Transportation Research Part D 57 (2017) 287–302

Contents lists available at ScienceDirect

Transportation Research Part D

journal homepage: www.elsevier.com/locate/trd

Life cycle ownership cost and environmental externality of alternative fuel options for transit buses

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ARTICLE INFO

Keywords:
Transit bus
Alternative fuel
Life cycle ownership cost
Externality
Greenhouse gas
Criteria air pollutant

ABSTRACT

This paper assesses alternative fuel options for transit buses. We consider the following options for a 40-foot and a 60-foot transit bus: a conventional bus powered by either diesel or a biodiesel blend (B20 or B100), a diesel hybrid-electric bus, a sparking-ignition bus powered by Compressed Natural Gas (CNG) or Liquefied Natural Gas (LNG), and a battery electric bus (BEB) (rapid or slow charging). We estimate life cycle ownership costs (for buses and infrastructure) and environmental externalities caused by greenhouse gases (GHGs) and criteria air pollutants (CAPs) emitted from the life cycle of bus operations. We find that all alternative fuel options lead to higher life cycle ownership and external costs than conventional diesel. When external funding is available to pay for 80% of vehicle purchase expenditures (which is usually the case for U.S. transit agencies), BEBs yield large reductions (17–23%) in terms of ownership and external costs compared to diesel. Furthermore, BEBs' advantages are robust to changes in operation and economic assumptions when external funding is available. BEBs are able to reduce CAP emissions significantly in Pittsburgh's hotspot areas, where existing bus fleets contribute to 1% of particulate matter emissions from mobile sources. We recognize that there are still practical barriers for BEBs, e.g. range limits, land to build the charging infrastructure, and coordination with utilities. However, favorable trends such as better battery performance and economics, cleaner electricity grid, improved technology maturity, and accumulated operation experience may favor use of BEBs where feasible.

1. Introduction

Transit buses provide short-distance public transportation service with multiple stops along fixed routes to serve citizens' mobility needs. Currently, there are 653 transit agencies operating in urbanized areas and 525 transit agencies in rural areas in the U.S. (Neff and Dickens, 2014). In 2013, these 1178 transit agencies operated a fleet of 65,950 active buses, which traveled 2.2 billion vehicle miles, and served 19.4 billion passenger miles (Davis et al., 2016). Altogether, transit buses consume 79 trillion Btu's of energy, or about 0.4% of energy consumed by on-road vehicles in the U.S. (Davis et al., 2016).

Alternative fuels and advanced technologies have the potential to reduce petroleum consumption and to mitigate unintended environmental consequences including climate change damages caused by greenhouse gases (GHGs) and health and environmental

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http://dx.doi.org/10.1016/j.trd.2017.09.023

Available online 08 October 2017
1361-9209/

damages caused by criteria air pollutants (CAPs) by substituting for conventional vehicles powered by petroleum fuels. Transit agencies are more willing, compared to mainstream private vehicle owners, to adopt alternative fuel vehicles. This is not only because they face a different cost structure (fueling costs are more important due to high mileages), but also because they have higher awareness and sometimes obligations to funding agencies to pursue fuel diversity and/or environmental sustainability (Werpy et al., 2010). In the past two decades, there has been an increase in the penetration of alternative fuels in the transit bus market. American Public Transit Association (APTA) reported that 20% of U.S. transit buses were powered by compressed natural gas (CNG) and liquefied natural gas (LNG) and blends in 2013. In addition, 13% of transit buses were diesel hybrid electric buses (HEBs) and another 7% used biodiesel. The so-called "zero-emissions buses" (which have zero tailpipe emissions during normal operation), such as battery electric buses (BEBs) and fuel cell electric buses, have also emerged in some regional markets (notably, California), as encouraged by state-level environmental regulations and incentive programs (California Air Resources Board (CARB), 2016a).

There is a growing literature that assesses alternative fuel options for transit buses. Table 1 provides a summary of the scope and conclusions of selected U.S. studies. We find that existing studies estimated lifetime ownership costs of purchasing and operating diesel, diesel HEBs, CNG, B20 (a liquid blend of 20% biodiesel and 80% diesel), and BEBs. All of these studies considered capital investment and lifetime operation costs related to bus purchases and uses, and most studies included capital investment related to supporting infrastructure such as refueling stations and garage modifications. We find that in addition to these techno-economic assessments, a few studies also conducted separate environmental assessments to estimate life cycle GHG and CAP emissions (Bi et al., 2016; Clark et al., 2007; Ercan et al., 2015; Lowell, 2012), and two recent studies monetized the impacts of GHGs or CAPs (Bi et al., 2016; Ercan et al., 2015). Furthermore, as summarized in Tong et al. (2015), a number of studies examined solely life cycle GHG emissions for the same set of fuel options.

Some insights emerged from Table 1. First, the focus of alternative fuel options has changed from studies published a decade ago (where CNG and diesel HEB are the primary focuses) to more recent studies (where BEBs are included), which clearly reflects the changing technology landscape. Second, baseline assumptions, in particular, diesel prices, assumed in these studies have changed over time to reflect market dynamics. This in turn changes conclusions from these studies because diesel prices impact life cycle costs of conventional diesel buses significantly (see, for instance, Clark et al., 2007, 2008). Finally, we find that technology assessments on transit buses still largely focused on ownership costs from transit agencies' perspectives. No study has included externality or external costs caused by by-products of bus operation, such as GHGs and CAPs in addition to ownership costs to estimate full societal costs. In our literature review, only two recent studies (Bi et al., 2016; Ercan et al., 2015) assessed external costs, but their assessments are incomplete. Bi et al. (2016) only included climate change damages, but recent studies have showed that CAP-related health and environmental costs from electricity generation are significant (Jaramillo and Muller, 2016; Tong, 2016). Ercan et al. (2015) considered external costs of both CAPs and GHGs. However, they used national-average damage estimates of CAPs, which may be inaccurate because CAP impacts are local.

In this paper, we estimate both life cycle ownership costs as well as life cycle externality of GHGs and CAPs for alternative fuel options for transit buses. In addition to a complete estimate of life cycle external costs using up-to-date emissions inventories and state-of-art marginal damage estimates, contributions of this paper also include a comparison between two types of BEBs (slow-charging and rapid-charging) and separate assessments for 40-foot buses and 60-foot buses. We believe that our contributions can help transit agencies, bus manufacturers, and policymakers gain a better understanding of benefits and costs of alternative fuel options. In addition, we also estimate the contributions from transit buses to CAP emissions inventory in hotspot areas of Pittsburgh, PA to understand the environmental impacts of bus operations at a finer geographic scale.

2. Method

2.1. Scope

We model a 40-foot bus and a 60-foot articulated bus separately. We consider new transit buses in Model Year 2015 with the following fuel options: a conventional diesel bus, a diesel HEB, a sparking ignition natural gas bus powered by CNG, a sparking ignition natural gas bus powered by LNG, a conventional diesel bus with B20, a conventional diesel bus with B100, a BEB with slow charging in a garage, and a BEB with rapid charging along a bus route. The two types of BEBs differ in onboard batteries and the charging infrastructure.

Table 2 lists key assumptions used in this study. Assumptions regarding fuel economy, battery size, and battery replacement are taken from Tong et al. (2015). Vehicle purchase prices are collected from California Air Resources Board (CARB) (2015a), and METRO Magazine (2015). Fuel costs are taken from U.S. Department of Energy (DOE) (2016). Vehicle operation and maintenance (O & M) costs (except fuels) are taken from California Air Resources Board (CARB) (2016b, 2016c). Infrastructure costs are taken from California Air Resources Board (CARB) (2015) and Gladstein Neandross & Associates (GNA) (2012). Finally, we assume the number of buses that share the refueling or charging infrastructure (100 CNG or LNG buses for a refueling station and 10 rapid-charging BEBs for a charging station) to calculate the per-bus infrastructure cost.

The system boundary for ownership costs is not limited to a bus itself, but also includes refueling infrastructure and maintenance garages. This is because transit agencies use refueling stations located within their property. In deploying alternative fuel buses, transit agencies should co-optimize bus fleets and refueling infrastructure (even though it may be contracted and owned by a third party) to maximize investment return. We assume the end-of-life impacts of alternative fuel technologies are roughly the same due to lack of data on disposal of new alternatives. We note that further study may be needed to investigate the end-of-life impacts as recently deployed alternative fuel buses reach their lifetime. In any event, end-of-life disposal should be small relative to operating

Table 1
Summary of alternative fuel assessment studies for transit buses in the U.S.

