Trends in onroad transportation energy and emissions

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ABSTRACT
Globally, 1.3 billion on-road vehicles consume 79 quadrillion BTU of energy, mostly gasoline and diesel fuels, emit 5.7 gigatonnes of CO$_2$, and emit other pollutants to which approximately 200,000 annual premature deaths are attributed. Improved vehicle energy efficiency and emission controls have helped offset growth in vehicle activity. New technologies are diffusing into the vehicle fleet in response to fuel efficiency and emission standards. Empirical assessment of vehicle emissions is challenging because of myriad fuels and technologies, intervehicle variability, multiple emission processes, variability in operating conditions, and varying capabilities of measurement methods. Fuel economy and emissions regulations have been effective in reducing total emissions of key pollutants. Real-world fuel use and emissions are consistent with official values in the United States but not in Europe or countries that adopt European standards. Portable emission measurements systems, which uncovered a recent emissions cheating scandal, have a key role in regulatory programs to ensure conformity between “real driving emissions” and emission standards. The global vehicle fleet will experience tremendous growth, especially in Asia. Although existing data and modeling tools are useful, they are often based on convenience samples, small sample sizes, large variability, and unquantified uncertainty. Vehicles emit precursors to several important secondary pollutants, including ozone and secondary organic aerosols, which requires a multipollutant emissions and air quality management strategy. Gasoline and diesel are likely to persist as key energy sources to mid-century. Adoption of electric vehicles is not a panacea with regard to greenhouse gas emissions unless coupled with policies to change the power generation mix. Depending on how they are actually implemented and used, autonomous vehicles could lead to very large reductions or increases in energy consumption. Numerous other trends are addressed with regard to technology, emissions controls, vehicle operations, emission measurements, impacts on exposure, and impacts on public health.

Implications: Without specific policies to the contrary, fossil fuels are likely to continue to be the major source of on-road vehicle energy consumption. Fuel economy and emission standards are generally effective in achieving reductions per unit of vehicle activity. However, the number of vehicles and miles traveled will increase. Total energy use and emissions depend on factors such as fuels, technologies, land use, demographics, economics, road design, vehicle operation, societal values, and others that affect demand for transportation, mode choice, energy use, and emissions. Thus, there are many opportunities to influence future trends in vehicle energy use and emissions.

Introduction
Over the last 100 years, the automobile was responsible for major social and economic trends in land use and personal mobility. U.S.-registered on-road vehicles grew from 8,000 in 1900 to 268 million in 2015 (Bureau of Transportation Statistics [BTS] 2016; Federal Highway Administration [FHWA] 1997). From 1950 to 2016, U.S. road miles increased by 25% whereas vehicle miles traveled (VMT) increased by 690% (Figure 1). Increases in vehicle fuel economy have helped to offset growth in vehicle energy use. As a response to the 1973 oil embargo (U.S. Department of State 2018), the 1975 Energy Policy and Conservation Act established Corporate Average Fuel Economy (CAFE) standards that were effective with the 1978 model year (U.S. Congress 1975). In 2009, the U.S. Environmental Protection Agency (EPA) issued an endangerment finding for six greenhouse gases (GHGs) because of their contribution to climate change (EPA 2009b). The U.S. Department of Transportation (USDOT) and EPA issued new fuel economy and tailpipe exhaust emission standards for carbon dioxide (CO$_2$), effective for the 2012 model year for light-duty vehicles (LDVs) and effective with the 2014 model...
What are the historic trends in on-road vehicle energy consumption and emissions? What are the current trends in on-road vehicle energy consumption and emissions? What are real-world on-road fuel economy and emissions consistent with fuel economy and emissions regulations, respectively? What are the emerging challenges and opportunities related to on-road vehicle energy consumption and emissions? What are the advances in measurement and monitoring of on-road vehicle energy consumption and emissions, and what new capabilities do these advances enable? What are the impacts of on-road transportation emissions on public health?

Because the United States has long had the world’s largest vehicle fleet, these questions are answered mostly in the context of the U.S. situation. However, different fleet mixes, regulatory programs, and rates of motorization enable comparative assessments based on an international perspective. Therefore, key lessons from other parts of the world, particularly from Europe and China, are also addressed. Transportation includes highway vehicles, non-road vehicles (e.g., construction, farm, and industrial equipment, lawn and garden equipment, recreational vehicles), rail, pipelines, waterborne vessels, and aviation. The focus here, however, is on highway vehicles. Background information on selected topics, such as postcombustion control technologies, is provided for the general reader.

Supplemental materials (SM), which are available at http://dx.doi.org/10.1080/10962247.2018.1454357, provide more details regarding factors affecting travel demand, vehicle energy consumption, vehicle emissions, methods for measuring emissions, and the impact of on-road vehicles on human health and pollutant exposures.

This review focuses on answering the following key technical and policy questions regarding trends in on-road transportation energy and emissions, motivated by factors identified in the preceding discussion:

- What are the historic trends in on-road vehicle technology, fuel efficiency, and emissions?
- What are the current trends in on-road vehicle technology and operation that affect energy use and emissions?
- Are real-world on-road fuel economy and emissions consistent with fuel economy and emissions regulations, respectively?
- What are the emerging challenges and opportunities related to on-road vehicle energy consumption and emissions?
- What are the advances in measurement and monitoring of on-road vehicle energy consumption and emissions, and what new capabilities do these advances enable?
- What are the impacts of on-road transportation emissions on public health?

National and global energy use and emissions

This section begins with definitions regarding types of vehicles and an overview of contextual issues, followed by an assessment of trends in energy consumption, the global on-road fleet, the U.S. on-road fleet, vehicle miles traveled, scenario-based projections of future energy consumption, the impact of autonomous vehicles on future energy use,
trends in GHG emissions, and trends in criteria pollutant and air toxic emissions.

Highway vehicle type terminology

EPA’s Motor Vehicle Emissions Simulator (MOVES) source types include motorcycles (MCs), passenger cars (PCs), passenger trucks (PTs), light commercial trucks (LCTs), three types of buses (intercity, transit, and school), refuse trucks, single-unit trucks, and combination trucks (EPA 2016a). The latter are further divided into short-haul and long-haul categories. EPA further divides the source types into regulatory classes based primarily on vehicle weight. However, the EPA classification scheme is not concerned with trailers nor with whether a vehicle is loaded or unloaded. Other often-used terms include light-duty gasoline vehicles (LDGVs), which refers to PCs and PTs fueled with gasoline, heavy-duty vehicles (HDVs), and heavy-duty diesel vehicles (HDDVs) (e.g., Denis and Lindner 2005; Robert et al. 2007).

Contemporary context

Although the U.S. market is saturated in terms of vehicle stock, the annual turnover of old vehicles leaving the fleet and new vehicles entering the fleet will change fleet characteristics over time (Bishop and Stedman 2014). More efficient gasoline engines and vehicle transmissions are enabling fuel economy improvements (EPA 2018a). Technologies introduced in the last decade or two, such as hybrid electric vehicles (HEVs), plug-in hybrid electric vehicles (PHEVs), battery electric vehicles (BEVs), fuel-cell electric vehicles (FCEVs), and alternative fuel vehicles (AFVs), are diffusing into the on-road fleet. Furthermore, the anticipated emergence of connected and autonomous vehicles could disruptively revolutionize trip-taking activity (Rubin 2016). Evolutionary improvements in truck design and operation are helping to improve fuel economy (EPA and USDOT 2016). Stringent emission standards and development and diffusion of new postcombustion emission controls for diesel engines, especially for NOx and PM emissions, have led to substantial reductions in real-world emissions (Sandhu and Frey 2012).

In Europe, the extent of motorization is not likely to increase substantially (Kuhnminhof, Zumkeller, and Chlond 2013). Vehicle energy and emission issues that are distinctively European are related, at least in part, to the much larger proportion (more than 50%) of light-duty diesel vehicles (LDDVs) in the LDV fleet in Europe compared to less than 1% LDDVs in the U.S. LDV fleet (EPA 2018a; International Council on Clean Transportation [ICCT] 2018). Diesel vehicles are more fuel-efficient than gasoline vehicles. European fuel prices are much higher than those in the United States, which provides an incentive for more fuel-efficient vehicles. However, diesel engines have characteristically higher engine-out emissions of NOx and PM than gasoline engines (Kalghatgi 2015). The United States regulates diesel and gasoline vehicles to the same emission limits, whereas in Europe there are different and, typically, more lax emission standards for diesel than gasoline vehicles. Furthermore, the regulatory framework in Europe differs from that in the United States in terms of test cycles and other factors (Nesbit et al. 2016). European LDV real-world fuel economy and emissions, especially for NOx, tend to deviate more from the official ratings and “type approval” certification values than is the case in the United States. In a recent survey, 90% of measured vehicles had real-world NOx emission rates higher than the Euro 6 limit (Baldano et al. 2017). The lack of conformity of real-world emissions with official values has fostered recent European Union (EU) efforts to quantify “real-driving emissions” (RDE) using portable emission measurement systems (PEMS) (EU 2018). China and many other countries adopt motor vehicle emission regulations based on the European approach, although China’s most recent “China 6” standard adopts best practices from both the United States and Europe (He and Liuhanzi 2017). Thus, challenges or issues associated with a particular regulatory framework are replicated in parts of the world.

Six countries, Brazil, China, Germany, Japan, Russia, and the United States, account for 50% of the global fleet, whereas 101 nations collectively account for less than 10% (Organisation Internationale des Constructeurs d'Automobiles [OICA] 2018). The vehicle fleet in China has quadrupled in size in one decade to become the second largest national fleet in 2015. Yet motorization in China, at 118 vehicles per 1000 people, lags far behind that of the United States. The global average motorization rate is 182 vehicles per 1,000 people.

Historical trends in transportation energy consumption

Globally, on-road transportation accounts for approximately 13% of total energy consumption, including 54.1 quadrillion BTU (Quads) for on-road passenger transport and 25.3 Quads for on-road freight transport in 2015 (Energy Information Agency [EIA] 2017d). Gasoline is the typical fuel for LDVs. Heavy-duty vehicles primarily use diesel (Davis, Williams, and Boundy 2017). Globally, gasoline fuel use has increased by about 33% from 1986 to 2014. Global diesel fuel use has increased by approximately 80% over the same period. Gasoline and diesel fuel use is either stable or declining in North America and Europe, but is
increasing substantially in Asia and other parts of the world, as shown in Figure 2.

