategies.

A review of 37 California local government CAPs (out of 58 counties and 482 cities in the state) was conducted to better understand current approaches to reducing transportation-related emissions at the local level. In addition to compiling proposed emissions reduction strategies, the review considered whether a CAP quantified the expected emissions reduction and cost of planned strategies. The review found that only half of the reviewed CAPs quantified the expected GHG mitigation of proposed strategies, and even fewer quantified both emissions and costs (Lozano et al 2020, appendix A—table A1). Beyond simply calculating the GHG reduction potential and direct costs of a strategy, quantifying the life cycle environmental and economic benefits and burdens of actions relative to business-as-usual (BAU) practice would permit prioritization of the most cost-effective mitigation solutions, and ensure that indirect effects are captured.

Many previous studies of CAPs have examined the factors that affect their adoption in the first place (e.g., Pitt 2010, Kraus 2011, Krause 2012, Sharp et al 2011, Reckien et al 2018). While understanding the reason for adoption is important for regions where CAPs are not common, California has policies directing cities and other local jurisdictions to develop CAPs. As such, the more relevant literature for this study focuses on understanding the quality and thoroughness of CAPs and their level of consideration of factors important for their implementation. Tang et al (2010) reviewed CAPs in the US and found that while most showed awareness of the problem of climate change, and the need to address it, they conducted simplified assessments of GHG reduction, did not consider co-benefits or harms, and did a poor job of establishing actionable steps. A more recent study developed a system to rank CAP robustness and found that some US CAPs could greatly benefit from intra-regional collaboration and extra-regional support to improve CAP development and implementation (Deetjen et al 2018). However, collaboration between local governments may be hindered by the lack of regulatory frameworks to guide such interactions (OECD 2010). A study of CAPs developed in Brazil also found issues with quality and robustness, and in particular with the completeness and transparency of carbon accounting methods used in CAP development (de Souza Leao et al 2020). Earlier research by Blackhurst et al (2011) anticipated this kind of problem, arguing that CAPs should report uncertainty in their emissions inventory, and additionally that they should disaggregate sectors and link emissions to local organizations to increase accountability.

An additional challenge for making CAPs more robust is the lack of consensus on criteria for evaluating CAP strategies, and an attendant prioritization framework as part of evaluation (OECD 2010, Neves 2013, C40 2017). Only one previous study was found that proposes a general framework for use in local government climate action planning. Balouktsi (2019) proposes a multi-criteria decision analysis framework that considers quantitative economic and environmental data, qualitative social and technical data, and stakeholder preferences. Because this framework attempts to integrate many complexities of decision-making, its application may not be feasible for many local governments. A separate study applied a CAP model that would aid in action prioritization for the City of New York (C40 2017). While this CAP framework includes scoring rubrics for (co)benefits and feasibility, it is apparent that the prioritization relies first and foremost on GHG reduction
potential, followed by cost. Unfortunately, this framework for the City of New York (as well as the framework proposed by Balouktsi) did not quantify the lifecycle emissions and costs of the proposed actions.

Given the lack of a quantitative, decision-oriented framework for CAPs and local government climate change mitigation more broadly, the study presented in this paper develops a lifecycle-based MACC framework and applies it to two case studies.

MACCs have previously been used to assess strategies across various sectors in the US (Lutsey and Sperling 2009), internationally across various sectors (Moran et al 2011, de Souza et al 2018), and in the transportation sector for a state department of transportation (Harvey et al 2019). While the MACC approach is limited, insofar as it is two-dimensional (cost and GHG mitigation potential only), it provides easily interpretable quantitative analysis that can help guide prioritization of climate action strategies. When jurisdictions have limited resources, this can be a first step for including quantitative analysis in deliberations over which strategies to prioritize.

2. Problem statement

The goal of this research is to deliver a decision support framework for assessing the expected life cycle GHG (LCGHG) mitigation and life cycle cost (LCC) of mitigation actions considered by local governments resulting in a GHG MACC. The mitigation actions considered here are limited to the transportation sector, but the framework could in theory be applied across all sectors. The framework and tool will provide two benefits: first a robust method that provides local governments with a set of actions with quantified GHG mitigation values; and second, given constraints on funding faced by all jurisdictions and agencies, the use of this tool could lead to increasing mitigation targets or achieving existing targets at less cost. These advantages are not unique to this particular application of an MACC. However, the vision for replicable MACC decision-support developed for CAPs has not previously been tested. In California, only one CAP-related MACC was identified (Romanow et al 2018). More broadly, two transportation-related MACCs were found (Lutsey and Sperling 2009, Harvey et al 2019), both focusing on state-level strategies, and only Harvey and colleagues considered emissions on a life cycle basis.

3. Materials and methods

A LCGHG MACC offers the ability to combine and compare the impacts and cost-effectiveness of a wide range and large number of GHG mitigation options. Borrowing from economics theory, the MACC approach shows graphically the supply of a given resource (on the x-axis) that is available at a given price (on the y-axis). Depending on the use and derivation of the costs and cumulative emission reduction data, the curves can more aptly be labeled as marginal abatement, incremental cost, cost of conserved carbon, or cost-effectiveness curves. When shown as blocks for the effects of discrete changes, such as from different actions, the curves can show the incremental contribution to achieving a goal and the decreasing cost-effectiveness as additional actions are taken. This approach also uses life cycle, rather than direct, emissions accounting. LCGHG emissions accounting considers emissions generated throughout the supply chain of a product or process, and also typically considers system-wide or consequential effects on emissions as well. A carbon footprint is a narrow implementation of life cycle assessment (LCA), since LCA typically includes a larger number of environmental impacts in addition to GHGs. The goal of LCAs, especially those implemented to understand the prospective impacts of a policy or technology, typically includes anticipating unintended consequences, positive or negative, of a product, policy or action.

To test the viability of the framework, it was applied to two case study jurisdictions. After cataloging potential GHG mitigation strategies pulled from existing CAPs, jurisdictions and stakeholders were contacted to
identify local governments interested in partnering to compile data and develop MACCs tailored to their conditions. Two California counties, Yolo County and Unincorporated Los Angeles County, were selected as partner jurisdictions, and were interested in evaluating several GHG mitigation strategies in their respective CAPs. For each selected strategy, the LCGHG emissions and LCC were calculated.