Study	Cost components[a]	Fuel options	Conclusions
Lowell et al. (2007)	Vehicle costs (purchase, fuel, O & M excluding fuel) and operator's labor costs.	Diesel, diesel HEB, CNG, hydrogen fuel cell electric bus, hydrogen fuel cell hybrid bus.	The net present value of projected total life cycle costs of fuel cell electric buses and fuel cell hybrid buses are higher than diesel, CNG, or diesel HEB buses for all scenarios considered.
Clark et al. (2007)	Vehicle costs (purchase, fuel, O & M excluding fuel) and infrastructure costs (refueling stations).	Diesel, diesel HEB, CNG, B20.	"Diesel buses are still the most economic technology. In the case where only 20% of the bus procurement cost was considered, as a result of subsidies, the four bus types had a sufficiently similar life cycle cost."
Clark et al. (2008)	Separate emissions estimates are available in Clark et al. (2007).	Diesel, diesel HEB, CNG, B20.	This report updated the results in Clark et al. (2007) using (higher) fuel costs in 2008. CNG buses are the most economic technology in four fuel price scenarios, and diesel HEBs are the least economic technology.
Clark et al. (2009)	Vehicle costs (purchase, fuel, O & M excluding fuel) and infrastructure costs (refueling stations and garages).	Diesel (pre-2007 and 2007), diesel HEB, gasoline HEB, CNG	"Each technology could possibly be a best choice in a real procurement and operation scenario, even when default values are used." Key factors include bus speed, annual mileage, cost assumptions, fuel prices, and purchase incentives may impact the comparison.
Johnson (2010)	Vehicle costs (purchase, fuel, O & M excluding fuel) and infrastructure costs (refueling stations and garages).	Diesel, CNG	CNG is profitable for large transit bus fleets (> 75 vehicles) unless one or multiple factors (such as diesel prices, CNG bus maintenance costs, bus annual mileage, and vehicle incremental costs) become unfavorable.
Science Applications International Corporation (2011)	Vehicle costs (purchase, fuel, O & M excluding fuel) and infrastructure costs (refueling stations and garages).	Diesel, biodiesel, gasoline, ethanol, CNG, LNG, hydrogen ICE, propane, dimethyl ether, electric trolleybus, BEB, diesel HEB, hydrogen fuel cell electric bus	"This guidebook begins with an overview of how to choose a transit bus fuel, followed by 13 chapters, each addressing one particular fuel or powertrain type." It also has an accompanying spreadsheet-based life cycle costs model, FuelCost2.
Gladstein Neandross & Asscoates (2012)		Diesel, CNG	"The overall economic feasibility to convert one bus depot to support CNG buses appears to be attractive."
Lowell (2012)	Vehicle costs (purchase, fuel, O & M excluding fuel) and infrastructure costs (refueling stations).	Diesel, CNG	"The pay-back period on the incremental purchase cost of CNG buses and fueling infrastructure, compared to diesel buses, is between five and eight years. CNG buses have 14% reduction in annual fuel costs compared to diesel buses."
McKenzie and Durango-Cohen (2012)	Separate emissions estimates.	Diesel, diesel HEB, CNG, hydrogen fuel cell bus.	"We find that the alternative fuel buses reduce operating costs and emissions, but increase life-cycle costs. The infrastructure requirement to deploy and operate alternative fuel buses is critical in the comparison of life-cycle emissions."
Trillium CNG (2014)	Vehicle costs (purchase, fuel, O & M excluding fuel) and infrastructure costs (refueling stations and garages).	Diesel, CNG	The payback periods of a small (50 vehicles) and a large (200 vehicles) fleet are 3.7/5.7 years and 2.0/4.0 years (without/with federal funding for bus purchase).
Ercan et al. (2015)	Vehicle costs (purchase, fuel, O & M excluding fuel), infrastructure costs (no details), and external costs (GHGs and CAPs).	Diesel, diesel HEB, B20, CNG, LNG, BEB	"This study finds an optimal bus fleet combination for different driving conditions to minimize life cycle cost, greenhouse gas emissions, and conventional air pollutant emission impacts. In heavily congested driving cycles such as the Manhattan area, the battery electric bus is the dominant vehicle type, while the hybrid bus has more balanced performances in most scenarios because of its lower initial investment comparing to battery electric buses."
Bi et al. (2016)	Vehicle costs (purchase, fuel, O & M excluding fuel),	Diesel, diesel HEB, plug-in charging BEB, wireless charging BEB.	"The wireless charging bus system has the lowest life cycle cost of US$0.99 per bus-

(*continued on next page*)

Table 1 (*continued*)

Study	Cost components[a]	Fuel options	Conclusions
	infrastructure costs (chargers), and external costs (GHGs).		kilometer among the four systems and has the potential to reduce use-phase carbon emissions attributable to the light-weighting benefits of on-board battery downsizing compared to plug-in charging"

Note:
[*]Acronyms explained: HEB, hybrid-electric bus; CNG, compressed natural gas; LNG, liquefied natural gas; BEB, battery electric bus; B20, A blend of 20% biodiesel and 80% petroleum diesel; B100, biodiesel (pure); O & M, operation and maintenance; GHG, greenhouse gas; CAP, criteria air pollutant.
[a] These papers have different details in estimating these cost components.

Table 2
Key technical and economic assumptions used in this study.

Variables	Bus size	Conventional diesel	Diesel HEB	CNG	LNG	Rapid-charging BEB	Slow-charging BEB	B20[d]	B100[d]
Fuel economy (MPGDE)	40-foot	4.8	5.76	4.3	4.3	22.1	18.9	4.8	4.8
	60-foot	3.3	3.96	3	3	15.2	13.0	3.3	3.3
Battery size (kWh/bus)	40-foot	0	5	0	0	88	324	0	0
	60-foot	0	5	0	0	102	377	0	0
Vehicle purchase price ($/bus)[a]	40-foot	$485,000	$758,000	$525,000	$525,000	$800,000	$800,000	$485,000	$485,000
	60-foot	$600,000	$1,115,000	$800,000	$800,000	$1,200,000	$1,200,000	$600,000	$600,000
Vehicle O & M cost (excluding fuel cost) ($/mile)	–	$0.85	$0.74	$0.85	$0.85	$0.60	$0.60	$0.85	$0.85
Battery replacement (probability during lifetime)	–	0%	50%	0%	0%	50%	50%	0%	0%
Range (mile)[b]	40-foot	690	720	600	640	41	130	690	690
	60-foot	475	565	480	510	33	104	475	475
Fuel cost ($/gallon of diesel equivalent)	–	$2.3	$2.3	$1.5	$2.1	$2.1	$2.1	$2.4	$3.0
Per-bus infrastructure cost ($/bus)	–	$0	$0	$50,000	$50,000	$45,000	$55,000	$0	$0
Electricity rate in Pittsburgh, PA	–	$0.055/kWh							
Discount rate	–	1%							
Bus annual mileage[c]	40-foot	37,761 miles/year (minimum 9882 miles/year, maximum 69,889 miles/year)							
	60-foot	32,719 miles/year (minimum 16,726 miles/year, maximum 44,912 miles/year)							
Bus lifetime	–	12 years							

Note:
[*]Acronyms explained: HEB, hybrid-electric bus; CNG, compressed natural gas; LNG, liquefied natural gas; BEB, battery electric bus; B20, A blend of 20% biodiesel and 80% petroleum diesel; B100, biodiesel (pure); O & M, operation and maintenance.
[a] All vehicles (except 60-foot BEBs) are available on the market. The prices of 60-foot BEBs are calculated from the 40-foot buses assuming the same relative costs with regard to conventional diesel. The battery sizes of 60-foot BEBs are calculated to achieve 80% of the range of the 40-foot BEBs.
[b] Range is calculated based on fuel economy, the size of fuel tanks/batteries, and usable fuel per tank/battery.
[c] Bus annual mileage data is provided by Port Authority of Allegheny County (PAAC).
[d] We assume B20 and B100 buses are identical to conventional diesel buses. This is a simplifying assumption.

impacts (MacLean and Lave, 2003). The metric that we use to compare across options is annualized costs evaluated over a bus lifetime of 12 years. We use a 1% discount rate following the Office of Management and Budget (2015). We use 2015 U.S. dollars and convert all other dollars using the Consumer Price Index (CPI) inflation calculator from the U.S. Bureau of Labor Statistics (2016).

We choose the Port Authority of Allegheny County (PAAC) in Pennsylvania for a case study. PAAC currently operates a transit bus fleet of 704 clean diesel buses and 32 hybrid diesel-electric buses (PAAC, 2015). Some assumptions we use are specific to PAAC (such as annual bus mileage, diesel price, electricity price, and GHG and CAP emissions of grid electricity in Allegheny County), but all the other assumptions are general to transit agencies in the U.S.