Demand for transportation energy depends on many factors, including the number of vehicles and the annual VMT. In 2015, global new vehicle sales were 94 million, of which 28 million were in China and 18 million were in the United States (OICA 2018). In general, European and North American sales have rebounded to pre-recession levels, whereas sales in Asia are rapidly growing. In addition to China, sales are increasing substantially in India and Indonesia, among others.

**Global on-road vehicle fleet**

The global on-road vehicle fleet, including light- and heavy-duty vehicles, has increased by 44% from 2005 to 2015, to 1.282 billion vehicles in 2015 (OICA 2018). The average U.S. LDV, which weighs over 1,700 kg, is heavier than the average LDV in any other region of the world, but the share of heavier vehicles, including sport utility vehicles (SUVs), in sales is rapidly growing globally, with 30% increasing average weight in China in the last 10 years (Cuenot 2017). The vehicle population in China is projected to grow by 6% to 11% per year (Wang, Teter, and Sperling 2011). The global LDV stock, which was approximately 1.1 billion LDVs in 2015, is projected to grow to more than 2.2 billion LDVs by 2050 (GFEI 2016). In 2015, diesel accounted for only 0.9% of new U.S. LDVs, whereas 98.6% were gasoline fueled (EPA 2018a). In contrast, 52% of new European cars were diesel powered in 2015 (ICCT 2018).

**U.S. on-road vehicle fleet**

With 268.8 million vehicles as of 2016, the United States has the world’s largest vehicle fleet. As shown in Table 1, this fleet includes 192.8 million passenger cars, 54.9 million light-duty trucks, 8.7 million motorcycles, 8.7 million single-unit trucks, 2.8 million combination trucks, and 1.0 million buses (FHWA 2017a). Although the overall U.S. fleet is growing slowly, there is a long-term increase in the share of light-duty passenger trucks versus light-duty passenger cars. GHG emission standards impose target values that differ for PCs and PTs and vary by vehicle footprint, which manufacturers have flexibility to meet based on averaging over the model year fleet and accruing or using credits (EPA 2018b; NHTSA and EPA 2010).

**Vehicle miles traveled**

In the United States, although only 31% of lane miles are classified as urban, 70% of VMT occurs in urban areas. Annual U.S. VMT increased by over 100% from 1980 to 2015 (USDOT 2017). Rural VMT has increased by only 38%, versus 153% for urban VMT. Light-duty vehicles account for about 90% of U.S. VMT (FHWA 2017b). Motorcycles and buses each account for less than 1% of VMT. Heavy-duty vehicles account for less than 10% of VMT.

The principle means of transportation to work in the United States has been relatively steady from 1989 to 2015, with approximately 86% to 88% of workers commuting by automobile. Typically 76% of commuters drive themselves, while less than 10% carpool (USDOT 2017). Freight...
movements by truck in the United States have increased from 1.27 billion ton-miles in 1980 to 2.00 billion ton-miles in 2014 (USDOT 2017).

Over the past 30 years, U.S. VMT grew at an average rate of 2%/year. However, future VMT growth is expected to be more moderate. The U.S. FHWA projects VMT growth will average 0.66%/yr to 0.89%/yr, from 2015 to 2045. Thus, VMT is estimated to be higher in 2045 by 20% to 30% compared to 2015.

U.S. vehicle fuel economy has been improving, as illustrated in Figure 3 (EPA 2018a). Tailpipe exhaust CO$_2$ emissions are inversely proportional to fuel economy. In 1975, average fuel economy was 13.5 mpg for a car and 11.0 mpg for a truck-based SUV. In 2017, these averages were 30.0 mpg and 22.2 mpg, respectively, which translates into tailpipe exhaust CO$_2$ emission reductions of 55% and 50%, respectively. Improved powertrain efficiency and reduced tractive effort, related to reduced aerodynamic drag, lower rolling resistance, and lower frictional losses, are among the reasons for improving fuel economy (Thomas 2016). The promulgation of fuel economy standards has contributed to improved fuel economy in the United States, Europe, and Japan. However, especially in the United States, consumer preference is for larger vehicles (Clerides and Zachariadis 2008; Cuenot 2017), which offsets potential reductions in total fuel use.

**Future global road transportation energy consumption**

From 2015 to 2050, global energy consumption for road transport is projected to increase by 33%, from 79 Quads to 106 Quads (Figure 4). Within the Organization for Economic Cooperation and Development (OECD) nations, including the United States, Europe, and Japan, road transport energy consumption is projected to decrease by 17% for passenger transport because expected growth in VMT could be offset by increased fuel economy. OECD road freight energy consumption is projected to increase by 3%. In contrast, outside of the OECD, led by growth in China, India, Indonesia, and others, increases in energy consumption are estimated to be 125% for passenger vehicles and 36% for freight vehicles. Much of the global growth in energy consumption is attributed to LDVs outside of the OECD (EIA 2017c; International Energy Agency [IEA] 2017a).

Some of the expected key trends that will affect transport energy consumption in the coming decades include, but are not limited to, population growth, global economic growth, urbanization, policies to reduce LDDV usage in parts of Europe, increased vehicle electrification, changes in electric generation mix, energy efficiency (e.g., fuel economy), policies and standards, oil prices, increased availability of liquefied natural gas (LNG), and policies aimed at air pollution and GHG emissions (IEA 2017a; Tietge and Diaz 2017).

Although in 2005 there were only 1,370 electric drive vehicles (EDVs), including BEVs and PHEVs, globally, in 2016 there were 2,014,220 EDVs, of which 32% were in China and 28% were in the United States. BEVs comprised

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### Table 1. U.S. annual distance traveled, number of vehicles, and fuel consumed per vehicle by vehicle type.

<table>
<thead>
<tr>
<th>Vehicle type</th>
<th>Rural</th>
<th>Urban</th>
</tr>
</thead>
<tbody>
<tr>
<td>Light duty—long</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Interstate</td>
<td>44</td>
<td>26</td>
</tr>
<tr>
<td>Other arterial</td>
<td>92</td>
<td>90</td>
</tr>
<tr>
<td>Other</td>
<td>39</td>
<td>100</td>
</tr>
<tr>
<td>Interstate</td>
<td>333</td>
<td>1,221</td>
</tr>
<tr>
<td>Other</td>
<td>658</td>
<td>2,192</td>
</tr>
<tr>
<td>Total</td>
<td>54.9</td>
<td>11,991</td>
</tr>
<tr>
<td>Number of vehicles (millions)</td>
<td>11,991</td>
<td>689</td>
</tr>
<tr>
<td>Average miles per vehicle</td>
<td>192.8</td>
<td>2,356</td>
</tr>
<tr>
<td>Fuel consumed per vehicle (gallons)</td>
<td>20</td>
<td>54</td>
</tr>
<tr>
<td>Average miles per gallon</td>
<td>8.7</td>
<td>43.9</td>
</tr>
</tbody>
</table>

| Buses | | |
| 2 | 2 |
| 2 | 2 |
| 3 | 3 |
| 8 | 16 |
| | |
| Single-unit trucks | 10 | 16 |
| 16 | 19 |
| 53 | 113 |
| | |
| Combination trucks | 50 | 29 |
| 12 | 42 |
| 41 | 175 |
| | |
| All light duty | 184 | 318 |
| 302 | 492 |
| 1,554 | 2,850 |
| | |
| All trucks | 60 | 45 |
| 28 | 61 |
| 94 | 288 |
| | |
| All vehicles | 247 | 368 |
| 335 | 558 |
| 1,666 | 3,174 |

Source: FHWA (2017a, Table VM-1).

**Figure 3.** U.S. light-duty vehicle adjusted fuel economy by model year and vehicle type from 1975 to 2017. Source: EPA (2018).
60% of the 2016 EDV stock, with PHEVs constituting the remainder. The EDV fleet in China included 343,500 electric buses. BEV and PHEV annual global sales were 466,420 and 286,750, respectively, in 2016. Several vehicle manufacturers have announced targets for electric car sales, including domestic Chinese manufacturers, Honda, Renault-Nissan, Tesla, Volkswagen, and Volvo. Furthermore, in 2016 thirteen national governments, including China, France, Germany, India, and Japan, and eight U.S. states set EDV targets. Global EDV stock is projected to range between 60 million and 200 million vehicles by 2030 (IEA 2018).

**Future U.S. road transportation energy consumption**

For the United States EIA (2017a) projects highway vehicle energy consumption will peak around 2018, drop monotonically through approximately 2040, and increase slightly thereafter, as indicated in Figure 5. However, total U.S. highway vehicle energy consumption in 2050 is estimated to be 11% lower than in 2015 based on a decrease of 21% for LDVs offset by increases for freight trucks and buses. The decrease in energy consumption for LDVs is a result of increased efficiency despite higher VMT. Increased fuel economy for heavy-duty vehicles offsets increased travel demand through approximately 2040. The share of energy consumption by LDVs decreases from 70% in 2015 to 62% in 2050, while that of freight trucks increases from 25% to 31%. The share of light-duty trucks is expected to increase, but their fuel economy is also projected to increase, from 22.2 mpg in 2016 to 34.6 mpg in 2040.

In the United States, gasoline and diesel will likely continue to be the dominant transportation energy sources. However, gasoline consumption is projected to decrease by 22% from 2015 to 2050, whereas diesel consumption is projected to increase by 10% (EIA 2017a). Although energy consumption from propane, compressed natural gas, E85, electricity, and hydrogen are projected to have large relative increases, ranging from 104% for propane to well over 1000% for electricity and hydrogen, these energy sources contributed 0.5% in 2015 and are expected to contribute only 5.6% in 2050. The expected increase in vehicle electricity consumption, which amounts to only 0.38 Quads in 2050, is attributed to increased shares of BEVs and PHEVs. Key factors to which the projections are sensitive include VMT, vehicle fuel economy, consumer preferences, energy prices, state policies such as California’s Zero-Emission Vehicle regulation, and battery prices.

**Potential errors in projections**

Projections of future energy use and emissions are scenario-based. The probability of actual results falling outside of predicted uncertainty ranges was found to be 70% to 90% (Shlyakhter et al. 1994). Assumed underlying trends in explanatory variables used to project future emissions, such as demographic trends, economic growth, energy intensity relative to gross domestic product (GDP), fuel switching, and application of technologies, were not consistent with later reality (Rafaj, Amann, and Siri 2014). For the Annual Energy Outlook 2016 published by EIA in the United States, prediction intervals were estimated for 18 forecasted quantities (Kaack et al. 2017). The width of the prediction intervals increased with each incremental forecast year. For transportation energy in the United States, which was approximately 28 Quads in 2016 (inclusive of all mobile sources, not just on-road sources), the 80% prediction interval increases from about plus or minus 1 Quad in 2017 to approximately plus or minus 5 Quads in 2026, compared to a mean prediction of 27 Quads. Prediction intervals beyond 10 years of forecast are not estimated. Thus, the projections given here should be taken as an example of a possible scenario but not a prediction of an
actual future. From a policy perspective, some assessment of potential futures is helpful, given that there is lead time to develop new technologies and programs to achieve long-term goals, such as improved energy efficiency and reduced emissions.