The LCGHG and LCC are calculated over a 25 years analysis period, and cost calculations include a 4 percent discount rate, a long-term rate typically used in California state government economic analyses. With respect to the LCC, there are costs and savings that apply to the implementing agency as well as other affected parties. Because these results are meant to inform the spending of agencies, the life cycle agency cost is reported, which excludes user and other social costs. A generalized system boundary is provided in figure 1, and equations describing LCGHG calculations and LCC are shown in equations (1) and (2), respectively.

\[
\text{LCGHG} = \sum_{x=0}^{25} \text{emissions produced}_x - \text{emissions reduced}_x
\]

\[
\text{LCC} = \sum_{x=0}^{25} (\text{incurred costs}_x - \text{generated benefits}_x) + (1 - 0.04)^x - \text{salvage value}_{x=25}
\]

Yolo County prioritized the evaluation of four transportation strategies: intercity bike lanes, converting stop–start intersections to roundabouts, installing solar panel canopies on county parking lots, and full depth reclamation (FDR) in lieu of conventional pavement rehabilitation methods. Interestingly, Yolo County’s strategies all focused on interventions having to do with pavements and hardscapes. Unincorporated Los Angeles County prioritized two strategies, both of which focused on changing vehicles: electrifying transit buses, and implementing alternative fuels for the county-owned vehicle fleet. The following sections describe the modeling approaches used to represent the strategies and the results of prioritization for each county.

3.1. Yolo County strategies
3.1.1. Intercity bike lanes

This strategy analyzes the impacts of building new bike paths and lanes between various communities across Yolo County. The new construction is expected to reduce vehicle travel by moving some fraction of drivers out of vehicles and onto bicycles, thereby reducing vehicle travel and associated emissions. In this LCA, the functional unit is defined as the new construction and maintenance of 1 km of bike path or lane over a 25 years analysis period. In addition to the construction and maintenance phases, this analysis also considers the effect of the new infrastructure on VMT.

Bike paths are designed and built to be physically separate from vehicle roadways. Newly constructed paths are 3.05 m (10 ft) wide and 0.115 m (4.5 in.) thick and are expected to last 15 years before needing maintenance (Bicycle Plan 2011). Bike lanes not separate from vehicle roadways are assumed to be constructed as 1.22 m (4 ft) wide extensions on both sides of an existing road and are the same thickness as bike paths (Yolo County TAC 2013). Both cases assume conventional asphalt concrete (6% binder and 94% aggregate) is used for construction and maintenance. The maintenance consists of milling 0.045 m (1.8″) of the surface later and overlaying a 0.06 m (2.4″) thick asphalt layer at 15 years. The lifecycle inventories (LCIs), which track the material inputs and outputs of processes and are used to quantify the resulting net changes GHG emissions, used for the analysis were developed by the UC Pavement Research Center (UCPRC) (Saboori et al 2021). The cost of materials, construction, and maintenance were estimated using the Caltrans Cost Data Book (CCDB) (Caltrans 2018). An adjusted unit price value was selected based on the average of several projects.

The net impact of the new bike paths and lanes on GHG emissions is dependent on the change in vehicular travel induced by the availability of bicycle infrastructure which could induce some vehicle trips to be replaced with bicycle trips, thereby reducing motorized vehicle travel. While there is debate surrounding the estimation of this change, this study used a calculator developed by the California Transportation Commission Active Transportation Program through the California Air Resources Board that estimates the impacts of new bike infrastructure based on the expected changes in VMT (CARB 2016a, 2016b). The calculator requires the user to input average daily travel (ADT) in both direction of the roads parallel to the proposed bike infrastructure. The ADT data relevant to the proposed infrastructure were acquired from the traffic count of three primary cities in Yolo County: Davis (City of Davis 2019), Woodland (City of Woodland 2015), and West Sacramento (City of West Sacramento 2017). Note that the city of Davis is home to the University of California, Davis (UC Davis), a major destination for residents across Yolo County. Davis has extensive bike infrastructure used by permanent and seasonal student residents alike. It is therefore assumed that the roads are used for 200 days annually, which is the average number of annual academic working days. Note, however, that Davis leads the country in commuters traveling by bicycle, at nearly 20% (McKenzie 2014). This calculator may, therefore, underestimate displaced VMT resulting from additional infrastructure in Yolo County. Additional information
on the methods, assumptions, specific data, and sensitivity analysis on VMT replacement can be found in Kendall et al (2020a) (section 3.2).

3.1.2. Full depth reclamation compared to conventional pavement rehabilitation methods

Yolo County plans to fund a rehabilitation project on a 5.2-mile-long portion of rural road south of West Sacramento. The considered options are as follows:

- Mill-and-fill that mills 5.1 cm (2 in.) and overlays 10.2 cm (4 in.) of asphalt
- FDR with a 6.4 cm (2.5 in.) asphalt overlay using 3 percent portland cement (FDR + PC)
- FDR with a 6.4 cm (2.5 in.) asphalt overlay using 2.5 percent foamed asphalt and 1 percent portland cement (FDR + FA + PC)

The methodology used follows the Federal Highway Administration guidelines for conducting pavement LCA (Harvey et al 2016). The LCA estimates and compares the energy and material consumption of the three rehabilitation options across the material production stage, transportation of materials to the site, and construction activities. The system boundary also includes the transportation of the waste materials to asphalt plants for recycling or landfills when conducting mill-and-fill. All transportation distances are assumed to be 80.5 km (50 miles).

It is assumed that each rehabilitation option must be repeated every 10 years. It is also assumed that the options perform equally (no difference in degradation or effects on travel) throughout the analysis period. The cost of construction was acquired from the CCDB (Caltrans 2018). The LCI used for the analysis were developed by the UCPRC (Saboori et al 2021). Additional information on the methods, assumptions, specific data, and sensitivity analysis can be found in Kendall et al (2020a) (section 3.5).

3.1.3. Converting stop–start intersections to roundabouts

There are busy intersections across Yolo County with stop signs which require vehicles to come to a complete stop before proceeding. This strategy examines the impacts of constructing intersections with roundabouts instead of typical intersections with stop signs. Roundabout intersections reduce the amount of braking and acceleration required, thereby reducing fuel consumption and related emissions. Specifically, this analysis considers the construction and maintenance of each option, as well as the difference in operation of the vehicles, for three intersections along County Road 98 over a 25 years analysis period. It was assumed the roundabout will be constructed using portland cement concrete (PCC) while the traffic lanes will have a hot mix asphalt (HMA) top layer. The central structure is assumed to require no maintenance, whereas the roads will require a ‘mill and overlay’ treatment every seven years, which consists of milling 0.045 m (1.8 in.) of surface layer and overlaying a 0.06 m (2.4 in.) thick conventional asphalt concrete layer.