2.2. Life cycle ownership costs

We estimate life cycle ownership costs for a transit agency when a fleet of alternative fuel buses are deployed and the supporting infrastructure is built. Life cycle ownership costs consist of four components: bus purchase costs, fuel costs, O & M costs (except fuels), and upfront infrastructure costs (including building refueling facilities unless they already exist and garage modifications). These costs are then summed and converted into annualized costs using the formulas below. Key assumptions are reported in Table 2.

$$Annualized\ ownership\ cost = \frac{Vehicle\ \&\ infrastructure\ capital\ cost}{Annuity\ factor_{lifetime,\ discount\ rate}} + Annual\ O\ \&\ M\ cost \tag{1}$$

$$Annuity\ factor_{lifetime,\ discount\ rate} = \frac{1-(1+discount\ rate)^{-lifetime}}{discount\ rate} \tag{2}$$

One factor that may change bus purchase costs from a transit agency's perspective is the availability of external funding. For instance, the Federal Transit Administration (FTA) Section 5307 provides funding that may cover up to 80% of bus purchase costs (Clark et al., 2007; CARB, 2015). Thus, we present two life cycle cost estimates: (1) external funding pays for 80% bus purchase costs, and transit agencies only pay for 20% bus purchase costs and (2) transit agencies pay for 100% bus purchase costs. While the literature (Clark et al., 2007; CARB, 2015) and local transit agencies suggest that external funding is usually available, these two cases nevertheless present lower and upper bounds of the actual costs that transit agencies need to consider. We assume there is a 50% probability that HEBs and BEBs will need to replace their batteries once in year 7, following Tong et al. (2015). We note some studies (Ercan et al., 2015) assumed a higher number of battery replacements during the bus lifetime (3 times for HEBs and 4 times for BEBs) but we think their assumptions are likely to be an underestimate of battery lifetime. Bus manufacturers already offer base battery warranty for 3–7 years, or optional battery warranty for 12 years (New Flyer, 2016a). In addition, PAAC reported that some HEBs it has operated did not require battery replacement throughout their lifetimes. We assume a \$700/kWh battery cost for battery replacement (CARB, 2015). Fuel costs over a given period are calculated based on annual mileage, fuel economy, and fuel prices (National Renewable Energy Laboratory (NREL), 2016; U.S. Department of Energy (DOE), 2016). We do not account for fuel price changes over the bus lifetime as the actual fuel price trajectory is hard to project. Instead, we run a sensitivity analysis on fuel prices to understand their impacts.

Infrastructure costs are estimated using an engineering economics approach. A key step is to examine if alternative fuel buses require new refueling infrastructure and/or garage modifications (such as CNG, LNG, and BEBs) or if they work well with existing infrastructure (such as diesel HEBs and biodiesel). The infrastructure costs for natural gas buses are taken from a recent PAAC design study (Gladstein Neandross & Associates (GNA), 2012). Here we assume a high utilization rate of the natural gas infrastructure, which supports 100 natural gas buses. If the actual utilization rate is lower than assumed, each bus's share of the infrastructure cost will increase. We estimate charging infrastructure costs for BEBs based on communications with officials at PAAC, which has invited major BEB manufacturers to present and demonstrate their buses. We note that some infrastructure may have a longer lifetime than transit buses, so our cost estimates are likely to be on the high end for these infrastructure. However, we only include direct equipment costs for infrastructure costs, as with most studies listed in Table 1. Indirect equipment costs, such as capital investment to update grid connections (which might be needed for CNG/LNG refueling stations and BEB chargers), are not included because these costs are case-specific. Similarly, labor costs associated with the design and construction of infrastructure are not included.

2.3. Life cycle external costs

Transit buses emit GHGs (carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O)) and CAPs (nitrogen oxides (NO_x), carbon monoxide (CO), volatile organic compounds (VOC), particulate matter (PM), and sulfur dioxide (SO_2)) over the life cycle of bus operation. The life cycle components consist of bus operation (tailpipe exhaust, tire and brake wear), the process to produce and deliver fuels used to power a bus, and upstream activities that extract primary energy and feedstock used in fuel production processes. In addition, we include GHGs and CAPs from manufacturing additional lithium-ion batteries for HEBs and BEBs.

We characterize health and environmental damages caused by GHGs and CAPs using the damage function approach (U.S. NRC, 2010). Emissions change air concentrations due to physical and chemical processes (accumulation, dispersion and removal process). There are multiple mechanisms linking concentration changes to physical impacts: elevated concentrations of GHGs affect the energy balance of the earth, which could lead to climate change, such as temperature increase, precipitation change, sea level rise, and ocean acidification (IPCC, 2014); increased levels of $PM_{2.5}$ and ground-level ozone due to CAP emissions impose higher mortality and morbidity risks on the exposed human population, and contribute to soil and water acidification, reduced tree growth, reduced agricultural yields, and impaired visibility (Muller and Mendelsohn, 2007; Heo et al., 2016a). All of these physical effects are valued in monetary terms using market prices or estimated price proxies (such as willingness-to-pay) of non-market goods (Muller and Mendelsohn, 2007; Heo et al., 2016a).

In this paper, we assume that GHGs and CAPs emitted by transit buses are marginal. So we estimate the resulting external costs by multiplying the amount of emissions (by species and by location) with the marginal damage of a unit emission (of the same species emitted in the same location). There is a key distinction between GHGs and CAPs. GHGs are globally mixed so their marginal damages are the same around the world, but CAPs are locally mixed thus their marginal damages vary from region to region. For example, it is problematic to compare a ton of CAP emissions in New York City to a ton in Pittsburgh. The formulas to calculate climate change damages and air pollution damages are as follows.

$$Climate\ change\ damages = life\ cycle\ GHG\ emissions \times Social\ cost\ of\ carbon \tag{3}$$

$$Air\ pollution\ damages_{life\ cycle\ stage} = \sum_{CAP\ species} CAP\ emission_{CAP\ species,location,life\ cycle\ stage} \times Marginal\ Damages_{CAP\ species,location} \tag{4}$$

$$Life\ cycle\ air\ pollution\ damages\ (APD) = APD_{vehicle\ operation} + APD_{battery\ manufacturing} + \frac{APD_{upstream\ activities}}{vehicle\ fuel\ efficiency} \tag{5}$$

We use life cycle GHG emissions estimates in Tong et al. (2015) with adjusted fuel economy assumptions. In addition, we assume that B100 reduces life cycle GHG emissions by 50% compared with conventional diesel (U.S. Environmental Protection Agency (EPA), 2010). We convert all GHGs to CO_2-eqivalent emissions using Global Warming Potential (GWP) (IPCC, 2014). We use both 100-year and 20-year GWP, the latter of which leads to higher CO_2-equivalent emissions per unit of methane than the former. The marginal damage from a unit of carbon emission is called the *social cost of carbon* (SCC). A U.S. interagency group published SCC estimates for use in decision-making process (U.S. Environmental Protection Agency (EPA), 2015a). The SCCs are estimated using integrated assessment models (IAM) that model Earth's physical systems and economic systems. The most recent SCC estimates range from \$13 to \$120 (in 2015 dollars) for a metric ton of CO_2 emitted in 2015. In this paper, we use a median estimate of \$41 per metric ton of CO_2 emitted.

We use life cycle CAP emissions and the resulting air pollution costs estimated in Tong (2016) with adjusted fuel economy assumptions. Tong (2016) constructed a spatial life cycle CAP emissions inventory by U.S. counties. It used data sources such as U.S. Environmental Protection Agency (EPA)'s National Emissions Inventory (NEI) (U.S. Environmental Protection Agency (EPA), 2016a), U.S. EPA's Continuous Emissions Monitor System (CEMS) (U.S. Environmental Protection Agency (EPA), 2016b), Altoona Bus Research and Testing Center (2016), and the GREET model (Argonne National laboratory (ANL), 2016) to characterize CAP emissions from energy production processes, electric power grids, and bus operations in the U.S. Tong (2016) used two state-of-the-art models, the AP2 model (Muller, 2011; Muller and Mendelsohn, 2007) and EASIUR model (Heo et al., 2016a, 2016b) to estimate the environmental and health damages resulting from one unit of CAP emission in every county in the contiguous U.S. The two models take into account atmosphere conversion and dispersion of air pollution, exposed population, and health impacts of $PM_{2.5}$ and ground-level ozone on the exposed population (see Tong (2016) and Heo et al. (2016a) for details). Since PAAC's bus fleet primarily operates within Allegheny County in Pennsylvania, the health and environmental impacts of bus tailpipe emissions are estimated using the marginal damages for Allegheny County. We assume the electricity used to charge BEBs in Allegheny County is balanced in the RFC region (which includes Midwest/Mid-Atlantic states such as DE, IN, MD, MI, NJ, OH, PA, WV and parts of IL, KY, VA, and WI) defined by the North American Electric Reliability Corporation (NERC) (U.S. Environmental Protection Agency (EPA), 2015b). The damages associated with electricity generation are calculated by multiplying the actual CAP emissions from each fossil fuel power plant and the marginal damages of CAPs in the counties where the fossil fuel power plant is located. Tong (2016) did not include biodiesel. We thus assume biodiesel (B100) reduces life cycle GHG emissions by 50% compared to conventional diesel (U.S. Environmental Protection Agency (EPA), 2010), but have the same air pollution damages as that of conventional diesel due to a lack of recent literature. More research may be needed to clarify biodiesel's air pollution damages.