**Impact of autonomous vehicles on energy consumption**

The emergence of “new mobility” could dramatically alter the on-road vehicle energy future. New mobility includes technologies such as connected vehicles and autonomous vehicles (AVs), and business models such as ridesharing, carsharing, and e-hailing. The potential market share of AVs is estimated to vary between 2% to 15% by 2030 and between 50% and 90% in 2050 (Slowik and Kamakate 2017). “Fifth-level autonomy” requires no control from a driver at any stage of the vehicle driving process. The timing of penetration of fifth-level autonomy into the on-road fleet depends on availability and social acceptance of AVs, fleet turnover, resolution of legal issues, institutional adaptation, and other factors. Widespread adoption of fifth-level autonomy might occur by mid-century (around 2050) or later (Igliński and Babiak 2017).

Reducing travel delay associated with improved coordination between vehicle driving strategy and signalized traffic control could reduce the frequency and duration of accelerations that will reduce energy use and emissions (Saust, Wille, and Maurer 2012). A potential benefit of AVs is to achieve more efficient driving along the concepts of “eco-driving” by modifying speed and acceleration to reduce energy use for a given traveled mileage. Eco-driving may reduce fuel consumption by 10% to 20% (Igliński and Babiak 2017). The use of connected and AV technology simultaneously can lead to new methods of traffic control that improve traffic flow, enable less variability in speed, and avoid stops, which in turn can reduce energy consumption and emissions (Lin et al. 2017). The actual energy efficiency of an AV will depend on its control scheme. AVs could be designed, for example, to achieve shortest travel time, or fastest speed, and not necessarily minimal energy use on a per trip basis. Fuel economy (and emissions) testing procedures should be adapted to characteristics of AVs (Mersky and Samaras 2016).

AVs are expected to improve travel time reliability. Time spent in an AV could be spent productively engaging in...
work or leisure activities. These factors could lead to growth in travel demand. AVs could lead to lower density land use patterns associated with tolerance for longer commuting distances (Rubin 2016). AVs could be inherently safer than human-operated vehicles and therefore be smaller, lighter, and more energy efficient. Furthermore, vehicles could be more specialized and “right-sized” (e.g., for a commuting trip versus camping trip) to their purpose (Morrow et al. 2014).

Low levels of automation may lead to energy use reductions through trajectory smoothing, whereas higher levels of automation could lead to increased travel activity. Eco-driving and platooning appear to offer substantial reductions in energy intensity. Unanticipated or excessive growth in travel demand could be met with dynamic road pricing schemes (Wadud, MacKenzie, and Leiby 2016). AVs would likely increase mobility for the elderly and other non-driving and travel-restricted populations, such as those with medical conditions. A bounding estimate of this source of latent demand is a 14% increase in annual LDV miles traveled for those 19 years and older (Harper et al. 2016).

AV deployment could change energy use by −90% to +150%, taking into account vehicle effects including platooning, efficient routing, efficient driving, faster travel, more travel (e.g., by underserved populations), lighter vehicles, less time looking for parking, higher vehicle occupancy, and electrification (Brown, Gonder, and Repac 2014). Although many AV studies focus on LDVs, automation has implication for HDDVs. For example, platooning combination trucks might realize fuel savings of 6.4% (Lammert and Gonder 2014). The overall effect of AV deployment on system-wide energy use is highly uncertain, scenario-based, and speculative.

Greenhouse gas emissions

Global CO₂ emissions from fossil fuel combustion reached approximately 32.3 metric gigatonnes (Gt) in 2015, compared to 20.5 GtCO₂ in 1990. Road transport accounts for approximately 18% of global CO₂ emissions. Global road transport emissions increased by 73% from approximately 3.3 GtCO₂ in 1990 to 5.7 GtCO₂ in 2015 (IEA 2017b).

Road transport GHG emissions include CO₂, CH₄, N₂O, and HFCs. When converted to a CO₂-equivalent basis taking into account global warming potential (GWP), the share of these four GHGs to total CO₂-equivalent emissions from road transport are 96.4%, 0.1%, 0.7%, and 2.8%, respectively, in the United States in 2015 (EPA 2017a). Thus, CO₂ accounts for the vast majority of the GHG emissions from transport. Aside from GHGs, black carbon is important with regard to short-term climate forcing (EPA 2012). Furthermore, ozone formed from precursor emissions of NOₓ and VOC is typically considered to be the third most important GHG after CO₂ and CH₄ (USDOT 2010).

The trend in U.S. road transport GHG emissions from 1990 to 2015 is shown in Figure 6 in terms of the share of total emissions by GHG, by vehicle type, and by fuel type (EPA 2017b). From 1991 to 2007, these emissions were steadily rising. The 2008 global economic recession led to a temporary decrease. However, compared to 1990, emissions are up by 16% for passenger cars, 79% for medium- and heavy-duty trucks, and 23% overall. The relative share of medium- and heavy-duty trucks and the share of diesel fuel consumption have slowly been increasing, from 19% to 27%. Alternative fuels, such as compressed natural gas (CNG) and liquefied petroleum gas (LPG), are less than 1% of total fuel consumption.

Because the majority of road transport GHG emissions are exhaust CO₂ emissions (EPA 2017a), the fraction of carbon in fuel converted to CO₂ in internal combustion engines that comprise the vast majority of the on-road fleet is typically more than 99% (Brimblecombe et al. 2015), and transportation fuels are mainly fossil fuels, the relative trends in GHG emissions from road transport are approximately similar to the relative trends in energy consumption. Ethanol and biodiesel contributed only 5% of total transportation sector energy (EIA 2017c). Thus, although they have lower net CO₂ emissions impact, they do not yet substantially affect trends in national road transport fuel CO₂ emissions.

From 2010 to 2050, GHG emissions reductions from adoption of new LDV technologies could potentially be 54% to 65% in OECD North America, 39% to 76% in Europe, and 51% to 55% in China, based on reductions for gasoline, PHEVs, BEVs, and diesel LDVs (Taptich, Horvath, and Chester 2016). For medium- and heavy-duty freight, the potential reductions are smaller, in the range of 31% to 33% in the three global regions. Intermodal substitutions have potential to reduce GHG emissions, but as vehicles become more efficient, the marginal benefit of such substitutions declines. For example, in OECD North America, substituting a bus for a gasoline auto reduced GHG emissions by an average 1.7 kg CO₂-eq/passenger-trip in 2010, but only by an estimated 0.39 kg CO₂-eq/passenger-trip in 2050.

GHG emissions benefits from PHEVs and BEVs are dependent on the energy mix used for power generation.
Similarly, GHG emissions benefits from FCEVs that use hydrogen as fuel are dependent on the hydrogen fuel cycle, which currently is based on steam reforming of methane and which emits 12 kg CO\textsubscript{2}-eq per kg H\textsubscript{2} as part of the fuel cycle (Lee, Elgowainy, and Dai 2018). A BEV using electricity generated from natural gas would emit less CO\textsubscript{2} than an FCEV whose fuel is based on steam reforming of methane (Ramachandran and Stimming 2015). Furthermore, if H\textsubscript{2} is produced from electrolysis using renewably produced electricity, more total energy would be consumed than needed by a comparable BEV. More deployment of FCEVs will depend in part on development of fueling infrastructure (NRC 2015).

Based on an energy-mix analysis using the MARKAL model, the deployment of EDVs, which include PHEVs and BEVs, may not have a substantial effect on road transport GHG emissions unless coupled with strategies for reducing the carbon intensity of power generation (Babaee, Nagpure, and DeCarolis 2014). The final rule for the Clean Power Plan was an attempt, at the federal level, to reduce GHG emissions from power generation (EPA 2015e). The recently proposed regulatory action to rescind, revise, or replace the Clean Power Plan could reduce benefits of the 2012 and later GHG vehicle standards to the extent that there is increased EDV use (EPA 2017d).

**Criteria pollutant and air toxic emissions**

This section summarizes empirically based estimates of historic or recent trends in road vehicle emissions of NO\textsubscript{x}, CO, PM, VOCs, and mobile source air toxic (MSAT) species, followed by an evaluation of emission-inventory based estimates of past trends in such emissions.

**Recent empirical trends in NO\textsubscript{x} emissions**

Global annual NO\textsubscript{x} emissions reportedly peaked in 2013. In 2010, motor vehicles contributed 29% of global NO\textsubscript{x} emissions. However, the contribution of vehicles to NO\textsubscript{x} emissions in developed countries (33% globally, 35% in North America) is greater than in developing countries (27%), based on differences in motorization rates. The
recent peak and subsequent downturn in global NOx emissions is attributed to increasingly stringent emission standards for vehicles in the United States, Europe, Japan, and China (Huang et al. 2017).

Based on remote sensing measurements made in selected U.S. cities in 1997, 1999, and 2002, a general reduction was observed in the percent of fleet CO, NOx, and HC emissions from the most recent four model years in each calendar year, even though the applicable Tier 1 emission standard had not changed during that time. Newer vehicles that have aged several years were found to be lower emitting than new vehicles from several previous model years. There was not significant evidence of emissions deterioration with increasing age (Pokharel et al. 2003).

Based on multiple types of measurements, including remote sensing and tunnel studies, Los Angeles Basin mobile source NOx emissions peaked around 1986 and thereafter decreased by approximately 65% through 2015. These trends are consistent with measured NOx/CO ratios (Hassler et al. 2016). Based on remote sensing and tunnel measurements in Los Angeles, CA, Houston, TX, and New York, NY, from 1990 to 2010, fuel-based (grams of NOx per kg of fuel) gasoline vehicle NOx emission rates per average vehicle steadily decreased. Total NOx emissions from gasoline vehicles decreased monotonically by 65%, whereas diesel NOx emissions increased from 1990 to about 2000, remained nearly flat, and then decreased after 2007 (McDonald et al. 2013). Although there have been many other tunnel studies in the United States, Europe, and Asia (e.g., for a review see Ropkins et al. 2009), the focus here is on repeat studies that enable direct quantification of trends at the same site.