The CCDB provided the relevant cost information used (Caltrans 2018). The use stage, which includes user costs and impacts, accounts for well-to-pump (WTP) and pump-to-wheel (PTW) impacts. WTP data were acquired from Argonne National Lab’s (ANL) GHGs, Regulated Emissions, and Energy Use in Transportation Model (GREET) for diesel and gasoline (ANL 2017). PTW data were estimated using the US Environmental Protection Agency’s (EPA) MOtor Vehicle Emission Simulator (MOVES) (EPA 2015). ADT data for County Road (CR) 98 were acquired from the City of Woodland traffic counts (City of Woodland 2015), while that of the three roads connecting to CR98 were acquired from City of Davis traffic counts (City of Davis 2019). Additional information on the methods, assumptions, specific data, and sensitivity analysis can be found in Kendall et al (2020a) (section 3.3).

3.1.4. Installing solar panel canopies on county parking lots

Parking lots can double as electricity production sites through the installation of solar canopies. These structures not only support solar photovoltaic (PV) panels that produce electricity from sunlight, but they also provide shade and protection to the vehicles parked underneath. This strategy assesses the installation of solar canopies in various Yolo County owned parking lots. The scope of installation was determined by assessing a list of potential county-owned sites developed primarily in conjunction with the Yolo County Department of General Services. Sites considered in the assessment had minimal to no tree cover, thereby reducing the environmental impacts and cost of plant removal required for the installation of solar canopies. Within sites, solar canopies were assumed to be installed over double-row parking spaces (where two rows of vehicles are parked facing each other) since single-lane parking spaces were often near the perimeter of the sites and therefore had more nearby plants and trees.

The foundational carport design modeled in this study covers six parking spaces—three rows of cars in double-row parking spaces—and is based on the design published by Structural Solar (2013). Each solar
canopy can support 48, 1 kW solar panels with dimensions of 1 by 0.68 m (39.7 by 26.7 in.), and it was estimated that a total of 104 such structures could be installed across all sites. The estimated emissions and cost consider the material requirement for the supporting structure and the solar panels, as well as construction, maintenance, and end-of-life. The approximate solar canopy model and the final list of sites can be found in Kendall et al (2020a) (section 3.4.2).

Because studies on the life cycle emissions of solar PV have varying assumptions about efficiency, irradiance, lifetime, and more, a critical study by Hsu et al (2012) harmonized results to provide an average emissions factor: 52 g CO2e per kW h. Using the harmonizing assumptions, these results could also be reported as 276 kg CO2e per square meter of panel. Combining this with the area covered across all installations results in a total rated solar capacity of 0.71 MW. Using average electricity production values in California, these installations are expected to produce nearly 3.2 MW h of electricity at peak performance (Sendy 2017). It is assumed that the solar panels have a lifetime of 25 years and a 0.5% annual performance degradation rate, consistent with the literature (Hsu et al 2012).

The solar panels are supported by the carport structure, the materials of which were modeled after the information published by Carport Structures Corporation (2019). All beams are steel, and the primary load-bearing beams are supported by a PCC base to protect from vehicular damage. The total material needed was determined by designing solar carport installations for each site and referencing the model developed by Structural Solar. The lifecycle emissions of steel were acquired from the Ecoinvent database (Wernet et al 2016), and of PCC from Saboori et al (2021). The cost of installation was estimated by referencing the prices listed by the California-based Solar Electric Supply (2019). A median price of installation for projects between 50 and 250 kW (a range which each individual location assessed fell into) was $1.40 per kW.

The electricity produced by the solar carports is added to the grid, and Valley Clean Energy (Yolo County’s primary utility) confirmed that the energy would most likely qualify for monthly net metering. That is, the electricity produced would offset the charges of electricity consumed through a monthly credit, as long as the energy produced is not greater than the energy consumed (which is unlikely). This means the electricity is valued at market price, which was assumed to be $0.10 per kW h given the variability in the size of the nearest facility and the chosen rate plan. Due to high uncertainty surrounding electricity price forecasting, the price of electricity is assumed to be constant. This electricity would offset emissions from electricity production at the statewide level. Therefore, this study assumes the displaced emissions are a function of the average California electric grid carbon intensity over the 25 years analysis period, which was estimated using the Energy Information Administration’s (EIA) Annual Energy Outlook’s projected grid mix (specifically from 2020 onward; the grid mix has been provided in the appendix A) combined with the fuel source emissions values provided by the GREET 1 model (ANL 2017). Additional information on the methods, assumptions, specific data, and sensitivity analysis can be found in Kendall et al (2020a) (section 3.4).

3.2. Unincorporated Los Angeles County strategies

3.2.1. Electrifying the Foothill Transit bus fleet

This strategy examines the electrification of the Foothill Transit bus fleet, which serves incorporated and unincorporated regions of Los Angeles County. This transition to electric buses (E-buses) reduces GHGs compared to compressed natural gas (CNG) buses through reduced use stage emissions. The transition plan was laid out in the In Depot Charging and Planning Study developed by Burns & McDonnell Engineering Company, Inc. (B & M 2019) for Foothill Transit. Specifically, it determined the fleet size, required infrastructure including installation of solar PV to support energy needs, maintenance requirements, expected electric energy needs, and expected cost. Information on the bus fleet can be found in table B5.1. The electrification scenario is compared to one where the organization continues relying on CNG buses.

The LCI data used in this analysis were acquired from the GREET model (ANL 2018). This included glider data (the vehicle excluding the powertrain) of both CNG and electric buses, as well as data on batteries for the latter. Though no studies estimated the production-phase emissions of 43-foot double-decker bus gliders, it was deemed justified to assume they produced comparable emissions to 60-foot single deck buses based on a comparison of curb weights (BYD 2019a, 2019b). Maintenance-phase emissions were estimated using an economic input–output life cycle assessment (EIO-LCA) model (Weber et al 2009), which relates the environmental impacts of a sector to the economic value of sets of activities, thereby allowing one to estimate impact from cost. Using methodology developed by Ercan and Tatari (2015), EIO-LCA data on automotive repair and replacement were used to estimate the emissions from engine repair. It was assumed that engine repair or replacement is required every six years. Electric buses were assumed to need battery replacements every 6 years as well. The resulting emissions were estimated using GREET-derived values.