2.4. Criteria air pollutant emissions in hotspot areas

While literature has shown that air pollution costs vary within the county boundary, it is currently computationally impossible to estimate air pollution impacts with a grid size smaller than 10 km by 10 km. So we model CAP emissions from PAAC's bus fleets in hotspot areas in Pittsburgh, PA to estimate PAAC's contributions at a finer geographic scale than a county. There are currently no real-time emissions monitoring systems on mobile sources (including transit buses) due to the size and cost of monitoring devices. Instead, we calculate emissions based on vehicle operation emissions measured during bus tests and estimated bus fleet mileage in hotspot areas. The hotspot areas (shown in Fig. 1) include the Downtown, North Shore, Station Square, Bluff, and Oakland areas in Pittsburgh, PA.

The bus fleet mileage in hotspot areas are calculated as the total bus miles from all bus trips within hotspot areas over a calendar year. The bus mileage in hotspot areas for any bus route is calculated using ArcGIS software and bus route shapefile files (PAAC, 2016). Fig. 1 shows bus routes and bus stops in the hotspot areas. The number of bus trips for any bus route in a calendar year is calculated using bus schedule files (General Transit Feed Specification (GTFS) files) (PAAC, 2016). In this analysis, we do not account for planned and unplanned bus service changes during holidays.

3. Results

3.1. Life cycle ownership costs

We consider two cases for life cycle ownership costs, one where external funding that pays for 80% of bus purchase costs is available, and the other where external funding is not available. We note that external funding (such as FTA funding) can have other competing uses, such as retrofitting existing buses and upgrading bus garages, so its availability for bus purchases may be less than assumed. However, upon our communication with PAAC, external funding is currently sufficiently available.

Fig. 2 shows life cycle ownership and external costs (i.e. the sum of life cycle ownership costs and life cycle external costs) as well as cost breakdowns for 40-foot and 60-foot transit buses. We find that the availability of external funding is crucial for transit agencies to adopt any alternative fuel option. Without external funding, conventional diesel is among the cheapest in terms of both life cycle ownership costs and life cycle ownership and external costs. When external funding is available to reduce bus purchase costs by 80%, BEBs and HEBs become more cost-effective than convention diesel. In particular, life cycle ownership and external costs of BEBs are 17–23% lower than conventional diesel buses. Other bus options cost more than a conventional diesel bus in terms of life

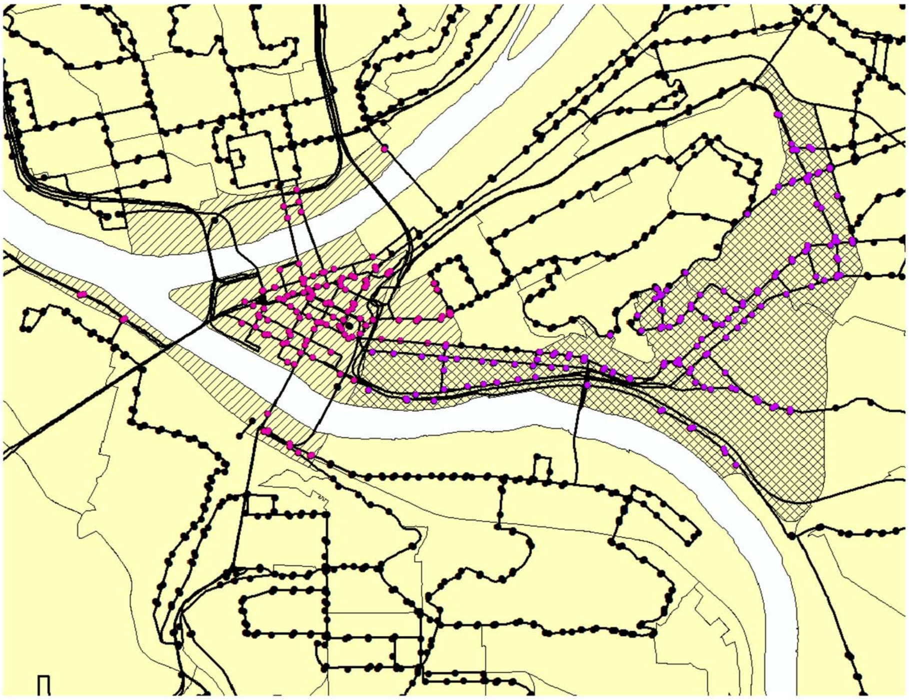

Fig. 1. PAAC's transit bus routes (black solid lines) and stops (pink and purple dots) in hotspot areas (shaded areas) in Pittsburgh, PA. The hotspot areas include the Downtown, North Shore, Station Square, Bluff, and Oakland areas in Pittsburgh, PA. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

cycle ownership and external costs (+1%, +2%, +5%, and +18% for B20, CNG, B100, and LNG respectively). The advantages of BEBs are their high vehicle efficiency, low electricity rates in PA, and low O & M costs as few mechanical devices and pollution control devices are needed. It is worth mentioning that BEBs see the largest impact from the external funding - either being the cheapest options or the most expensive options depending on the availability of external funding. This is because they have the highest vehicle purchase costs (capital expenditure) and the lowest vehicle operation costs.

A 60-foot bus is more capital-intensive and has a lower fuel economy than its 40-foot counterpart, but it carries more riders during one trip. When evaluating the two options in terms of ownership costs or ownership and external costs, a 60-foot bus is more expensive than a 40-foot bus. When external funding is available, the rank of technology options is similar to that of the 40-foot transit bus (except that diesel HEBs become relatively worse). In this case, BEBs reduce life cycle ownership and external costs by 11–18% compared to conventional diesel. The rank of technology options remains unchanged compared with that of 40-foot buses when external funding is not available. However, the relative cost performance of alternative fuel buses are worse for the 60-foot buses compared to 40-foot buses. We believe two reasons collectively explain this phenomenon. First, 60-foot transit buses face some unfavorable conditions compared to 40-foot buses – they are relatively more expensive because of a smaller demand; they have worse fuel efficiency because of heavier weight; and they have lower annual mileage as they are used less often on weekends and holidays. Second, the metric used ($/bus/year) does not account for the additional service provided by 60-foot transit buses compared to 40-foot buses. Alternative metrics such as passenger-miles and seat-miles may favor 60-foot transit buses. While 60-foot transit buses are more valuable in rush hours, they are less valuable in non-rush hours.

3.2. Factors that change the ranks of alternative fuel options

Fig. 3 shows sensitivity analysis results of alternative fuel technologies at higher diesel prices, lower annual bus mileage, higher electricity rates, higher infrastructure costs, and higher discount rates. Table 3 explains these sensitivity scenarios in detail. We consider these five factors because they are uncertain and are likely to impact the ranks of transit bus technologies (especially those between BEBs and conventional diesel buses). For each of the five factors, we determine a likely value different from the baseline assumption. Higher diesel price is chosen because current diesel price is at a decade-low point (U.S. Energy Information Administration (EIA), 2016a). We consider reduced annual mileage because alternative fuel buses may have less due to lower vehicle range. Electricity rate is doubled because the electricity rate in Pittsburgh one of the lowest in the country, where the state-average electricity rate to transportation customers are $0.046–0.19 $/kWh (U.S. Energy Information Administration (EIA), 2016b). The infrastructure cost is doubled or reduced by half to examine the impact of both underestimates and overestimates. Finally, we test a higher discount rate because the current discount rate suggested by U.S. Office of Management and Budget (OMB) is historically low (U.S. OMB (2015)). We then run the sensitivity analysis holding all other assumptions unchanged.