Fuel-based NOx emission rates, which are inferred from the ratio of concentrations of nitrogen oxides and carbon species (CO2, CO, HC) measured in the plumes of passing vehicles using remote sensing, were found to decrease by 21% to 43% based on repeated remote sensing measurements in Denver, CO, Tulsa, OK, and Los Angeles, CA. Measurements were made in 2005, 2005, and 2008, respectively, and repeated in 2013 in all three cities. NH3 emission rates were also found to decrease, but at a lower rate (Bishop and Stedman 2015). Moreover, vehicle NH3 emissions appear to be underestimated in the EPA National Emission Inventory (NEI) (Sun et al., 2017).

Based on chassis dynamometer measurements of 64 in-use LDGVs, the emission rates for vehicles certified to the California LEV2 standard were lower than pre-LEV vehicles by approximately a factor of 100 for NOx, 10 for CO, and 10 for total hydrocarbons (THC) (May et al. 2014).

Trends in vehicle emissions and their impact on roadside air quality can differ substantially among cities. For example, in London, UK, roadside concentrations of NO2 increased from 2005 to 2009 as a result of increased emissions from diesel vehicles. The introduction in 2005 of diesel particle filters (DPFs), which control particulate matter less than 2.5 µm in aerodynamic diameter (PM2.5) and black carbon (BC) emissions, but lead to an increased ratio of NO2 to NO in vehicle exhaust, contributed to the NO2 concentration increase. The Euro 5 standard introduced in 2009 mandated lower NOx emission rates for heavy-duty diesel vehicles. From 2010 to 2014, reductions in roadside concentrations of NOx, PM2.5, and black carbon were observed (Font and Fuller 2016). Based on remote sensing of 84,269 vehicles at multiple UK locations, NOx emissions from LDDVs appear to be higher than the level of their applicable standards, and these emissions have changed little despite increasingly stringent regulations (reasons are discussed later). For heavy-duty vehicles, NOx emissions for Euro 4 vehicles were about 30% lower than for older vehicles; however, there is not much evidence of changes in emissions for transit buses from Euro 1 to Euro 4 standards (Carslaw et al. 2011).

Based on plume-chasing measurements, fuel-based NOx and BC emission factors were quantified for diesel franchised buses and HDDVs in Hong Kong. The results demonstrate that the real-world emission rates generally decreased with increasingly stringent standards. NOx emissions of Euro 4 vehicles were lower than for Euro 1 vehicles by 55% and 35% for franchised buses and HDDVs, respectively, versus a reduction of 50% according to the standards. Not all of the identified high emitters were from the oldest, least stringent regulatory class. Furthermore, a vehicle that was a high emitter for one pollutant was not necessarily a high emitter for another pollutant (Lau et al. 2015). Intervehicle variability in results is a fairly common characteristic of measurement studies.

Recent empirical trends in CO emissions
Ambient concentration decreases in CO and VOCs from 1960 in the Los Angeles Basin are associated with lower emission rates of new vehicles and turnover of the vehicle fleet (Warneke et al. 2012). During the 1990s, annual CO emissions from vehicles were estimated to decrease by 42%, which is consistent with observed trends in the ratio of CO to NOx emitted by vehicles (Parrish et al. 2002). Empirical evidence of reduction in vehicle CO emissions was reported along with measurements of NOx emissions reductions (Pokharel et al. 2003).

From 1990 to 2010, CO emission rates from on-road vehicles in Los Angeles, CA, Houston, TX, and New York, NY, decreased by approximately 80% to 90%. Emissions
of nonmethane hydrocarbons (NMHC) followed a similar trend. These estimates are based on fuel-based emission rates measured in remote sensing and tunnel studies (McDonald et al. 2013). Also in Los Angeles, estimated NOx/CO ratios from remote sensing and tunnel studies were compared to other data sources, such as ambient monitoring. These ratios have changed over time as a result of implementation of successive generations of oxidation, dual-bed, and three-way catalytic converters in light-duty gasoline vehicles, coupled with growth in activity of diesel trucks. The NOx/CO ratio typically increases with these trends, because CO emissions have been reduced at a faster rate than NOx emissions. Mobile source CO emissions were estimated to peak in 1978 and to decline monotonically, except for a slight increase just after the 2008 recession, through 2015, for an overall decrease of approximately 88% (Hassler et al. 2016).

**Recent empirical trends in particulate matter emissions**

From 1990 to 2012, ambient concentrations of diesel particulate matter (DPM) decreased by 68% in California even though diesel VMT increased by 81%. These reductions are attributed to cleaner fuel and vehicle emission regulatory programs in California (Propper et al. 2015). Some of these programs are similar to those adopted nationally, while others are unique to California and are unlikely to be replicated nationally. For example, California requires that by 2023 nearly all HDDVs will need to meet 2010 or newer emission standards, which in turn requires retrofit or replacement of older vehicles.

Based on chassis dynamometer measurements of 485 in-use LDGVs in Kansas City, MO, PM2.5 emissions decreased by 86% to 97% on average when comparing 1996 and newer model year vehicles to pre-1981 model year vehicles, depending on vehicle type (passenger car, passenger truck), and season. Similarly, emissions of gaseous pollutants decreased by a factor of approximately 10 for THC and CO and by a factor of approximately 3 for NOx (Kishan et al. 2008).

Based on chassis dynamometer measurements of 64 in-use LDGVs spanning California pre-LEV to LEV2 emission standards, primary emissions of particulate matter (PM) mass and organic carbon also decreased. Five diesel vehicles were also measured. Cold-start emissions of secondary organic aerosol (SOA) precursors decreased 80% for LEV1 versus pre-LEV vehicles, and by 60% for LEV2 versus LEV1 vehicles. However, the ratio of SOA precursors to primary PM from LDGVs ranged from 20 to 90 for measurements that included cold starts, versus only 2 to 4 for hot-start tests of diesel vehicles. Thus, SOA formation is likely to be proportionally more significant for gasoline than diesel vehicles, although both contribute SOA precursors to the atmosphere. Among diesel vehicles measured, those with after-treatment devices, such as diesel oxidation catalyst (DOC), DPF, and selective catalytic reduction (SCR), had much lower emissions of targeted pollutants than those without. For example, CO, THC, and PM emissions were reduced by approximately 99% and NOx was reduced by approximately 80% (May et al. 2014). Both diesel and gasoline vehicles are known to emit ultrafine particles (UFP) in size ranges of less than 100 nm (Slezakova et al. 2013).

**Recent empirical trends in VOC and MSAT emissions**

The ratio of near-road ambient VOC to NOx declined by approximately 40% at U.S. locations from 1987 to 1999. VOCs emitted by vehicles are typically more reactive with respect to ozone formation than VOCs emitted from other sources (Parrish et al. 2002).

VOC emissions were found to decrease from 1999 to 2009 in the Southern California region based on comparison of trends in tunnel studies, dynamometer measurements, and speciated ambient concentrations. For example, ambient concentrations of VOC species that serve as markers for tailpipe exhaust emissions, such as acetylene, were found to decrease, whereas species associated with other sources were stable. Furthermore, tailpipe exhaust VOC, rather than evaporative losses, was identified as the dominant contributor to vehicle VOC emissions in the tunnel studies. The real-world VOC emissions may be higher than those predicted from the EMFAC vehicle emission model used in California (Pang et al. 2015).

Based on analysis of remote sensing, tunnel study, and ambient concentration data, the contribution of on-road emissions in California to ambient concentrations of BC, primary organic aerosols (POA), and SOA was approximately 90% in 1970, but has declined through 2010 as a result of improved emissions control. Trends in the formation of SOA are influenced by fuel formulation, VOCs emitted from vehicles, and downward trends in ambient organic aerosol that could serve as condensation sites for higher molecular weight semi-volatile organic compounds (SVOCs) that contain 20–26 carbon atoms. The yield of SOA in 1970 was predicted to be 1.5 and 2 times higher for gasoline and diesel vehicles, respectively, compared to 2010 based on these factors. The speciation of VOCs emitted by LDGVs has changed over time in California, with LEV1 and LEV2 compliant vehicles producing higher fractions of intermediate VOCs (IVOCs), containing 13–19 carbon atoms, that have high potential to form SOA. Diesel vehicles tend to emit a higher proportion of IVOCs compared to gasoline (McDonald, Goldstein, and Harley 2015).
Speciated nonmethane organic compounds (NMOC) and oxygenated hydrocarbons, and CO and NO\textsubscript{X} (total reactive nitrogen include NO\textsubscript{X} and HNO\textsubscript{3}) ambient concentrations from 1999 to 2007 in Atlanta, GA, were consistent with downwind trends in vehicle emission inventories and with roadside and tunnel measurements (Blanchard et al. 2010).

Markers of vehicle exhaust and evaporative emissions from vehicle operation and fueling, such as benzene and 1,3-butadiene, decreased by 88% or more from 1990 to 2012 in California, even though LDV VMT increased by 31%. Tailpipe benzene and 1,3-butadiene vehicle emissions, based on California’s vehicle surveillance program, decreased by 89% and 86%, respectively, from 1995 to 2003 (Propper et al. 2015).

With the introduction of the catalytic converter, lead was phased out of gasoline in the United States to prevent damage to the catalyst, and was fully phased out by 1996. In 1970, highway vehicles emitted an estimated 170,000 tons of lead of a total of 225,000 tons emitted by all sources. By 1990, highway vehicles emitted less than 200 tons. By 2008, the contribution of highway vehicles to national lead emissions was negligible. Lead emissions from current on-road vehicles could occur in small amounts because of wear or loss of vehicle parts that contain lead, such as wheel weights. Lead can be a component of brake dust, depending on brake pad formulation. Lead may be emitted in exhaust as a result of lead impurities in motor fuels and lubricating oils (EPA 2013d).

**Implications of recent empirical trends**

Measurement studies indicate downward trends in real-world vehicle emissions of NO\textsubscript{X}, CO, PM, VOC, and MSATs consistent with the implementation of federal and California emission standards. Despite limitations of the data, such as the potential for variation in fleet and operational characteristics not addressed by the selected measurement sites and time periods, the data support a robust conclusion that vehicle emissions prevention and control strategies have been effective.

For example, a comparison of tailpipe exhaust measurements of 39 Tier 2 PCs versus 24 Tier 1 PCs based on real-world PEMS measurements of in-use vehicles indicated that the Tier 2 vehicles had 23% to 64% lower NO\textsubscript{X} emission rates, on average, depending on vehicle specific power (VSP), which is an indicator of engine load. Average rates for CO were 41% to 69% lower for Tier 2 than Tier 1 vehicles depending on VSP, and HC rates were 18% to 36% lower (Liu and Frey 2015b). U.S. Tier 1 emission standards apply to all vehicles less than 8,500 lb gross vehicle weight rating (GVWR) and thus apply equally to PCs and PTs. Later emission standards, including Tier 2 and Tier 3, also apply to PCs and PTs (DieselNet 2018).