The report by B & M estimated the annual electricity consumption for the fleet. Additionally, they considered the installation of new solar panels to be installed gradually up to a total capacity of over 1.3 MW. Therefore, the demand would be met partially through on-site solar, with the remainder provided by the local
utility. This study considers emissions rates for both sources. The emissions rate for solar and for the average grid were calculated using the same methodology outlined previously in the section 'installing solar panel canopies on county parking lots.' The quantity of energy provided by each source was specified in the B & M report, and this information was combined with the calculated emissions rates to estimate the use-phase emissions of the electric bus fleet. As for charging the buses, it was assumed that 325 W chargers are installed annually as the fleet grows and are replaced every 12 years. The emissions rates were taken from a study by Bi et al (2018) which provided values for chargers up to 100 W. Bi et al argued that charging capacity scales linearly with material required, so this study assumed that environmental impacts scale linearly as well. It was also assumed that the trend can be extrapolated beyond the charging capacity examined by Bi et al.

The use-phase emissions for the electric buses were derived from the reported annual electricity use provided by B & M. These data were combined with a model of the annual average carbon intensity of the California electric grid (by combining EIA’s projected grid mix with GREET’s fuel source emissions— as described at the end of section 3.1.4) to estimate the annual GHG emissions generated through charging. The use-phase emissions of CNG buses are linked to fuel use. B & M did not report current fuel use rates or the annual distance traveled by the bus fleets. Annual distance traveled was derived by combining the reported electricity use of electric buses with an assumed average fuel economy for electric buses. It was assumed that CNG buses and electric buses travel the same distance. This value came out to be 30.6 million kilometers (approximately 19 million miles) traveled by the buses annually. The kilometers driven by either type of bus was estimated by comparing the electric buses available in a given year to the average number of electric buses available over the analysis period. The emissions rate for CNG travel was acquired from the Mobile Source Emission Inventory (EMFAC) released by the California Air Resources Board (2017). This study assumed an average bus speed of 32 km h$^{-1}$ (20 mph). Additional information on the methods, assumptions, and sensitivity analysis can be found in Kendall et al (2020a), (section 4.1).

Only part of Foothill Transit’s total service affects areas in Unincorporated LA County. Therefore, representatives of Foothill Transit counted the number of bus stops across their entire service area, as well as those in Unincorporated LA County. Those numbers were 1935 and 260, respectively. Approximately 13.4% of Foothill Transit’s bus stops are in Unincorporated LA County. Therefore, if bus stops are assumed to be linearly related to the service provided and consequent emissions, 13.4% of total emissions reductions achieved by electrifying Foothill Transit’s bus fleet can be attributed to Unincorporated LA County. However, the number of routes that stop at each bus stop is closer to 10%. Thus, an estimate using bus stops finds that between 10 and 13.4 percent of costs and emissions reductions from a transition to E-buses can be attributed to Unincorporated LA County. An average of 11.7 percent was used.

### 3.2.2. Implementing alternative fuel vehicles for the LA county fleet

This strategy examines the lifecycle environmental impacts and costs of transitioning all 3913 vehicles in the LA County fleet to alternative fuel vehicles (AFVs): either electric or biodiesel. The study compares the BAU case, a gradual transition to AFVs based on each vehicle’s date of purchase, and an all-at-once scenario where all vehicles transition to AFVs within the first year. The analysis is split into two parts: the vehicle cycle, which includes vehicle production (all processes from raw material extraction to delivery of the vehicle to the end user) and vehicle end-of-life (which includes recycling, landfilling, or transferring to a third party for which a salvage value is assigned); and vehicle use, which captures the emissions and cost of fuel production (WTP) and combustion (PTW), as well as maintenance and repairs.

Current information on the LA County fleet was provided by the county’s Internal Services Department (ISD)$^5$. These data included the model year, make, fuel type, lifetime accrued miles, fiscal year (FY) 2018–2019 distances driven, fuel dispensed, fuel economy (if known), FY 2018–2019 maintenance and repair costs, and department of use. Historical data on vehicle fuel efficiency was compiled from the EPA (2019) and the EIA (2019). This information was combined with data on the annual VMT to estimate the fuel consumption of each vehicle in the fleet. The EIA also provided projections of fuel efficiency according to vehicle and fuel type.

Lifecyle cost was calculated by accounting for fuel prices, purchase prices of new vehicles, maintenance, and salvage value of vehicles at the end of the analysis period. The Alternative Fuels Data Center (2019) published historical prices of alternative fuels, reported in dollars per gasoline gallon equivalent. These values were combined with the previously estimated fuel efficiency data to calculate the cost per mile traveled for each vehicle–fuel combination. While most projected prices for fuels are available through the EIA (2019), they only provide projections for regular diesel. To estimate the projected price of B100 (100 percent biodiesel) and B20 (biodiesel blended with petroleum diesel at 20%), a price ratio of these fuels compared to regular diesel was calculated for the past three years. These price ratios were then applied to the projected price of regular diesel

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$^5$ Provided on November 27, 2019 by Randy Martin (RMartin@isd.lacounty.gov).
Table 1. Lifecycle GHG mitigation and LCC results of each strategy evaluated.a

| Initial agency cost | Lifecycle agency cost | Lifecycle emissions reduction (tonnes CO2e) | Cost per tonne CO2e reduction |
|---------------------|-----------------------|------------------------------------------|-------------------------------|
| Intercity bike lanes | $103 000 000 | $146 000 000 | −15 000 | N/A |
| FDR options for pavement rehabilitation | $(3080 000) | $(3080 000) | −60 | N/A |
| Converting intersections | $60 000 | $(19 000) | 87 000 to 107 000 | −$0.2 |
| Solar canopies | $979 000 | $(770 000) | 1470 to 2210 | −$572 to −$763 |

Unincorporated Los Angeles County

| Foothill bus fleet electrification | Undetermined | $25 400 000 | 78 000 to 87 000 | $133 to $327 |
| AFVs | $718 000 000 | $173 000 000 | 117 000 to 118 000 | $1477 to $1494 |

*aNote: parenthetical costs are net savings (negative cost), and negative lifecycle emissions reduction values are net positive emissions.

to estimate a projected price for B20, B100, and RD100 (100 percent renewable biodiesel). Correction factors were determined by comparing historical energy prices of California and the US national average, and subsequently applied to the price of gasoline, diesel, electricity, and natural gas.