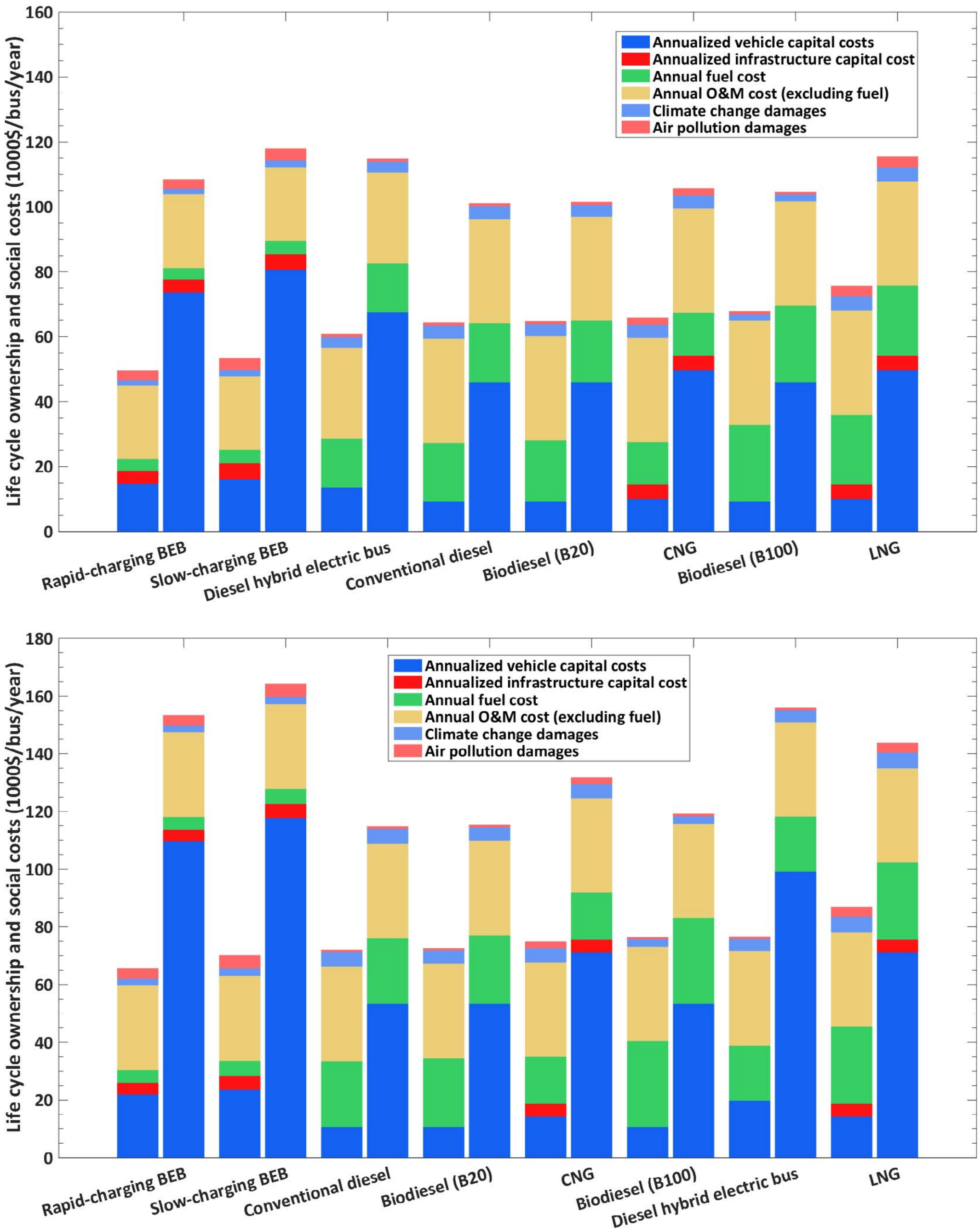

Fig. 2. Annualized life cycle ownership and external costs for a 40-foot transit bus (top) and a 60-foot bus (bottom). In each figure, left bars assume reduced vehicle purchase costs (80% paid by external funding) and right bars consider full vehicle purchase costs without external funding. The project lifetime is assumed to be the same as the lifetime of a bus (12 years) and we assume 1% discount rate. External costs include climate change damages (using 100-year global warming potential (GWP)) and air pollution damages (using AP2 model).

We find that all five factors, independently or jointly, do not change our conclusions that BEBs achieve large reductions in ownership and external costs compared to conventional diesel, when external funding is available. Higher diesel price is more important than reducing infrastructure cost to achieve cost savings from alternative fuel technologies. When external funding is not available, we find that lower annual mileage have higher impacts on the life cycle cost differences between BEBs and diesel than the other factors (higher discount rate, higher electricity rate, and higher infrastructure cost). When these conditions happen together, BEBs lead to 25–36% higher costs than diesel. It is also worth mentioning that slow-charging BEBs always lead to the highest costs across all technology options when external funding is not available. This highlights the heavy burden of high capital expenditure on purchasing slow-charging BEBs for transit agencies. In this sense, the availability of external funding not only lowers the life cycle

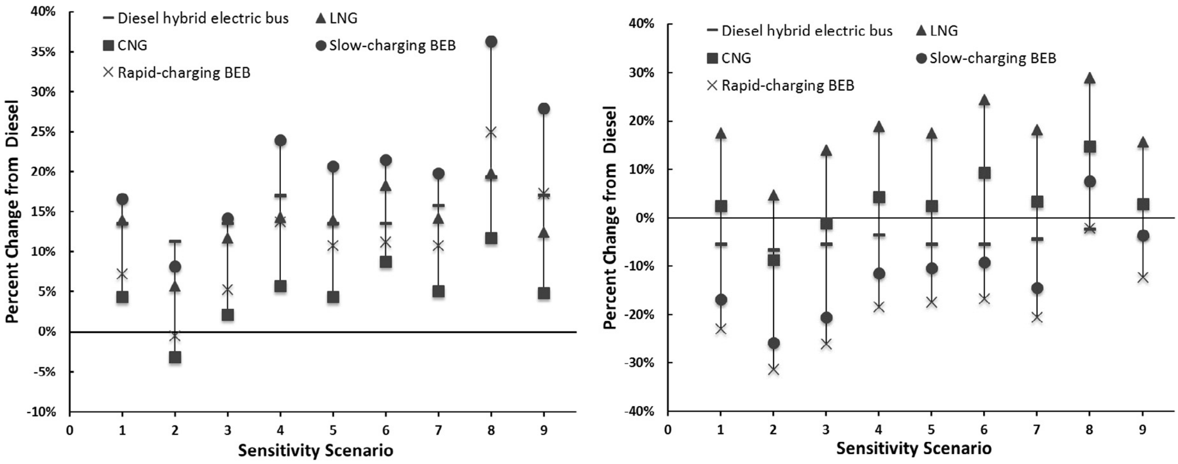

Fig. 3. Sensitivity analysis results for 40-foot transit buses without external funding (left) and with external funding (right). Percentages are calculated as differences between life cycle ownership and external costs of alternative fuel options and conventional diesel. Negative percentages mean alternative fuel options reduce ownership and external costs than diesel.

Table 3
Scenario descriptions for sensitivity analysis. Baseline assumptions are used unless otherwise stated.

Scenario	Assumptions
1 – Baseline	Annual mileage of 37,761 miles/year and 1% discount rate
2 – Higher diesel price	Diesel price $1/gallon higher the baseline
3 – Reduced infrastructure cost	Assuming 50% less infrastructure cost for the same capacity
4 – Reduced annual mileage	Annual mileage reduced to 30,000 miles/year
5 – Doubling electricity price	Double electricity price from the baseline
6 – Doubling infrastructure cost	Double the per bus infrastructure cost from the baseline
7 – Higher discount rate	Increase discount rate to 3%
8 – Combine scenarios 4, 5, 6, 7	See above
9 – Combine scenarios 2, 4, 5, 6, 7	See above

ownership and external costs but also helps transit agencies better prepare for the unfavorable operating conditions that may happen during the lifetime of transit buses.

The diesel price is currently low due to a combination of strong supply and weak demand in global crude oil and refined product markets (U.S. EIA, 2016a). In the baseline scenario, we assume the diesel price to be $2.30/gal based on PAAC's data and recent diesel markets (U.S. DOE, 2016). We note, however, the large variability in diesel prices in the last decade (2007–2016), where diesel prices ranged between $2.00/gallon and $4.70/gallon (U.S. DOE, 2016). Because the conventional diesel bus serves as the baseline in our assessment, changes in diesel prices significantly affect the comparison between alternative fuel options. As the diesel price is currently at a decade-low point (U.S. EIA, 2016a), we expect the diesel price to rebound slightly back as global market adjusts towards equilibrium. In the sensitivity analysis, we consider a diesel price of $3.30/gallon. We note that higher diesel prices can happen in the future. At a diesel price of $3.30/gallon, rapid-charging BEBs and CNG buses achieve lower ownership and external costs compared to conventional diesel, with or without external funding. Our estimates show that this diesel price is not high enough to balance out all unfavorable conditions happening together (lower annual bus mileage, higher electricity rate, higher infrastructure costs, and higher discount rate) for BEBs (without external funding). Further analysis shows that the break-even diesel price is around $6.10/gal, a significantly higher diesel price to cancel out all of the unfavorable conditions for rapid-charging BEBs.

3.3. Life cycle external costs

For the baseline results (Fig. 2), we find that including life cycle externality does not change the rank of technologies. This is because these external costs are small compared to ownership costs. For 40-foot buses, the ratio between external costs and ownership costs fall between 3% and 7% (without external funding), or 5% and 12% (with external funding) - with biodiesel and conventional diesel on the lower end and LNG and BEBs on the higher end. A similar pattern exists for 60-foot buses although the range of ratios becomes 3–7% (without external funding) or 5–13% (with external funding). Nevertheless, technology assessments that ignore environmental externality are incomplete because these are actual costs paid by people not just the emitter.