Vehicle emission reduction programs based on the U.S. and European regulatory frameworks are reducing emissions of vehicles in many parts of the world, although the U.S. standards are more effective because of differences in regulatory procedures (Nesbit et al. 2016) and differences in vehicle fleet mix.

**Recent trends in vehicle emissions based on emission factors**

Many emission inventories are based on a “bottom-up” approach in which vehicle emission factors, in units of mass of pollutant emitted per distance traveled per vehicle, are multiplied by vehicle miles traveled to arrive at a total mass of emissions for a given region and time period. These assessments are typically stratified by vehicle type, fuel type, and road type, and may be further stratified by link-based average speeds and account for inspection and maintenance (I/M) programs, hourly variability in VMT and ambient conditions, and multiple emissions processes such as cold start, hot stabilized emissions, running losses, and others (EPA 2015f). NARSTO (a public–private partnership that was dedicated to improving air quality in the United States, Canada, and Mexico) identified key weaknesses specific to mobile source emission inventories. These include that some estimates based on small sample sizes may not be representative of real-world activity, insufficient information is available on speciation of organic compounds and particulate matter, and uncertainties are rarely or not rigorously quantified (Miller et al. 2006; Werner et al. 2005). NARSTO characterized the confidence level in on-road mobile source inventories as being medium–high for NO\textsubscript{X}, low–medium for VOC, and low–medium for hazardous air pollutants. Quantitative assessments of uncertainty in mobile source emissions have often revealed substantial intervehicle variability that leads to large ranges of uncertainty in fleet average emission factors (Frey and Li 2003; Frey and Zheng 2005; 2002).

Since the NARSTO assessment, EPA released a new mobile source emissions model, MOVES, that was motivated by a National Research Council (NRC) evaluation of mobile source modeling (NRC 2000). In response to a recommendation from the NRC to better account for “superemitters,” EPA derives MOVES mean emission estimates from frequency distributions of vehicle emissions (EPA 2015d). The NRC found that model validation and evaluation had not been adequately addressed by EPA for its predecessor MOBILE series of models. EPA has responded by conducting evaluations using data independent of those used for
model calibration, such as from I/M programs, tunnel studies, and remote sensing (Sonntag 2017). EPA engaged a working group of outside experts under the Federal Advisory Committee Act to provide periodic advice and comment (EPA, 2016c). The NRC recommended that EPA conduct sensitivity and uncertainty analysis. EPA has been more active in conducting sensitivity analysis, as have agency partners such as FHWA (Noel and Wayson 2013; Sonntag 2015). However, EPA has not included a MOVES uncertainty analysis capability for the general user. Concerns have arisen regarding apparent overestimation of vehicle NO\textsubscript{x} emission rates (Bai 2016), although bias may be attributable to using national defaults instead of site-specific data, especially for the percentage of trucks and vehicle age distribution (Bai et al. 2017). EPA typically does not have adequate resources to sponsor its own large-scale surveillance programs and, instead, often relies on data from other entities such as independent researchers. Despite these limitations, the available emission inventories typically represent the best estimate of recent trends and current status in vehicle emissions. They are used here for the purpose of assessing relative trends.

Relative trends in U.S. highway vehicle annual emission inventories for NO\textsubscript{x}, CO, VOC, and PM\textsubscript{10} are shown in Figure 7 based on the National Emission Inventory (NEI) (EPA 2015b). There was a methodology change in how the emission inventories were developed in 2002 that led to a discontinuity in the mass emissions time series. As a simplifying assumption, for the purpose of showing relative trends, the relative emissions in 2002 were assumed to be the same as in 2001. From 1970 to 2015, vehicle emissions are estimated to decrease nationally by approximately 75% to 90%, depending on the pollutant. These reductions are the combined effect of increasing vehicle population and VMT offset by decreases in mass per distance emission rates as a result of programs aimed at vehicle fuels and emissions.

Trends in vehicle emission factors for selected pollutants and vehicle types are shown in Figure 8 based on values reported by the U.S. Department of Transportation (USDOT 2017). These estimates go back only to 1990, and thus account for only part of the total emissions reductions shown in Figure 7. However, Figure 8 illustrates that emission rates per VMT have continuously declined in the last two decades for NO\textsubscript{x}, CO, and HC for LDGVs and HDDVs. These two vehicle groups make up the largest share of the U.S. on-road fleet.

The incorporation of MSAT emissions into mobile source emissions models was recommended by the NRC in 2000 and implemented by EPA in MOVES (EPA 2016d; NRC 2000). U.S. highway vehicle MSAT emissions were estimated to decrease from 1990 to 2011 by 43% for acetaldehyde, 68% for acrolein, 81% for benzene, 80% for 1,3-butadiene, and 73% for formaldehyde (EPA 2014a). These emissions are mostly from LDGVs. Many of the emissions processes relevant to MSATs are evaporative and thus are influenced by changes in fuel formulation as a result of fuel standards. A portion of MSATs emissions are from tailpipe exhaust. These emissions are declining because of changing fuel formation coupled with more efficient emission controls.

**Estimated future trends in vehicle emissions**

Unlike the case for energy and CO\textsubscript{2} emissions, there are not readily available national projections of future on-road vehicle emissions of selected criteria pollutants and MSATs. Therefore, projections were developed for the United States using MOVES (EPA 2015f). These projections focus on mass per distance emission factors and therefore do not account for increases in VMT. The projections focus on selected vehicle types and pollutants as illustrative examples.

Figure 9 is a projection of U.S. national average NO\textsubscript{x} emission rates for gasoline passenger cars based on 2015–2050 calendar years. In each calendar year, MOVES models a 30-year-old vehicle age distribution. MOVES applies a deterioration rate to zero mileage emission factors to estimate trends in average emissions versus age associated with loss of three-way catalyst (TWC) activity and other causes (EPA 2015d). A 0-year-old vehicle in 2015 is certified to the Tier 2 emission standard. A 0-year-old vehicle in 2017 is certified to the Tier 3 emission standard. Tier 3 is assumed to continue into the future. Thus, predicted emission rates for 0-year-old vehicles are approximately constant for 2017 and beyond. However, for a given calendar year, older vehicles are subject to deterioration, and there can be vehicles certified under previous standards. For example, in 2015, all vehicles from 2007 onward were Tier 2, most vehicles between 2004 and 2006 were certified under Tier 2 as part of a phase-in period, vehicles from 1994 to 2003 were Tier 1, and vehicles prior to 1994...
Figure 8. Trend in U.S. national average emission rates from 1990 to 2010 for (a) light-duty gasoline vehicles, (b) heavy-duty diesel vehicles, and (c) average vehicle. Source: U.S. DOT (2017), (Table 4-43).

Figure 9. Trends in U.S. national average NOx emissions rates for gasoline-fueled passenger cars for calendar years 2015–2050 and ages 0–30 years, estimated using MOVES2014adefault inputs.
were subject to earlier standards that were modified in 1982 and 1987 (USDOT 2017).

The highest part of the peak in Figure 9 represents deteriorated vehicles from the pre-1994 period. As these older vehicles leave the fleet, emissions of 30-year-old vehicles drop gradually from 2015 to 2025, and then fall substantially through 2035. By 2035, Tier 1 vehicles will have mostly left the on-road fleet.

Figure 10 illustrates backward and forward trends in NO\textsubscript{x} emission rates for long-haul combination truck HDDVs for the 1990–2050 calendar years with 30-year age distributions in each calendar year. The figure illustrates that HDDVs just prior to 1990 are estimated to have NO\textsubscript{x} emission rates of approximately 22 g/km. HDVs have survival rates of approximately 20% or more at age 30 (Davis, Williams, and Boundy 2017). Emissions of vehicles older than 30 years are not included in MOVES.

Figure 11 illustrates the trend in primary PM\textsubscript{2.5} emission factors for diesel-fueled long-haul combination trucks based on MOVES for calendar years 1990–2050 and ages 0–30 years in each calendar year. The figure illustrates that trucks will have much lower emission rates effective with the 2007 model year and that fleet average emission rates will decrease as pre-2006 vehicles age out of the fleet.

The contribution of individual emission processes to gasoline passenger car emission rates is illustrated in Figure 12 for calendar years 1990–2050. These emission factors are fleet averages that are based on a 30-year age distribution in each calendar year. Figure 12a illustrates that THC emissions are the result of nine emissions processes, including running exhaust, start exhaust, crankcase running losses, crankcase start losses, leaks, venting, permeation, displacement, and spillage (EPA 2015d). Running exhaust, crankcase start, and fuel tank venting releases are the main contributors. With the implementation of successively more stringent standards and fleet turnover, the fleet average emission rate is estimated to stabilize around 2030. However, uncertainty in these estimates is not quantified, and the data upon which some of these estimates are based are relatively limited. Figure 12b illustrates that running exhaust emissions have historically dominated the total NO\textsubscript{x} emissions from gasoline PCs.

Temporal trends in fleet average NO\textsubscript{x} emission factors for 12 vehicle types are illustrated in Figure 13a. These estimates are based on gasoline fuel for MCs, PCs, PTs, and LCTs, and diesel fuels for the remaining vehicles, which are the predominate fuel–vehicle combinations in the U.S. fleet. Diesel vehicles, because of their much higher compression ratios and lean-burn fuel/air mixtures, have much higher NO\textsubscript{x} emission rates than gasoline engine vehicles (Kalghatgi 2015). The emission factors for all 12 vehicle types have projected decreases from 1990 to the present as a result of increasingly stringent emissions standards. Continued reductions are projected as a result of fleet turnover in the coming decades. However, even with SCR, diesel vehicles are expected to have larger NO\textsubscript{x} emission rates than gasoline vehicles in the future. Thus, as shown in Figure 13b, the share of total NO\textsubscript{x} emissions is projected to decrease from PCs and PTs and increase for combination long-haul trucks. The larger share of motorcycles in later years is because emission factors from other vehicle types decrease faster than for motorcycles.