Vehicle prices included in this study include both past and future values. Historical purchase price data were acquired from the California Department of General Services, which captures purchases made by all state agencies. Their 2011–2014 database was used to acquire information on purchases made after 2004. Using linear regression, a trend was estimated to relate the vehicle price and age for all vehicle types. Price projections for all vehicle–fuel combinations were provided by the EIA (2018). This information was also used to estimate the salvage value of vehicles at the end of the analysis period given the amount of time before their end of useful service life.

The environmental impacts of the vehicle cycle include the energy consumption and GHG emissions from raw material extraction through delivery of the new vehicle. There are also impacts at the end of the vehicle’s service life, particularly from landfilling or recycling. Additional items included in this LCA are fluids, batteries, and tires. Nearly all data used in this portion of the study were compiled from the GREET model (ANL2017), unless stated otherwise.

The environmental impacts of fuel use considered in this study include pre-combustion (WTP) and combustion (PTW). Combined, these impacts are referred to as well-to-wheel (WTW) and are reported in grams of CO2e per mile traveled (the scope of each of these categories is depicted in figure A1). The LCI data needed to characterize fuel use impacts was taken from the GREET WTW Calculator tool (ANL2018). The fuel mix for the 2018 California grid and the pathways for biofuel production (specifically for different blends of ethanol and gasoline as well as of biodiesel and diesel) were acquired from the GREET model (ANL2017). Additional information on the methods, assumptions, specific data, and sensitivity analysis can be found in Kendall et al (2020a) (section 4.2).

3.3. Sensitivity analysis
To understand how results change by varying a number of factors, various sensitivity analyses were conducted. The following are included in the results. For solar canopy installation, a range of installation costs was considered (±$0.20 per W), as well as lower solar PV lifecycle emissions (24 g CO2e/kW h, from Fthenakis and Kim 2011). For converting intersections, sensitivity was conducted on the traffic rate (±10 percent) as well as on user fuel cost, but the latter does not affect agency cost so it is not reported alongside the MACC. For AFVs, a range of adoption rates was considered, from gradual implementation to all-at-once.

For electrifying the Foothill bus fleet, the study considered the following scenarios: (i) acquisition of relevant subsidies, and (ii) omitting the assumption that solar panel installation offsets electricity consumption from the grid. In reality, the electricity generated from the installed solar panels is added to the grid through a net-metering arrangement, and the bus fleet therefore still pulls all of its electricity from the grid. Therefore, two methods of electricity accounting are considered: first, in the baseline case, the fleet is modeled as if the solar energy it produces is used to directly charge its buses, and draws the remaining demand for electricity from the grid. In the second case, recognizing that 1.3 MW of solar has only a marginal effect on the GHG intensity of the greater grid mix, the carbon intensity of the electricity consumed by the fleet is essentially unchanged from the projected value and thus the fleet is assigned the average projected grid emissions for all their demanded electricity. Under both scenarios, the fleet is still credited for the monetary benefits generated by net metering from their solar panel installations.

The data for all evaluated strategies can be found in a data repository (Kendall et al 2020b).
4. Results

Summaries of individual results are followed by table 1, which reports the results for each strategy evaluated based on net LCCs to the agency and net life cycle greenhouse gas emissions, including a range of values based for relevant sensitivity analyses. Results are then presented in MACCs.

Results for intercity bike lanes show that the strategy fails to achieve GHG reductions and does not lead to agency savings. Emissions associated with the installation and maintenance of the bike lanes and paths were greater than the emissions avoided through reduced vehicle travel. Because only agency costs were considered, and the strategy requires investing in new infrastructure, it was inevitably going to lead to a net increase in cost to the agency. If the scope of this calculation had included user costs, there is some chance that user savings (by reducing costs of vehicle use) could have reduced the net total cost associated with this strategy.

Installing roundabouts also requires a higher initial agency cost for construction than stop–start intersections. However, in a life cycle perspective, almost $19 000 is saved in life cycle agency costs due to lower material needs and subsequent costs during maintenance. About 97 000 tonnes of reduction in LCGHG emissions is estimated due to decreases in pavement infrastructure maintenance and fuel emissions.

While there is an initial cost associated with the installation and maintenance of the solar canopies, net metering reduces the county’s utility costs, leading to a negative lifecycle agency cost. The emissions reductions achieved by producing renewable electricity (which reduces the amount of electricity from non-renewable sources) also offsets the positive emissions associated with installing and maintaining solar canopies. Thus, installation of solar PV on county parking lots provides the dual benefits of LCC savings and emissions savings.

Both types of FDR examined for pavement rehabilitation are cheaper to employ than the traditional mill-and-fill, but they produce a negligible increase in GHG emissions. Therefore, employing either FDR strategy on South River Road would likely save money, but they are not GHG reduction strategies.

Including production, maintenance, use-phase, and end-of-life emissions for bus fleet electrification shows that it would indeed reduce GHG emissions, but at increased cost compared to BAU. The referenced report provides the lifecycle cost of this transition but does not clearly lay out what the initial costs (purchase of the buses and required charging infrastructure) would be.

The results show that AFV adoption is a viable path to GHG reduction but at a higher cost than other strategies. The sensitivity of the adoption rate was explored and found that the results were consistent across adoption rates.

The analyses found that of the four strategies analyzed for Yolo County, intercity bike lanes are expected to lead to a net increase in emissions over their lifecycle, as do FDR options for pavement rehabilitation (though that increase is negligible). For this reason, the MACC produced for Yolo County includes just two strategies of the four examined (figure 2(a)). On the other hand, both strategies assessed for LA County are expected to reduce GHG emissions and are thus included in the MACC (figure 2(b)). Additional insight on the breakdown of life cycle emissions and costs can be found in the appendix A (figures A2–A6, tables A2–A8).
5. Discussion

The MACC approach presented, and tested for Yolo and LA counties, demonstrates the practicality of quantifying GHG reductions of CAP strategies and prioritizing them based on their cost-effectiveness. An additional benefit of quantification is identification of strategies that may not deliver GHG mitigation. For example, during the quantification process, the bike lanes indicated increased emissions for the assumptions made, while the use of FDR strategy on the county road indicated similar emissions to BAU. However, it must again be emphasized that these conclusions and others should be interpreted with care, and additional work should be done considering sensitivity analysis to help determine the robustness of the prioritization, and areas of where a strategy can be changed to improve its viability. For example, the GHG reduction associated with the bike lanes hinge on their ability to reduce vehicle travel and replacing vehicle travel with bike travel will depend on geographic considerations, such as whether bike lanes are likely to serve commuters. For this reason, site- or corridor-specific data collection of potential users could improve estimates of VMT change due to bike paths and is particularly relevant for Yolo County since UC Davis is its largest employer, and the University and City of Davis, CA have high bicycle mode shares and extensive cycling infrastructure (Lee 2019, City of Davis 2019).