If we limit the scope to include only external costs, we find that biodiesel (B100 and B20) and diesel HEBs have lower costs compared to conventional diesel for both 40-foot and 60-foot buses (Fig. 4 and Appendix A). Higher fuel efficiency of HEBs reduces energy consumption and the associated emissions to power one vehicle mile traveled. On the other hand, LNG, CNG and slow-

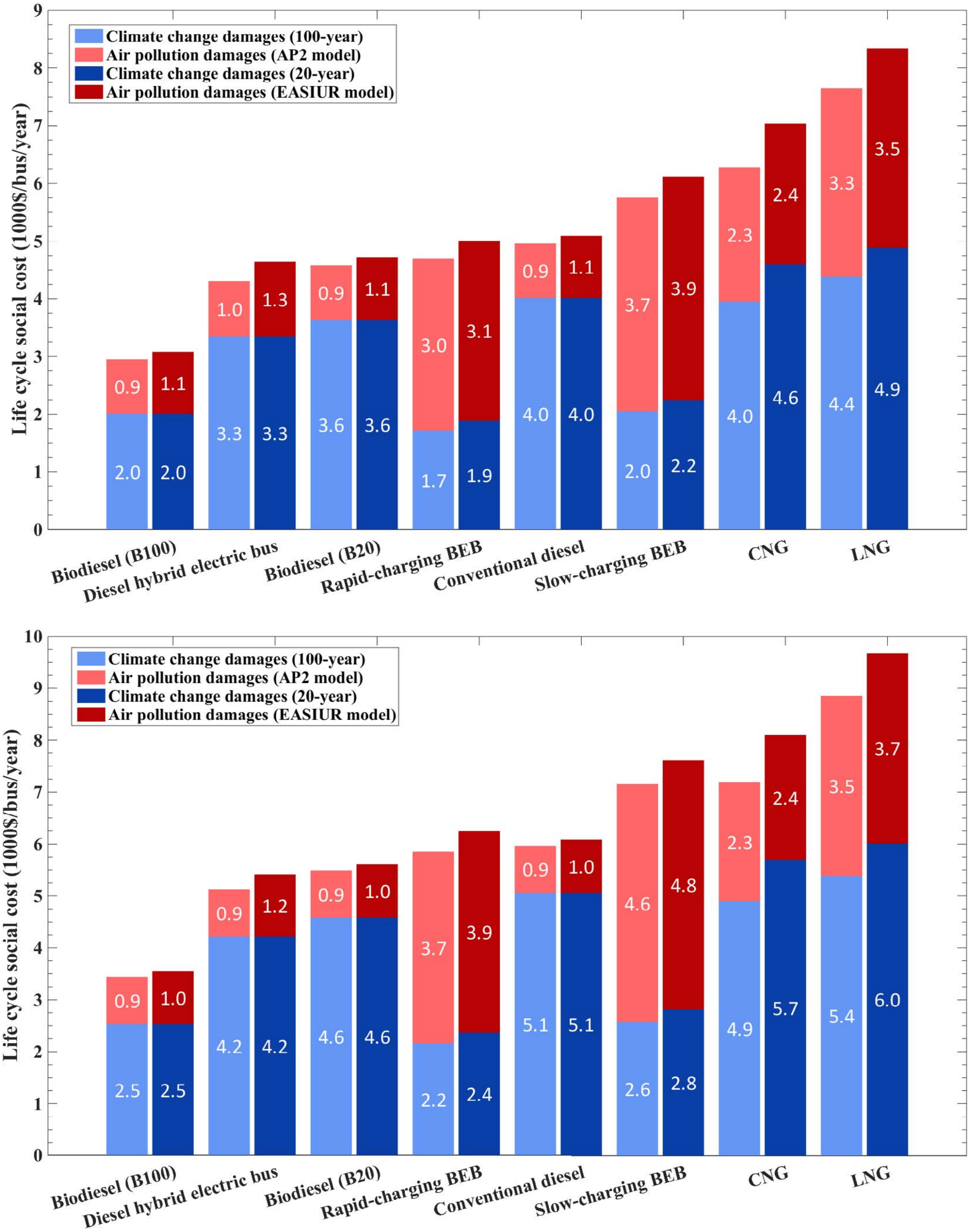

Fig. 4. Life cycle external costs for a 40-foot (top) and a 60-foot transit bus (bottom). Left bars represent climate change damages (based on 100-year time horizon) and air pollution damages (based on AP2 model). Right bars represent climate change damages (based on 20-year time horizon) and air pollution damages (based on EASIUR model). The left bars and right bars show lower bounds and upper bounds of external costs using different time horizons of global warming potential (GWP) and criteria air pollutant (CAP) marginal damage models.

charging BEBs have higher external costs than conventional diesel, and LNG 60-foot buses more than double external costs of conventional diesel. LNG and CNG buses have large externality because they have significantly higher air pollution damages than other technology options. Although these natural gas buses reduce tailpipe SO_2, NO_x, and VOC emissions, they emit very high tailpipe CO emissions (Table A3) and also have high air pollution damages associated with compression or liquefaction (Table A6). Both compression or liquefaction are intensive in electricity use, which lead to high air pollution damages because electricity generated in the Mid-Atlantic region emits large SO_2 and NO_x emissions and can affect large population through dispersion (Tables A7 and A8). In addition, LNG buses also have higher GHG emissions than conventional diesel, contributing further to high external costs (Tables A1

Table 4
Estimated criteria air pollutant (CAP) emissions from PAAC's bus fleet in the hotspot areas in 2015. Unit: metric ton/year. Note only emissions directly from vehicle operation are included. Emissions proxies (*) are calculated assuming the whole bus fleet is composed of new buses. N/A means not available.

Scope	PAAC all					Hotspot areas				
	$PM_{2.5}$	SO_2	NOx	VOC	CO	$PM_{2.5}$	SO_2	NOx	VOC	CO
Existing fleet	2.7	N/A	135	N/A	N/A	0.3	N/A	13.7	N/A	N/A
New diesel*	0.9	0.4	24.8	3.0	13.2	0.1	0.04	2.5	0.3	1.3
New diesel HEBs*	0.9	0.3	39.0	2.1	5.0	0.1	0.03	4.0	0.2	0.5
New CNG*	0.9	0.3	15.6	1.9	844	0.1	0.03	1.6	0.2	86.0
New LNG*	0.9	0.0	15.6	1.9	844	0.1	0.00	1.6	0.2	86.0
New BEBs*	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0

and A2). Finally, rapid-charging BEBs have lower external costs than diesel but slow-charging BEBs have higher external costs than diesel. These comparisons are largely driven by the externality from electricity generation in the Mid-Atlantic region as well as those from tailpipe GHG and CAP emissions of diesel buses operating in Pittsburgh.

3.4. Criteria air pollutant emissions in hotspot areas

PAAC currently operates 100 bus routes including 2 temporary routes to make up for reduced light rail service. 83 of these 100 bus routes serve Downtown Pittsburgh, and 89 bus routes serve either Downtown or the Oakland area. Over a calendar year, these 89 routes make more than 900,000 bus trips, or 94% of all PAAC's bus trips, in the hotspot areas (Downtown and Oakland). The bus fleet mileage within hotspot areas is 2.7 million miles per year, or roughly 10% of PAAC's total bus mileage. The actual emissions in hotspot areas are calculated using fleet mileage in hotspot areas and weighted-average emissions factors of PAAC's bus fleet (Bradley and Associates LLC, 2014). We find that PAAC's bus fleet emitted 135 metric tons of NO_x and 2.2 metric tons of $PM_{2.5}$ in 2015 (Table 4). Around 10% of these emissions happened in hotspot areas.

To compare emissions reduction potential of alternative fuel options, we calculate an emissions proxy using emissions factors of new buses. In other words, the emissions proxy represents emissions if the whole bus fleet is composed of new buses. Although this is an unlikely scenario, without referring to a complex bus turnover model, the emissions proxy should help identify relative benefits of alternative fuel options. Table 4 shows that BEBs can eliminate all tailpipe emissions (but still have $PM_{2.5}$ emissions from break and tire wear), achieving the largest emissions reduction potential of all technologies considered. Diesel HEBs reduce SO_2, VOC, and CO emissions but increase NO_x emissions by 50% relative to new diesel buses. LNG and CNG buses reduce SO_2, NO_x, and VOC emissions but increase CO emissions significantly by a factor of 64!