Figure 14 similarly illustrates trends in PM\textsubscript{2.5} emission rates over time for 12 vehicle groups. Primary PM\textsubscript{2.5} emission rates from gasoline vehicles are currently very low compared to those from diesel vehicles. Diesel vehicle
primary PM$_{2.5}$ emission rates are estimated to drop by approximately two orders of magnitude as the result of fleetwide deployment of DPFs, assuming that they are durable with low malfunction rates. Even though the share of fuel consumption and VMT of heavy diesel vehicles are expected to increase, the proportional contribution of such vehicles is projected to decrease, as indicated in Figure 14b. However, a source of uncertainty not accounted for currently in MOVES is the extent to which the increased adoption of gas direct injection (GDI) vehicles in the on-road fleet might increase primary particle emissions (e.g., Saliba et al. 2017), particularly for UFPs in the nanoparticle size range, largely as a result of impingment of fuel on piston or cylinder wall surfaces leading to “wetting” and cooling effects that promote particle formation (Myung and Park 2012). GDI particle emissions can be mitigated by installation of a gasoline particle filter (GPF) (e.g., Saffaripour et al., 2015), but this is not yet an established practice in the U.S.

Factors affecting travel demand and vehicle operation

Factors that encourage mode choice in favor of private automobile travel include decreased street connectivity, lack of mixed land use, low dwelling density, longer distance to transit, lack of sidewalks, high income level, and others (Boulange et al. 2017; Ding et al. 2017; Shay and Khattak 2012; Kay, Noland, and Rodier 2014; Garikapati et al., 2017; Sun et al., 2017; Shekarchian et al. 2017). Increasing the price of driving through various mechanisms, such as taxes on vehicle purchase, ownership, age, and CO$_2$ emissions, cordon pricing, and pricing schemes for parking, could decrease private vehicle use (Kay, Noland, and Rodier 2014; Welch and Mishra 2014; Liu and Cirillo 2016; Feng, Fullerton, and Gan 2013; Chu 2015; Nourinejad and Roorda 2017; Ottosson et al., 2013; Ahmadi Azari et al. 2013). Low emissions zones

Figure 12. Estimated U.S. fleet average gasoline passenger car emission factors from 1990 to 2050 for (a) total hydrocarbons and (b) nitrogen oxides by emissions process based on MOVES.
LEZs) restrict vehicle access to designated parts of a city but also change traffic and parking outside of the zone (Dias, Tchepel, and Antunes 2016; Malina and Scheffler 2015). More discussion of factors affecting travel demand, and their implications for energy consumption and emissions, are given in the supplemental materials.

Vehicle energy consumption

Fuel economy for the on-road fleet began to improve in Europe and Japan by 2000 and in the United States starting around 2003. In Europe, a shift toward diesel cars did not lead to as much of a fuel economy benefit as expected (Schipper 2011). Compared to 1975, U.S. LDGVs have the same average weight but 65% more horsepower and 90% better fuel economy (EPA 2016e). Changes in vehicle technology, as described in the following, have led to improved LDGV fuel economy.

Fuel economy improvements for HDDVs are closely related to practices for reducing GHG emissions. Practices for reducing diesel fuel use by HDDVs include anti-idling, air conditioning system improvement, aerodynamic drag reduction, rolling resistance improvement, transmission improvement, hybridization with regenerative braking, improved engines, reduction in accessory load, and modifications in driver operational practice (Frey and Kuo 2007; USDOT 2010).

Additional information is given in the supplemental materials regarding the effect of weight on HDDV energy consumption and driver-selectable operating modes in newer LDVs.
Fuels

The Clean Air Act Amendments of 1990 added requirements to enable lower emissions from gasoline, mandating reduced sulfur content, lower volatility, and use of reformulated gasolines containing oxygenated additives such as MTBE (methyl tert-butyl ether) and ethanol in certain areas and seasons. Later, with the 2005 Energy Policy Act, the oxygenate requirement for reformulated gasoline was removed and a renewable fuel standard was developed that led to removal of MTBE and blending of ethanol (EPA 2013a). Most gasoline sold throughout the United States is a blend of 90% gasoline and 10% ethanol by weight, referred to as E10 (EIA 2017b). Under the authority of the Clean Air Act, EPA has adopted gasoline standards to reduce emissions of multiple air pollutants indirectly, by reducing sulfur content to enable the use of catalytic converters, and directly, by reducing precursors to MSAT emissions and reducing evaporative emissions of VOCs. EPA has also adopted diesel fuel standards aimed at reducing emissions of NO\(_x\) and PM by reducing sulfur content that would adversely affect postcombustion controls.

Gasoline and diesel components and properties

Reduction of fuel sulfur content in both gasoline and diesel vehicles enables deployment of postcombustion controls including catalytic converters, SCR, and DPF. Sulfur competes with pollutants, such as CO, NO\(_x\), and HC, for reaction sites on catalyst surfaces. Exposure to sulfur reduces catalyst activity and leads to premature aging of the catalyst (Blumberg, Walsh, and Pera 2003). In the United States, gasoline sulfur standards have been revised periodically to enable greater reductions of CO, NO\(_x\), and HC, most recently with the implementation of the Tier 3 emission standard in 2017 that requires an annual average limit of 10 ppm of sulfur (EPA 2014b).

Sulfur in diesel fuel is oxidized to SO\(_2\) and absorbs into exhaust water vapor to form sulfate aerosols. Sulfur emitted...
as gaseous $\text{SO}_2$ can lead to the formation of secondary sulfate aerosols. Diesel oxidation catalysts, which oxidize CO, HC, and the soluble organic fraction of PM, will also oxidize $\text{SO}_2$, which enhances formation of sulfate nanoparticles. The net efficiency of a DOC in reducing particle emissions is typically as much as 30% with low-sulfur fuel versus 15% with higher sulfur fuel. Sulfur reduces the efficiency of DPFs and increases the temperature required for DPF regeneration. Low NO$_x$ traps (LNTs), which have been used typically as an interim step between no post-combustion control and later implementation of SCR, can store sulfur as sulfates, which reduces the ability of the trap to store NO$_x$. SCR catalysts can oxidize $\text{SO}_2$ and, in combination with ammonia slip (unreacted ammonia), lead to the formation of ammonium bisulfate salts, which can deposit on downstream surfaces and are a respiratory irritant (Blumberg, Walsh, and Pera 2003). In the United States, ultralow-sulfur diesel (ULSD) not exceeding 15 ppm sulfur was mandated for highway use as of July 15, 2006, just prior to the introduction of 2007 model year heavy-duty diesel trucks equipped with DPFs in response to new exhaust emission standards (EPA 2000b).

Several composition-related properties of gasoline affect vehicle emissions including aromatic content, ethanol volume, Reid vapor pressure (RVP), T50, and T90. RVP is an indicator of the volatility of the fuel. The potential for evaporative emissions, as well as cold start characteristics, is related to RVP. T50 and T90 are the temperatures at which 50% and 90%, respectively, of the volume of the fuel is distilled (evaporated) (EPA 2013c). Adding ethanol to gasoline tends to increase RVP, particular at the E10 blending ratio (Andersen

*Figure 15. Estimated trend in emission rates of selected mobile source air toxics (MSATs) based on MOVES for U.S. gasoline passenger cars from 1990 to 2050 based on a July weekday.*
et al. 2010). EPA requires the use of reduced RVP fuel blends in summer months depending on the state, with more stringent (lower RVP) requirements in ozone National Ambient Air Quality Standards (NAAQS) nonattainment areas, but can allow an increase in RVP of 1.0 psi related to E10 blends (EPA 2015c).

Increased aromatic content is associated with increases in emissions of NMHC, NO\textsubscript{x}, PM, CO, and MSATs, and a decrease in methane emissions. Increased RVP leads to lower emissions of THC and MSATs with a trade-off of increased CO emissions. Increases in T50 and T90 are associated with increased emissions of most pollutants, but no significant change in NO\textsubscript{x} (EPA 2013c). Summertime RVP levels have been increasing, in part because of the 1.0 psi RVP waiver and reduction in the number of areas that must use low-RVP fuels (EPA 2017c).

To address seasonal variability in ambient air quality, particularly related to CO, many locations in the United States used oxygenated winter fuel blends. In a 1994 tunnel study in California, gasoline with 2.0 wt% oxygen was associated with CO and VOC emission rates that were 21% and 18%, respectively, lower than for a low oxygenate (0.3 wt%) fuel blend. There was no change in NO\textsubscript{x} and acetaldehyde emissions, whereas benzene emissions decreased by 25% and formaldehyde emissions increased by 13% (Kirchstetter et al. 1996). A follow-on study in 1997 further observed that fleet turnover has an important role in reducing emissions of CO, VOC, and NO\textsubscript{x} (Kirchstetter et al. 1999). A separate research group also found that reformulated fuel use in California led to lower benzene and aromatic compound emission rates (Gertler et al. 2016). However, there were no air quality nonattainment areas in the United States for CO as of 2010, which was the key motivator for using oxygenated winter fuel blends.

Based on Federal Test Procedure (FTP) chassis dynamometer measurements of 1984–2007 model year vehicles, ethanol blends (including E10, E20, E50) typically had lower CO, THC, and NMHC emissions than for certification gasoline, and mixed results for NO\textsubscript{x}. Acetaldehyde emissions, but not other carbonyls, increased with increasing ethanol blend, and benzene and 1,3-butadiene emissions were lower for all blends including E85 (Karavalakis et al. 2012). Based on analysis of emissions certification data for flex fuel vehicles (FFVs) operated on both gasoline and E85, the use of E85 led to significant decreases in NO\textsubscript{x} and CO emission rates of 40% and 23%, respectively, for the FTP, and an increase of 19% for THC. However, based on real-world measurements of five FFVs using PEMS, tailpipe emission decreases were observed for all three of these pollutants (Delavarrafiee and Frey 2017).

Because GDI engines are rapidly growing in market share in the United States and elsewhere, many recent studies of fuel properties have focused on these types of engines. Changes in aromatic and olefin content appear to have little impact on fuel consumption rate. Regulated gaseous pollutant emissions are affected by aromatic content and T50. Decreases in aromatic content and T90 were associated with reductions in both primary particle mass and primary particle number (PN) emissions. Decreases in aromatics can also lead to reduced SOA formation, as discussed later. However, decreases in T50 were associated with an increase in particle mass emissions, although particle number emissions decreased slightly. Overall, aromatic content and T90 appear to be the most sensitive fuel properties (Zhu et al. 2017). A study of a GDI engine found that decreasing aromatic content led to lower PM mass and particle number (PN) emissions, as well as lower polycyclic aromatic hydrocarbon (PAH) emissions. Reduced olefin content appeared to reduce PM and PN emissions under high engine loads and to change the speciation of the PAHs. E10 was found to reduce PM emissions, but there was some increase in PN emissions at low engine load (Yinhui et al., 2016). Thus, characterization of emissions from GDI vehicles is likely to be conditional on the fuels used.