These conditions mean that a site-specific analysis could result in a higher substitution rate for vehicle travel on some bicycle corridors in unincorporated Yolo County, and could reverse the findings presented here. In addition, impacts other than GHG reduction must also be considered. For example, bike lanes may provide other benefits to communities, such as recreation and co-benefits such as improved health, so their failure to reduce emissions does not mean they should not be pursued for other reasons.

Overall, the framework piloted in the studies presented in this paper demonstrated that it provides information that is currently missing from many CAPs: quantification of the effects of a proposed strategy on GHG emissions, and information regarding the cost-effectiveness of alternative strategies for an agency (almost all of which have constrained budgets). This pilot study also identified challenges and opportunities for this approach. It is noted that the expected emissions and costs of a given strategy varies based on the assumptions made, as evidenced by sensitivity analyses conducted. For example, solar canopies can have a net cost under a low price assumption for generated electricity, and net savings under a high price assumption. This highlights the importance of evaluating projects with site-specific conditions prior to implementation.

Discussion is also warranted for the assumptions made regarding the carbon intensity of the electric grid. Interestingly, the two types of electricity-based projects considered—electricity generation (solar canopies) and fleet electrification (Foothill transit and LA County fleet)—have opposite relationships with changes in grid carbon intensity. For solar canopies, a grid with lower carbon intensity than initially modeled means that the emissions offset attributed to solar generation is smaller, so the expected lifecycle emissions reduction of the strategy decreases. Conversely, an electricity grid with lower carbon intensity decreases the use-phase emissions of electric vehicles, thereby increasing the expected lifecycle emissions reduction of electrifying a fleet (as compared to BAU). There are certainly complications surrounding these strategies, such as considering the marginal emissions rate during times of electricity production (solar canopies) or consumption (charging), not to mention the additional impacts of California’s duck curve, which requires massive and inefficient load ramping in the late afternoon and early evening as solar power generation rapidly decreases, and which can lead to the curtailment of solar power (Denholm et al 2015). These uncertainties in how to account for electricity generation and consumption arise not just in the quantification of CAP strategies, but a myriad of other policies and practices that attempt to reduce GHG emissions, and merit additional research.

The calculation of an MACC is a snapshot in time. As such, there is a need to update data and calculations over time. For example, if an MACC value is calculated for the year starting 2020, what should that value be if it is implemented in 2025? While some changes could be anticipated, such as the future electricity grid mix, others cannot be and would likely require reanalysis. CAPs are typically updated every 10 years, but it is possible that relevant technologies or other factors change enough in that timeframe to warrant a reassessment. This could potentially be captured with sensitivity analyses, but might otherwise require additional resources to conduct new analyses. Similarly, the case studies in the two counties illustrated the challenge of data collection from the multiple divisions and agencies required to complete a CAP MACC. Implementation of MACCs for local governments will require engagement by multiple divisions and agencies, and thus requires sufficient resources and authority for coordination. There is an additional complication related to reporting. The MACC curve reflects a life cycle perspective, and a total present value of abatement. However, a jurisdiction subject to a CAP is required to submit annual production based GHG inventories. The required inventories are annual, not life cycle (nor consumption based) GHG estimations and thus the MACC estimates do not translate directly to the annual inventories. Thus, the decision-making basis—the MACC—is not directly related to how emissions are reported.

MACCs as used here assume independence of strategies, which is not always a valid assumption. Consideration should be given to interactions with the larger system of projects and policies, which some studies
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indicate is necessary in a prioritization framework (Givoni et al 2013, Taeihagh et al 2013). Further, the current implications of MACCs are limited because they are two dimensional—they consider only GHG reduction and cost—so the performance of strategies in other dimensions is not evident. Additionally, while the reporting of agency cost is relevant to the implementors of the GHG reduction strategies, a more complete perspective of the impacts would also include other stakeholders (those affected by the approach) and social costs, including consideration of the equity distribution of negative and positive impacts other than GHG reduction. Co-benefits for air quality are of particular relevance for many transportation interventions. For example, while E-buses may not immediately stand out as highly cost-effective measures for GHG reduction, mitigation of air quality emissions through electrification will likely have significant benefits for human health that also confer economic benefits to society such as reduced illness and health care costs (i.e., externalities). This study did not consider these and other co-benefits when calculating cost-effectiveness and is an opportunity for enhancing the scope of environmental benefits considered in prioritization of GHG mitigation strategies at the local scale. In other words, environmental justice concerns should be considered in the decision-making process. Many of the aforementioned shortcomings of MACC have been laid out in Kesicki and Ekins (2012). However, as stated in Eory et al (2018), one of the primary purposes of MACCs is to visualize the relative cost and emissions reduction opportunities of considered strategies in order to promote the complex discussion surrounding GHG reduction. Specifically, they are a useful tool when not used exclusively (Kesicki and Strachan 2011).

The framework presented in this research can be applied across various sectors, locations, and scales. A similar MACC framework was applied to projects in Toronto, specifically those to reduce fossil fuel use in transportation, reduce building energy use, and change the electricity energy supply (Ibrahim and Kennedy 2016). One paper presents the MACC framework applied to the industrial sector, including the hardscape sector, in Brazil, and found various cost-saving emissions reduction strategies over a 15 years analysis period (de Souza et al 2018). This framework could also be applied to urban hardscape and water management strategies in the United States, such as by combining with the efforts presented in Butt et al (2018). While not all of the aforementioned applications consider the lifecycle perspective, their application of an MACC framework to a variety of projects, combined with the results of the study presented in this paper, suggest that lifecycle MACCs could be developed for more projects and sectors than just the ones considered herein.

Future research should pursue solutions to challenges and opportunities for improving the MACC framework for CAP development and prioritization, with the ultimate goal of supporting quantification and prioritization for local and regional jurisdictions that face resource constraints and need decision-support for prioritizing CAP strategies.

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Data availability statement

The data that support the findings of this study are openly available at the following URL/DOI: https://doi.org/10.25338/B84615.