Michanowicz et al. (2012) estimated that 224 tons of $PM_{2.5}$ were emitted from mobile sources in Allegheny County in 2009 and 43% (or 96.3 tons) came from diesel vehicles. Thus PAAC's bus fleet only contributes to slightly more than 1% of $PM_{2.5}$ emissions from all mobile sources in Allegheny County. However, it is worth noting that reduction of $PM_{2.5}$ emissions is important to human health. Literature shows that diesel particulate matter (DPM) is the leading additive cancer risk air toxic in Downtown Pittsburgh and in Allegheny County (Michanowicz et al., 2013). Thus alternative fuels (CNG, LNG, and BEBs) have the added benefit of reducing cancer risk by replacing diesel buses in Downtown Pittsburgh and in Allegheny County.

4. Discussion

In this paper, we estimated life cycle ownership and external costs for alternative fuel options, and estimated CAP emissions from PAAC's bus fleet in hotspot areas. If external funding is available, purchasing and operating BEBs results in significant savings compared to diesel buses. We find that rapid-charging BEBs achieve lower costs than slow-charging BEBs due to double dividends of smaller batteries used in rapid-charging BEBs. The battery replacement costs are smaller, and rapid-charging BEBs are lighter in weight, thus achieving better fuel efficiency.

4.1. Regional variations

We emphasize that the results and findings are limited by the assumptions we have made. As we have mentioned in the Method section, some PAAC-specific assumptions, such as electricity rates and emissions associated with grid electricity vary from region to region. Performing the same assessments with region-specific electricity-related assumptions may yield different conclusions. For instance, average electricity rates across utilities are \$0.08–0.28/kWh for slow-charging and \$0.14–0.44/kWh for fast-charging in California (CARB, 2016b). These electricity rates are significantly higher than electricity rates in Pittsburgh, PA (\$0.055/kWh), because utilities in CA have demand charges and dynamic pricing.

We expanded our sensitivity analysis to test impacts of these electricity rates. If external funding is available, rapid-charging BEBs still have lower ownership and external costs than diesel for an electricity rate as high as \$0.27/kWh (five times higher than the baseline electricity rate in Pittsburgh). Further, when demand charges and dynamic pricing are in place, slow-charging BEBs may result in lower ownership and external costs than rapid-charging BEBs, because slow-charging BEBs can take advantage of lower electricity rates.

The electricity grid of the Midwest/Mid-Atlantic region where Pittsburgh is located has the largest share of coal-fired power plant plants in the country (U.S. EIA, 2016b). So other regions could find that BEBs achieve lower external costs than conventional diesel if electricity grids in those regions are cleaner.

Finally, fuel economy assumptions and tailpipe GHG and CAP emissions from vehicle operation may also vary across region, because of varying factors such as speed, weight, road grade, and weather (Alam and Hatzopoulou, 2014; Reyna et al., 2015; Yuksel and Michalek, 2015). Indeed, the measured fuel economy values from Altoona Bus Research and Testing Center (2016) may not apply for extreme weather conditions. A previous study has identified large variations in fuel economy of light-duty vehicles under extreme weather (Yuksel and Michalek, 2015). Similar studies on transit buses are needed when there are more data.

4.2. Practical challenges for BEBs

While BEBs are estimated to have the lowest life cycle ownership and external costs, both types of BEBs face practical challenges to immediate operation for a typical bus route. First, BEBs have limited ranges (33–41 miles for rapid-charging BEBs and 104–130 miles for slow-charging BEBs), which are significantly smaller than other bus technologies (Table 2), and would demand special routes or specialized planning and scheduling. Indeed, rapid-charging BEBs require tight control of bus schedules to ensure a bus is charged at a specific bus stop at a specific time. Even though buses are operated on a planned schedule, the actual schedule is determined by traffic congestion, weather and other road factors. As a result, bus routes on dedicated bus lanes or fixed busways may be more feasible for rapid-charging BEBs. Additionally, BEBs require dedicated charging infrastructure, which, in addition to higher capital expenditures and O & M costs, require land to install and coordination with local utilities. Finally, charging infrastructure for BEBs is currently not compatible among bus manufactures.

4.3. Favorable trends for BEBs

We identify several trends that make BEBs more attractive in the near future. BEBs will become more technologically mature as more buses are delivered and operated across the country. The costs of batteries are declining rapidly while the performance is improving quickly (Nykvist and Nilsson, 2015) due to increased battery deliveries in light-duty vehicle markets. Thus, future BEBs will have better economics and longer range.

Equally important are federal and state energy policies such as U.S. EPA's Clean Power Plan (CPP) and state-level Renewable Portfolio Standards (RPS) (National Conference of State Legislatures, 2016; U.S. Environmental Protection Agency (EPA), 2016c). They will lead to more renewable energy sources and less coal-fired power plants in U.S. electricity grids in the next two decades. In particular, U.S. EIA (2016b) projected a 26% decline in direct CO_2 emissions from the electricity grid in the Midwest/Mid-Atlantic region from 2015 to 2030 (in the reference case in Annual Energy Outlook 2016) as a result of a more than 40% reduction in coal-fired electricity generation during the same period. Since coal-fired power plants also have high CAP emissions, we expect a similar reduction in direct CAP emissions from the electricity grid. If we assume a 26% reduction in external costs from grid electricity in the Midwest/Mid-Atlantic region, and assume conventional diesel's external costs remain the same over the next 15 years, then BEBs in 2030 will result in lower life cycle external costs than conventional diesel. If we further consider battery and other technology improvements, BEBs advantages will be even larger.

Finally, we note that BEBs are easier to integrate with intelligent control technologies. For instance, BEBs already have the capability to communicate key information (such as battery's state of charge (SOC) and GPS locations) remotely to a control room to facilitate scheduling, charging, and operation (New Flyer, 2016b). In the future, sensing and communication capacities of BEBs could help build a smart transportation system where connected and automated vehicles dominate.

4.4. Uncertainty in externality estimates

While we have used the most recent data to build emissions inventories and used state-of-art marginal damage estimates of GHGs and CAPs, we emphasize that there are high uncertainties in externality estimates due to conflicting emissions estimates and evolving scientific understandings of health and environmental impacts of GHGs and CAPs. First, Tong (2016) found that upstream (well-to-pump) air pollution costs from petroleum fuels would increase by a factor of 4 using GREET model's emissions data rather than using U.S. EPA's NEI (used in this paper), and life cycle air pollution costs of diesel buses increase by 87%. However, because of the relatively low ratios between external costs and ownership costs, using alternative externality estimates does not change the ranking of fuel options in terms of ownership and external costs. Second, the SCC has a large range of estimates from a few dollars to hundreds of dollars per metric ton of CO_2 emission. The two-order-of-magnitude difference is mainly due to different assumptions regarding discount rate and climate change damage functions (U.S. NRC, 2010). However, even at a 10-times-larger SCC ($410/metric ton CO_2), the only substantive changes in conclusions are BEBs and biodiesel buses achieve lower life cycle ownership and external costs than conventional diesel buses without external funding. Similar sensitivity analyses for other regions and for other vehicles are available in Tong et al. (2017). Third, CAPs' social damage estimates do not include all known health impacts due to data and methodological issues. In particular, currently available marginal damage estimates of VOCs and CO are likely to be underestimates (Tong, 2016), and cancer risks of diesel particulate matter are not monetized at all (Michanowicz et al., 2013). Furthermore, current estimates of CAPs' social damages cannot go smaller than a 10-km-by-10-km resolution, which is still too large to accurately characterize CAPs' damages.

4.5. Policy implications

The assessment on alternative fuel options for transit buses indicates that BEBs are promising technology options. While BEBs were not included in previous assessments, they exhibit high fuel efficiency, zero tailpipe emissions, and low life cycle ownership and external costs. BEBs should attract attention and strong interest from transit agencies, bus manufacturers, and public officials who want to maximize public interest. We note that some alternative fuel options, such as CNG and LNG buses and BEBs, have strong lock-in effects because of the refueling/charging and supporting infrastructure required. It is unlikely that any transit agency can operate more than one of these alternative fuel options given the limited financial, human, and land resources. We thus recommend transit agencies to consider both the short-term and long-term perspectives when purchasing new vehicles. This forward-looking and long-term vision is particular important as the transportation systems and mobility services are likely to undergo a large change.

Any transit agency that plans to operate BEBs should prepare for changes in planning and scheduling, operation and maintenance, fuel procurement, and supporting infrastructure. As HEBs have already been widely used across the U.S., transit agencies have gained experiences in maintaining and calibrating batteries and in operating buses that share some similar technologies with BEBs. These experiences will help transit agencies prepare for operating BEBs. As discussed previously, BEBs could also help transit agencies adopt intelligent technologies and fit into the future intelligent transportation systems that are likely to happen.

Our paper extends the framework and method of economic assessments on alternative fuel options by including life cycle external costs of unintended air emissions. While the inclusion of external costs does not change the rank of fuel options, it provides more accurate accounts of private and social impacts caused by transit buses. Furthermore, we highlight uncertainty and methodical limitations of state-of-the-art damage function approaches and point out potential research directions. We also estimate emissions from bus fleets in hotspot areas to show the implications of high-resolution emissions estimates. We believe that this updated framework of life cycle ownership and external costs will help transit agencies and other interested audiences to determine the best alternative fuel option, and to maximize private and social net benefits.