Fuel properties affect nonexhaust emissions. Evaporative losses depend on RVP, MSAT emissions, such as for benzene, depend on the fuel benzene content. U.S. gasoline benzene content was limited to 0.62 weight percent on an annual average beginning in 2011, based on concerns regarding benzene toxicity and carcinogenicity (EPA 2007). Ethanol can permeate through fuel lines and other parts more than nonethanol gasoline blends can (Kallio and Hedenvqvist 2010), which motivates a search for alternative materials to reduce such permeation (Manufacturers of Emission Controls Association [MECA] 2010).

Diesel engine-out NO\textsubscript{x} emissions are typically proportional to cetane number and oxygen content, and inversely related to fuel density (Singh et al. 2017). Higher cetane number indicates shorter times between injection of the fuel and its ignition. Cetane number is also associated with changes in engine-out PM emission rates.

Plant-based biodiesel fuel blends, such as soy-based B20 (20% volume transesterized soy oil, 80% petroleum diesel, by volume), which can lead to reductions in net CO\textsubscript{2} emissions, typically reduce engine-out emissions of CO, HC, and PM, related in large part to the higher oxygen content of the fuel (EPA 2002). Engine-out NO\textsubscript{x} emissions can be either higher or lower on B20 compared to ULSD, depending on the vehicle, but on average appear to be slightly higher by about 2%.

Fuel additives may also affect emissions. For example, based on laboratory measurements of a single-cylinder engine, ceria (CeO\textsubscript{2}) added to diesel fuel was found to reduce emissions of some pollutants (e.g., CO\textsubscript{2}, CO, total
Comparison of annual interstate variability in indirect stack emissions for electricity consumption (Frey 2016). Gasoline detergents helped reduce injector fouling in direct injection engines, which helps prevent increases in particle number emissions (Henkel et al. 2017).

Given the close coupling between fuel properties, engine design, engine operation, and emissions, there may be opportunities to optimize fuels and engines jointly as one system (Sarkar 2016). For example, gasoline engine efficiency can be increased if fuel octane increases. Expected future trends in road vehicle fuel consumption include a growing share of diesel fuel for heavy-duty vehicles. A global shift toward diesel, which is based on heavier distillates than gasoline, could lead to a surplus of naphtha and other low-octane gasoline components. Compression ignition engines could be developed to run on such fuels (Kalghatgi 2015).

Aside from fuels, engine lubricating oils can influence emissions, especially for crankcase losses and for some exhaust pollutants (Ireson et al. 2011). For example, engine lubricating oil was found to emit nonvolatile exhaust particles depending on oil Zn content (Pirjola et al. 2015). A key source of exhaust PM emissions from natural gas fired engines is lubrication oil (Hajbabaei et al. 2013). Lubricating oil-based additives and wear metals contributed to PM emissions from a CNG-fueled bus (Thiruvengadam et al. 2014).

**Electricity**

With the growing share of EDVs, the vehicle energy source is gradually shifting toward electricity. The effective emissions of these vehicles under battery power depend on the energy mix used for power generation (Figure 16). For example, a gasoline-fueled 2017 Honda Civic two-door coupe typically emits approximately 8,100 lb CO₂/year. A 2017 Chevrolet Bolt BEV could emit anywhere from approximately 10 lb CO₂/year assuming Vermont’s power generation mix to nearly 9,000 lbs CO₂ in West Virginia. The Bolt has CO₂ emissions comparable to or much less than that of the Civic in most states. A 2017 Tesla Model S would typically have CO₂ emissions lower than a 2017 Chevrolet Impala except for a few cases. There are some caveats to these comparisons. The actual power generation mix varies with time of day, typically with a higher fraction of coal and nuclear at night, and a higher fraction of natural gas during the day. Furthermore, the electrons that reach a charging station could come from a regional grid that transcends state boundaries. The interstate variability in energy mix is indicative of sensitivity of CO₂ emissions to varying proportions of coal, natural gas, nuclear, hydroelectric, wind, and solar. This comparison, however, does not account for the full life cycle of each energy source or vehicle.

Unless the intended contributions and longer term goals of the Paris Climate Agreement are adopted, EIA projects that the global electric power generation fuel mix will continue to be dominated by fossil fuels over the coming decades. Although much of the growth in power generation will be based on renewable energy sources, growth is also projected in the amount of power generated from natural gas, as well as some growth in the amount of power, although not the share, from coal (EIA 2017d). Even though renewables are likely to capture a major share of power generation, it may be decades before fossil fuel drops below 50% of total power generation. Thus, electrification of vehicles has the potential to shift some emissions (e.g., NOₓ, PM) from dense high-traffic areas to the downwind regions affected by power plant plumes.

**Engines and powertrains**

The fuel consumption of LDVs is affected by the fuel delivery system, engine design, transmission, and hybridization.

**Gasoline-fueled vehicles**

Vehicle transmissions have improved significantly in the last four decades. The two main reasons for improvement are reduction of mechanical losses (e.g., slippage) in automatic transmissions via torque converter lockup, and...
increasing the number of gears to allow better matching of torque and RPM to the optimal “sweet spot” of the engine efficiency map (NRC 2015). The average number of gears for passenger cars has gone from three to more than six for automatic transmissions from 1980 to 2017, while simultaneously the effectiveness of torque converters has improved. By 2010, the fuel economy of new U.S. vehicles with automatic transmissions matched, on average, that of vehicles with manual transmission. In the last two years automatic transmissions have had a slight (~1 mpg) fuel economy advantage. Manual transmissions now comprise less than 3% of U.S. new car sales. The share of continuously variable transmissions (CVTs) has rapidly grown in the last few years to more than 20% (EPA 2016e).

The mix of fuel delivery systems for new U.S. LDVs is undergoing dramatic changes with the rapid emergence of GDI. In the last 10 years, GDI has gained a growing share of the U.S. market, now exceeding 50% of new LDV sales (EPA 2016e). GDI delivers fuel directly into the cylinder, thereby eliminating fuel transport delay and enabling very precise control of timing and air/fuel ratio. Fuel can be delivered to better match engine load and with more than one injection pulse to achieve staged combustion (Zhao, Lai, and Harrington 1999). Staged combustion helps prevent knock, which enables an increase in compression ratio for improved engine efficiency. Estimates of the fuel economy advantage of GDI engine LDVs versus those with port fuel injection (PFI) range from 1.5% to 30% (NRC 2015). Spray-guided systems, which are referred to as second-generation GDI, produce fewer particles than the first-generation wall-guided systems (Short et al. 2017). This is largely because fuel spray in wall-guided systems impinges on piston and cylinder surfaces more so than for spray-guided systems, which leads to cooling of the fuel spray and more particle formation (Seo et al. 2016).

Some other technology trends include variable valve timing (VVT), cylinder deactivation for lower power demand driving situations, fuel cutoff during deceleration, and engine stop–start during deceleration and idle (EPA 2016e; NRC 2015). Stop–start reduces fuel use only during operational conditions when fuel consumption is typically already low. The reduction in fuel consumption for individual measures such as these ranges from approximately 0.4% to 2.5% (NRC 2015).

**Hybrid electric and plug-in hybrid vehicles**

Alternative power trains have emerged that offer substantial fuel economy benefits, among the most notable of which are the HEV and the PHEV. U.S. HEV sales have fluctuated in the last few years between 250,954 vehicles in 2011 and 536,383 in 2013 (Table 2). HEVs have an internal combustion engine (ICE), a traction battery (TB), and one or more motor-generators (MGs). The MGs can provide power to drive wheels and generate electricity during braking. The ICE of an HEV typically shuts off below levels of power demand that are dependent on speed and acceleration (Zhai, Frey, and Rouphail 2011). HEVs typically achieve better fuel economy in stop-and-go driving than they do in free-flow highway driving, in part because of regenerative braking.

Annual PHEV sales in the United States have increased in recent years, to more than 70,000 vehicles in 2016 and 2017 (Table 2). When the PHEV traction battery is fully charged from the electric grid, the vehicle can typically be driven for 12 miles to 53 miles, based on 2017 model year PHEVs available in the United States (Plug In America 2018), in charge-depleting mode before the first engine start. After exhausting the grid-based charge, a PHEV operates in charge-sustaining mode like an HEV.

**Other engines**

Compression ignition diesel engines are more efficient than spark-ignited gasoline engines because of higher compression ratio, lack of intake air throttling, and lean-burn mixtures. However, CO₂ emissions from burning a gallon of diesel fuel are 15% more than burning a gallon of gasoline, which partially offsets the efficiency advantage of diesel with respect to achieving CO₂ emission targets (NRC 2015). CNG and gasoline bifuel engines are spark-ignited, stoichiometric engines that typically include separate fuel injectors for each fuel. They are typically started on gasoline and can run on either fuel under load (Yao et al. 2014). Bifuel vehicles typically have a TWC. Older generations of CNG vehicles used lean-burn engines. However, the trend has been toward stoichiometric engines with TWC in recent years.

**Table 2. U.S. sales of nonconventional electrified powertrain vehicles from 2011 through 2017.**

<table>
<thead>
<tr>
<th>Year</th>
<th>Hybrid electric vehicles (HEVs)</th>
<th>Plug-in hybrid electric vehicles (PHEVs)</th>
<th>Battery electric vehicles (BEVs)</th>
<th>Fuel cell electric vehicles (FCEVs)</th>
<th>All (sum of HEVs, PHEVs, BEVs, and FCEVs)</th>
</tr>
</thead>
<tbody>
<tr>
<td>2011</td>
<td>250,954</td>
<td>6,966</td>
<td>9,074</td>
<td>19</td>
<td>267,013</td>
</tr>
<tr>
<td>2012</td>
<td>387,188</td>
<td>37,558</td>
<td>12,795</td>
<td>30</td>
<td>437,571</td>
</tr>
<tr>
<td>2013</td>
<td>536,383</td>
<td>41,376</td>
<td>46,832</td>
<td>19</td>
<td>558,593</td>
</tr>
<tr>
<td>2014</td>
<td>441,988</td>
<td>56,548</td>
<td>60,368</td>
<td>49</td>
<td>565,893</td>
</tr>
<tr>
<td>2015</td>
<td>365,732</td>
<td>49,118</td>
<td>64,175</td>
<td>75</td>
<td>479,100</td>
</tr>
<tr>
<td>2016</td>
<td>336,125</td>
<td>73,146</td>
<td>72,424</td>
<td>1,030</td>
<td>482,725</td>
</tr>
<tr>
<td>2017</td>
<td>365,320</td>
<td>91,724</td>
<td>96,261</td>
<td>1,862</td>
<td>555,167</td>
</tr>
<tr>
<td>Total</td>
<td>2,689,900</td>
<td>356,058</td>
<td>362,058</td>
<td>3,084</td>
<td>3,411,706</td>
</tr>
</tbody>
</table>

Fuel economy and greenhouse gas regulations

Seventy-eight percent of cars sold globally are subject to some form of fuel economy or CO₂ emission standard. Regulations and test procedures developed in the United States are used in countries such as Brazil, Canada, Mexico, and South Korea, whereas China and India employ procedures similar to those of the EU. From 2001 to 2014, there was a growing gap in the EU between the “official” CO₂ emission ratings and the real-world CO₂ emission rates, with real-world rates 10% higher in 2001 and more than 35% higher by 2014. During this time, the EU used the New European Driving Cycle (NEDC) for “type approval” chassis dynamometer testing, which has been criticized as failing to represent real-world driving conditions. Real-world CO₂ emissions in Japan have ranged from 25% to 45% greater than rated based on the JC08 cycle. In China, which has adopted the EU approach based on the NEDC, real-world rates increased from 10% to 25% higher than ratings from 2008 to 2014. In contrast, the U.S. fuel economy label values tend to accurately estimate real-world fuel consumption and CO₂ emission rates, on average, with a bias of 10% or less since the labeling standard was revised in 2008 (Tietge et al. 2017).