Appendix A

Table A1 summarizes relevant information that was collected when reviewing CAPs across California. It provides the name of the jurisdiction, the year of final CAP release, whether the CAP includes any quantitative
Table A1. Documentation of review CAP.

| Jurisdiction            | Year of CAP | Quantitative values? | Source                                                                 |
|-------------------------|-------------|----------------------|----------------------------------------------------------------------|
| Benicia                 | 2009        | No                   | http://sustainablebenicia.org/files/cap/TransportationandLanduse.pdf |
| Berkeley                | 2009        | No                   | https://cityofberkeley.info/uploadedFiles/Planning_and_Development/Level_3__Energy_and_Sustainable_Development/Berkeley%20Climate%20Action%20Plan.pdf |
| Chula Vista             | 2017        | No                   | https://chulavistaca.gov/home/showdocument?id=15586                 |
| Cupertino               | 2015        | Both                 | https://cupertino.org/our-city/Departments/environmentsustainability/climate-action |
| Emeryville              | 2016        | Both (scaled)        | https://ci.emeryville.ca.us/DocumentCenter/View/9328/Emeryville-CAP-2016-Implementation-Plan?bidId= |
| Fremont                 | 2012        | No                   | https://fremont.gov/DocumentCenter/View/19857/Climate-Action-Plan    |
| Fresno                  | 2014        | No                   | https://fresno.gov/darm/wp-content/uploads/sites/10/2016/11/F-2-Greenhouse-Gas-Reduction-Plan.pdf |
| Hayward                 | 2009        | No                   | https://hayward-ca.gov/services/city-services/climate-action         |
| Humboldt County         | 2012        | No                   | https://humboldtgov.org/DocumentCenter/View/1347/Draft-Climate-Action-Plan-PDF?bidId= |
| Lakewood                | 2015        | Both (scaled)        | http://lakewood.org/SustainabilityPlan/                              |
| Lancaster               | 2016        | Both (scaled)        | https://cityoflancasterca.org/Home/ShowDocument?id=52356            |
| Los Angeles County      | 2015        | No                   | https://planning.lacounty.gov/assets/upl/project/ccap_final-august2015.pdf |
| Manhattan Beach         | 2010        | No                   | https://citymhb.info/home/showdocument?id=16913                     |
| Marin County            | 2015        | Both                 | https://marincounty.org/~media/files/departments/cd/planning/sustainability/climate-and-adaptation/chpt4marincapupdate_final_20150731.pdf?fla=en |
| Monterey                | 2016        | No                   | https://monterey.org/Portals/0/Reports/ForPublicReview/Draft_Climate_Action_Plan.pdf |
| Oakland                 | 2018        | Cost only            | http://2.oaklandnet.com/oakcai/groups/pwa/documents/policy/oak069942.pdf |
| Palo Alto               | 2016        | Both                 | https://cityofpaloalto.org/civicax/filebank/documents/648148         |
| Piedmont                | 2018        | No                   | https://pi.c.piedmont.ca.us/climate-action-plan-2-0/                |
| Riverside County        | 2018        | Emissions only       | https://planning.rclma.org/Portals/14/CAP/CAP_071717.pdf           |
| Sacramento              | 2016        | Both                 | https://cityofsacramento.org/home/CorpFiles/Public-Works/Facilities/CityOfSacramento_1606_ClimateActionPlan_InternalOps_FINAL.pdf?fla=en |
| San Bernardino County   | 2014        | Emissions only       | https://gosbcta.com/plans-projects/plans-greenhouse.html           |
| San Francisco (city and county) | 2013    | Both (summed total)  | https://sfenvironment.org/sites/default/files/engagement_files/elle_cc_ClimateActionStrategyUpdate2013.pdf |
| San Jose                | 2018        | Both, agency and lifecycle | http://sanjosecity.gov/DocumentCenter/View/75035 |
| San Leandro             | 2009        | No                   | https://ca-ilg.org/sites/main/files/file-attachments/resources_ClimateActionPlan.pdf |
| San Rafael              | 2017        | Emissions only       | http://cityofsanrafael.granicus.com/MetaViewer.php?view_id=38&event_id=1118&meta_id=132004 |
| Santa Ana               | 2015        | Both                 | https://santa-ana.org/sites/default/files/Documents/climate_action_plan.pdf |
| Santa Barbara           | 2012        | No                   | https://santabarbaraca.gov/civicax/filebank/blob?oid=17716           |
| Santa Cruz              | 2012        | Both                 | http://cityofsanctacruz.com/home/showdocument?id=29361              |
| Shasta County           | 2012        | Both                 | https://co.shasta.ca.us/index/drm_index/ech/index/programs/BCAP/Draft/BCAP.aspx |
| Solana Beach            | 2017        | Both                 | http://solana-beach.hdso.net/docs/CM_ClimateActionPlan_Draft.pdf   |
| Sonoma                  | 2016        | Both                 | https://rcpca.ca.gov/wp-content/uploads/2016/07/CA2020_Plan_7-7-16_web.pdf |
| Stockton                | 2014        | Both                 | http://stockton.granicus.com/MetaViewer.php?view_id=48&clip_id=25016 &meta_id=4183169 |
| West Hollywood          | 2011        | Both (summed total)  | https://weho.org/home/showdocument?id=7949                          |
| Woodland                | 2017        | Both                 | https://cityofwoodland.org/DocumentCenter/View/834/Climate-Action-Plan-PDF |
| Yolo County             | 2011        | No                   | https://yolocounty.org/home/showdocument?id=18005                    |
| Yountville              | 2016        | Both                 | http://townofyountville.com/home/showdocument?id=4864               |
Figure A1. A system diagram of fuel-related emissions.

Figure A2. A breakdown of the emissions generated to produce bike paths and lanes, as well as the expected emissions reduction achieved through vehicle VMT reduction.

Figure A3. This figure shows the lifecycle emissions of the solar PV and infrastructure required to build solar canopies, as well as the emissions reduction achieved through the electricity that is generated.

data (emissions and/or cost), and the accessed link. It is specified whether the CAP provides emissions and/or costs, whether the data it considered a 'scaled' value (i.e. high, medium, or low expected emissions reduction), as well as whether information is provided for each strategy or summed across all strategies.
Figure A4. The emissions breakdown of three pavement rehabilitation methods, categorized by materials, transportation, and construction related emissions.

Figure A5. An emissions breakdown for the various life cycle stages of the two buses analyzed for Foothill Transit’s transition to electric buses.