Acknowledgement

This research was made possible through support from the Center for Climate and Energy Decision Making (CEDM), and the Richard King Mellon Foundation. The CEDM has been created through a cooperative agreement between National Science Foundation (SES-0949710) and Carnegie Mellon University. Fan Tong thanks the support from 2013-14 Northrop Grumman Fellowship, 2013-14 Steinbrenner Institute Graduate Research Fellowship, and 2016 Ji Dian Liang Fellowship. We thank Port Authority of Allegheny County for its collaborative sharing of system information. We also thank the anonymous reviewers for their insightful comments. Any opinions, findings, and conclusions expressed in this material are those of the authors and do not necessarily reflect the views of these organizations.

Appendix A

A.1. Life cycle climate change damages

We rely on Tong et al. (2015) to estimate life cycle GHG emissions from diesel and alternative fuel transit buses, as shown in Tables A1 and A2. Climate change damages are then estimated using formula (3).

A.2. CAP emissions estimates and marginal damages

Life cycle external costs are calculated using formulae (3)–(5) in Section 2.3. Key inputs for formulae (4) and (5) are presented in Tables A3–A5. In addition, external costs of CAP emissions from battery manufacturing are $\$_{2015}$ 9/kWh, and marginal damages of CO are $\$_{2015}$ 808/metric ton. See Tong (2016) for details. Table A6 reports life cycle external costs due to CAP emissions.

Table A1
Life cycle climate change damages of diesel and alternative fuel options for 40-foot transit buses. Unit: $\$_{2015}$/year/bus.

	Diesel	Diesel HEB	CNG	LNG	Rapid-charging BEB	Slow-charging BEB
100-year global warming potential (GWP)						
Battery manufacturing	0	1	0	0	16	59
Upstream (Well-to-refueling station)	768	640	1215	1520	1702	1991
Operation	3249	2708	2738	2862	0	0
Total (Well-to-wheel)	4017	3349	3954	4382	1719	2050
20-year global warming potential (GWP)						
Battery manufacturing	0	1	0	0	16	59
Upstream (Well-to-refueling station)	768	640	1857	2024	1870	2186
Operation	3249	2708	2738	2862	0	0
Total (Well-to-wheel)	4017	3349	4596	4886	1886	2246

Table A2
Life cycle climate change damages of diesel and alternative fuel options for 60-foot transit buses. Unit: $\$_{2015}$/year/bus.

	Diesel	Diesel HEB	CNG	LNG	Rapid-charging BEB	Slow-charging BEB
100-year global warming potential (GWP)						
Battery manufacturing	0	1	0	0	16	60
Upstream (Well-to-refueling station)	968	806	1509	1888	2146	2509
Operation	4094	3412	3391	3499	0	0
Total (Well-to-wheel)	5062	4220	4901	5386	2162	2569
20-year global warming potential (GWP)						
Battery manufacturing	0	1	0	0	16	60
Upstream (Well-to-refueling station)	968	806	2307	2514	2357	2756
Operation	4094	3412	3391	3499	0	0
Total (Well-to-wheel)	5062	4220	5698	6012	2373	2815

Table A3
Vehicle operation CAP emissions from transit buses. Unit: gram/mile. Due to data availability, we assume CAP emissions from vehicle operation are the same for a 40-foot and a 60-foot transit bus.
Source: Tong (2016).

	Diesel	Diesel HEB	CNG	LNG	BEB
$PM_{2.5}$	0.0335	0.0335	0.0335	0.0335	0.0124
SO_2	0.0160	0.0114	0.0093	0.0000	0.0000
NO_x	0.9175	1.4450	0.5775	0.5775	0.0000
VOC	0.1121	0.0787	0.0695	0.0695	0.0210
CO	0.4900	0.1850	31.2750	31.2750	0.0000

Table A4
Marginal damages of CAP emissions from ground-level sources in Allegheny County, PA. Unit: $\$_{2015}$/metric ton.
Source: Heo et al. (2016a, 2016b), Muller and Mendelsohn (2007), and Muller (2011).

	$PM_{2.5}$	SO_2	NO_x	VOC
AP2 model	$270,596	$84,823	$5422	$25,912
EASIUR model	$272,885	$27,439	$13,309	N/A

Table A5
Marginal damages of CAP emissions from upstream activities of fuel pathways used in Allegheny County, PA. Unit: $\$_{2015}$/MJ.
Source: Tong (2016).

	Diesel[a]	CNG[b]	LNG[b]	Electricity[c]
AP2 model				
Energy/feedstock production and transportation	0.01	0.01	0.01	0.01
Fuel production and transportation	0.01	0.05	0.13	1.18
Upstream (Well-to-refueling station) total	0.02	0.06	0.14	1.19
EASIUR model				
Energy/feedstock production and transportation	0.01	0.01	0.01	0.01
Fuel production and transportation	0.01	0.05	0.14	1.24
Upstream (Well-to-refueling station) total	0.02	0.06	0.15	1.25

Note:
[a] External costs due to air emissions from diesel are estimated for U.S.-average diesel due to data availability.
[b] External costs due to air emissions from CNG and LNG are estimated for Allegheny County, PA where electricity used to compress or liquefy natural gas is assumed the average electricity delivered in the RFC region (U.S. Environmental Protection Agency (EPA), 2015b).
[c] Social damages from electricity are estimated for average electricity delivered in the RFC region.

A.3. Electricity grid emissions and the resulting external costs

Tables A7 and A8 report weighted-average CAP and GHG emissions and the resulting external costs per unit of electricity produced by NERC region. As discussed in the main text, we assumed the electricity is balanced in each NERC region.

Table A6
Life cycle external costs due to CAP emissions of diesel and alternative fuel transit buses. Unit: $\$_{2015}$/year/bus.

	Diesel	Diesel HEB	CNG	LNG	Rapid-charging BEB	Slow-charging BEB
AP2 model						
Battery manufacturing	0	6	0	0	99	363
Energy production and transportation	83	69	142	142	23	26
Fuel production	150	125	584	1557	2727	3189
Operation	708	758	1596	1566	127	127
Total (Well-to-wheel)	941	958	2322	3266	2975	3705
EASIUR model						
Battery manufacturing	0	6	0	0	99	363
Energy production and transportation	85	71	143	143	23	26
Fuel production	151	126	615	1640	2872	3358
Operation	840	1090	1683	1673	128	128
Total (Well-to-wheel)	1075	1292	2441	3456	3121	3875

Table A7
Weighted-average CAP and GHG emissions factors by NERC region. Unit: gram/MWh. Only direct emissions from power plants are accounted for.

Air pollutant	$PM_{2.5}$	SO_2	NO_x	CO_2
FRCC	82	348	232	492
MRO	57	942	577	601
NPCC	29	92	92	242
RFC	70	1093	452	523
SERC	54	679	337	499
SPP	43	840	472	645
TRE	39	671	221	523
WECC	38	178	317	387
Contiguous U.S.	53	652	350	484

Note: NERC region definition is available in U.S. Environmental Protection Agency (EPA) (2015b).

Table A8
Weighted-average air pollution damages and climate change damages by NERC region. Unit: $\$_{2015}$/MWh. Only direct emissions from power plants are accounted for.

	AP2 model				EASIUR model				Climate change damages
Air pollutant	$PM_{2.5}$	SO_2	NO_x	CAP total	$PM_{2.5}$	SO_2	NO_x	CAP total	CO_2
FRCC	3.2	9.3	0.5	13.0	5.4	5.2	0.8	11.4	20.3
MRO	1.3	25.2	3.9	30.4	3.1	17.2	4.2	24.4	24.7
NPCC	4.1	2.8	0.1	7.0	5.6	2.6	1.8	10.0	9.9
RFC	4.4	50.2	1.9	56.4	7.3	25.7	4.3	37.3	21.5
SERC	2.1	22.7	1.6	26.3	3.5	12.0	1.5	17.0	20.5
SPP	0.9	15.9	3.0	19.8	2.0	12.1	1.9	15.9	26.5
TRE	1.0	14.5	1.2	16.7	2.0	9.4	0.6	12.0	21.5
WECC	0.9	3.0	1.5	5.5	1.1	1.6	0.5	3.2	15.9
Contiguous U.S.	2.4	22.5	1.7	26.5	3.9	12.4	2.0	18.3	19.9

Note: NERC region definition is available in U.S. Environmental Protection Agency (EPA) (2015b).

Appendix B. Supplementary material

Supplementary data associated with this article can be found, in the online version, at http://dx.doi.org/10.1016/j.trd.2017.09.023.

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