Figure A6. The WTW emissions of the various vehicles considered in Los Angeles County’s transition to AFVs.
Table A2. A breakdown of costs associated with bike lane construction and maintenance.

|                      | Bike path | Bike lane | Total cost |
|----------------------|-----------|-----------|------------|
| Cost per lane-km     | $449 116  | $474 891  | —          |
| Initial cost         | —         | —         | $102 845 710 |
| Maintenance (discounted) | —       | —         | $42 880 486 |

Table A3. A breakdown of emissions associated with producing the two types of intersections, as well as the difference in emissions of the drivecycle associated with driving through each intersection. Note that drivecycle emissions are five orders of magnitude larger than pavement-related emissions.

| Life cycle stages | Current | Intersection with a roundabout |
|-------------------|---------|--------------------------------|
|                   | Tonne CO₂e | Tonne CO₂e |
| Conventional HMA  |           |                   |
| Material stage    | 4.97     | 4.46              |
| Transportation    | 0.03     | 0.027             |
| Construction stage| 0.34     | 0.305             |
| Maintenance stage (at every year 7) | 5.55 | 4.98 |
| Cement concrete for minor concrete (without secondary cementitious materials) | Material stage | — | 1.54 |
| Transportation    | —        | 0.229             |
| Construction stage| —        | 0.0147            |

Use phase (drivecycle) WTW emissions 1 109 687 1 012 700

Table A4. A breakdown of costs associated with producing the two types of intersections. The maintenance costs listed are not discounted since they repeat three times.

|                  | Stop–start intersection | Roundabout |
|------------------|-------------------------|------------|
| HMA-A            | $92 308                 | $108 418   |
| PCC              | —                       | $43 960    |
| Milling          | $38 400                 | $38 400    |
| Maintenance (every 7 years) | $130 708 | $87 953 |

Table A5. The projected California grid mix, acquired from the EIA, is provided in the table below.

| CAMX grid mix | 2020 | 2025 | 2030 | 2035 | 2040 | 2045 |
|---------------|------|------|------|------|------|------|
| Coal          | 4%   | 0%   | 0%   | 0%   | 0%   | 0%   |
| Petroleum     | 0%   | 0%   | 0%   | 0%   | 0%   | 0%   |
| Natural gas   | 33%  | 30%  | 27%  | 22%  | 20%  | 19%  |
| Nuclear       | 10%  | 5%   | 0%   | 0%   | 0%   | 0%   |
| Renewables    | 53%  | 65%  | 73%  | 78%  | 80%  | 81%  |

Table A6. A cost breakdown for solar canopy installation. Benefits are generated from electricity production.

Installation cost $979 200.00
Benefits generated $1749 418.07

Figure A1 outlines which aspects of a fuel’s lifecycle are considered WTP, PTW, and WTW.

Figures A2–A6 and table A3 provide additional information on the breakdown of emissions determined as determined by this research. Table A5 provides the projected California grid mix that is referenced for a number of strategies. Tables A2, A4, and A6–A9 provide a breakdown of the cost for the strategies.
Table A7. The cost breakdown for the three pavement maintenance options considered.

| Case                              | Item                | Unit  | Amount | Cost (million $) |
|-----------------------------------|---------------------|-------|--------|------------------|
| FDR, 0 foamed asphalt (%)         | Mill & fill         | CY    | 0      | 0.00             |
| 0.03 portland cement (%)          | Foamed asphalt      | TON   | 0      | 0.00             |
| 2.5 overlay thickness (in.)       | FDR                 | SQYD  | 73 216 | 0.60             |
|                                  | Portland cement     | TON   | 624    | 0.13             |
|                                  | Overlay             | TON   | 10 402 | 1.00             |
|                                  | Total               |       |        | 1.74             |
| FDR, 0.025 foamed asphalt (%)     | Mill & fill         | CY    | 0      | 0.00             |
| 0.01 portland cement (%)          | Foamed asphalt      | TON   | 520    | 0.24             |
| 2.5 overlay thickness (in.)       | FDR                 | SQYD  | 73 216 | 0.60             |
|                                  | Portland cement     | TON   | 208    | 0.04             |
|                                  | Overlay             | TON   | 10 402 | 1.00             |
|                                  | Total               |       |        | 1.89             |
| Mill & fill, 0 foamed asphalt (%) | Mill & fill         | CY    | 8228   | 3.12             |
| 0 portland cement (%)             | Foamed asphalt      | TON   | 0      | 0.00             |
| 4 overlay thickness (in.)         | FDR                 | SQYD  | 0      | 0.00             |
|                                  | Portland cement     | TON   | 0      | 0.00             |
|                                  | Overlay             | TON   | 0      | 0.00             |
|                                  | Total               |       |        | 3.12             |

Table A8. A comparison of emissions across the three considered scenarios.

| GHGs (tonne CO₂e) | BAU   | Gradual | All at once |
|-------------------|-------|---------|-------------|
| WTP               | 70 233| −32,739 | −33,744     |
| PTW               | 301 732| 232 874 | 232 343     |
| WTW               | 371 965| 200 135 | 198 599     |
| Net vehicle cycle | 364 054| 418 577 | 419 374     |
| **Total GHG emissions** | 736 019| 618 713 | 617 973     |
| Change in GHG emissions vs BAU | — | −117,306| −118,046    |
| Percent change vs BAU        | — | −15.9%  | −15.9%      |
| **Abatement cost ($/tonne CO₂)** | — | $1477   | $1494       |

Table A9. A breakdown of costs associated with transition the LA fleet to AFVs.

| Item                              | BAU   | Gradual | All at once |
|-----------------------------------|-------|---------|-------------|
| Fuel cost                         | 76.4  | 73.7    | 73.6        |
| New vehicles                      | 3105.0| 3827.0  | 3837.9      |
| reg & fees                        | 0.0   | 0.0     | 0.0         |
| Insurance                         | 0.0   | 0.0     | 0.0         |
| Maintenance                       | 108.4 | 113.2   | 113.2       |
| Salvage value                     | −2098.0| −2564.0| −2571.4     |
| **Total net cost**                | 1191.8| 1450.0  | 1453.2      |
| Net present value                 | 737.9 | 911.3   | 914.3       |
| **Total net cost (w/o Regulation & Insurance)** | 1191.8| 1450.0  | 1453.2      |
| **Net present value (w/o Reg & Ins)** | 737.9 | 911.3   | 914.3       |

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