U.S. CAFE standards applied to sales-weighted fuel economy of light-duty passenger vehicles beginning with the 1978 model year. For passenger cars, the standard increased from 18.0 mpg in 1978 to 27.5 mpg in 1985 and, with some minor variations, was essentially unchanged through 2008. The CAFE standard for light-duty trucks was 20.2 mpg in 1991 (Atabani et al. 2011). Beginning with the 2009 model year, California promulgated its own vehicle GHG emissions standards, which were equivalent to a CAFE standard of 27.6 mpg for passenger car and light-duty truck 1 (PC/LDT1) vehicles, with a goal of 43.4 mpg by the 2016 model year.

In addition to CAFE standards, the United States has separate regulations that pertain to the rating and labeling of fuel economy for individual makes and models of vehicles (EPA and NHTSA 2011). The city rating was based on the FTP, and the highway rating was based on the Highway Fuel Economy Test (HFET). In 2008, the rating method was revised because real-world fuel economy was perceived to be worse (lower) than rated (EPA 2006, 2014d). Three additional tests were introduced, which include US06 and SC03 cycles and a cold-temperature FTP (20°F instead of 75°F). The five cycles represent a wider range of operating conditions than the original two-cycle method. Manufacturers can elect to use all five cycles or to calculate a downward-adjusted rating based on the FTP and HFET. A new rating scale for GHG emissions was introduced with the 2012 model year. Beginning with the 2017 model year, EPA updated parameter values in the rating calculation to more accurately rate high-fuel-economy vehicles, such as HEVs, for which fuel economy appeared to have previously been overestimated (EPA 2015a).

The current U.S. fuel economy standard requires average LDV fuel economy of 40.3–41.0 mpg by 2021 and 48.7 mpg to 49.7 mpg by 2025. The current U.S. LDV CO₂ emission standard of 163 g/mile is equivalent to 54.5 mpg if it were to be met only by fuel economy improvement of gasoline-fueled vehicles, but can be met with vehicles powered by other fuels or electricity. Furthermore, the standard is based on FTP and HFET test results, not the downward-adjusted fuel economy rating (EPA 2018b). The NRC found that the analyses conducted by EPA and NHTSA to develop the 2017–2025 model year fuel economy standards was “thorough and of high caliber.” A likely option for further improving LDV fuel economy is to replace naturally aspirated engines with downsized (smaller displacement) turbocharged engines (NRC 2015), and the market share of new LDGVs with turbocharged engines has grown from less than 5% in 2010 to more than 20% in 2016 (Davis et al. 2016). Effective strategies for meeting the current 2025 model year U.S. fuel economy target of 54.5 mpg could include either increased hybridization, increase in the 0 mph to 60 mph acceleration time, a decrease in interior volume, or combinations of these (Luk, Saville, and MacLean 2016; Whitefoot, Fowlie, and Skerlos 2017).

The U.S. federal government, under the Trump administration, has been identifying regulatory agency actions that “potentially burden the development or use of domestically produced energy resources.” Among such actions identified are the CAFE and GHG emissions standards. A Mid-Term Evaluation (MTE) of the standards was required and was completed by the Obama administration in 2016. However, under the Trump administration, the EPA and USDOT have announced that they will reconsider the standards as part of the MTE process (EPA 2017e). As of this moment, a decision has not yet been reached. Revisions, if any, to these standards could alter the future U.S. energy and CO₂ emission trajectories.

Real-world versus rated fuel economy

Labeled fuel economy estimates in the United States appear to be accurate compared to self-reported fuel economy data on the fueleconomy.gov website; however, there is substantial variability in self-reported fuel economy above and below the rated values. Thus, drivers might perceive the fuel economy rating to be
uninformative even though on average it is accurate (Greene et al. 2017).

Based on a sample of 122 light-duty gasoline vehicles measured under real-world driving conditions using PEMS, the rating scheme revised in 2008 was found to underestimate, by an average of 5% to 18%, city and highway fuel economy, respectively. However, real-world city fuel economy was lower than rated for 21% of the measured vehicles. Thus, there are likely to be drivers who experience fuel economy shortfalls and, therefore, who might complain that the ratings are inaccurate. Cold starts were found to reduce city average fuel economy by 4%, whereas air conditioner (AC) usage appeared to reduce fuel economy by 12–17%, with the percentage being higher for cycles with lower average power demand (Khan and Frey 2016).

Vehicle emissions

This section addresses some of the most commonly used technologies that affect tailpipe exhaust emissions, including GDI, TWC, hybridized vehicles, DOC, DPF, and SCR. Other factors that affect exhaust emissions, such as cold starts, ambient conditions, acceleration, grade, driving cycles, inspection and maintenance, and enforcement, are discussed. Finally, we offer observations regarding whether on-road emissions are consistent with emission standards. Additional information is given in the supplemental materials regarding exhaust gas recirculation (EGR), LNT, extended idling for long-haul trucks, emissions deterioration, nonexhaust emissions processes including crankcase, evaporative, and brake and tire wear emissions, eco-driving, vehicle load, air conditioning, and lubricating oil.

Technologies

This section briefly reviews and evaluates technologies for prevention or control of tailpipe exhaust emissions, and technological trends that affect emissions, with priority on LDGVs and HDDVs. The role of fuel composition and properties with regard to emissions is discussed in the preceding.

Gas direct injection versus port fuel injection gasoline engines

PM mass and particle number emission rates were higher for a GDI versus PFI Euro 4 passenger car tested on the NEDC (Liang et al. 2013). A GDI LDGV had BC emissions an order of magnitude higher than for PFI engine vehicles based on chassis dynamometer measurements, and the PFI emissions were elevated during high-speed driving (Zheng et al. 2017). At warm ambient temperature (30°C) and compared to a PFI vehicle, the GDI vehicle had higher emission rates for THC, PM mass, and solid particle number (SPN). However, at cold ambient temperature (7°C), the GDI vehicle had lower CO, THC, and PM mass emission rates and similar SPN rates. Thus, GDI may adapt better to cold conditions than PFI. The GDI vehicle was more fuel efficient and therefore had lower CO₂ emissions, in all conditions (Zhu et al., 2016). BC emissions from vehicles measured on the FTP and US06 for wall-guided GDI were highest compared to spray-guided GDI and PFI. In contrast, spray-guided GDI BC emissions were lower on the US06 compared to PFI (Bahreini et al. 2015).

To mitigate the higher particle emissions of GDIs compared to PFIs, GPFs have been proposed to address both PM mass and particle number emissions (Guan et al. 2015; Mamakos et al. 2013), although they are more likely to be required in markets that use the Euro 6 or similar PN standards (DieselNet 2017). A prototype GPF, based on a design similar to a DPF, reduced BC emissions by 73% to 88% on the FTP and by 59% to 80% on the US06, depending on temperature (Chan et al. 2014). GPFs are estimated to add approximately $100 to the cost of a vehicle (Minjares and Sanchez 2011).

Catalytic converter

The evolution of catalytic reduction of LDGV emissions began with oxidation catalysts, focused on control of CO and VOCs, followed by addition of dual-bed catalysts that could perform both oxidation and reduction of NOₓ to N₂, followed again by single-bed “three-way” catalysts (TWCs) that can perform oxidation of CO and VOC and reduction of NOₓ simultaneously (Collins and Twigg 2007).

The TWC, coupled with computer-based control of fuel injection systems, was fully implemented in new U. S. cars as of the 1996 model year. These technologies led to greater than 90% reductions in VOC, CO, and NOₓ tailpipe exhaust emissions (Koltsakis and Stamateles 1997; Sawyer 2010). In 2007, the TWC was estimated to have prevented emissions of 4 billion tons of HC, 4 billion tons of NOₓ, and 40 billion tons of CO in the United States (Mooney 2007). However, TWCs are not fully effective until they reach their “light-off” temperature, which leads to a cold-start effect (Jeong and Choi 2002). Catalysts with increased thermal stability can be placed closer to the engine exhaust manifold, thereby shortening the time required to reach “light-off” temperature (Collins and Twigg 2007).

Once the engine has warmed up, the air/fuel ratio is typically controlled based on feedback from oxygen sensors mounted before and after the TWC (Rajagopalan et al. 2014). At high engine load, with high flow rate of engine-out CO and HC emissions,
the exothermic CO and HC oxidation reactions can lead to catalyst overheating. To prevent permanent catalyst damage from sintering, the electronic control unit commands a fuel-rich air/fuel mixture, which reduces the availability of oxygen and thus curtails the oxidation reactions (Eriksson and Nielsen 2014). This fuel “enrichment” operating mode leads to short episodes (usually a few seconds) of very high tailpipe CO emissions. Fuel cut-off after acceleration contributes to deactivation of catalyst and therefore poses challenges to the formulation of durable catalysts (Johnson 2015).

TWCs are applicable to stoichiometric burn engines, including PFI and GDI. TWCs are applicable to other fuels as well, such as ethanol blends, LPG, LNG, and CNG (Bishop et al. 2012; Collins and Twigg 2007; Dardiotis et al. 2015; Lyu et al. 2016; Yoon et al. 2014). TWC-equipped CNG goods movement vehicles were found to emit lower NO\textsubscript{x} emissions than diesel trucks equipped with SCR, by as much as 96% (Misra et al. 2017; Thiruvengadam et al. 2015). CNG vehicles with TWC were found to have low NO\textsubscript{x} emissions over a range of observed driving cycles and nondetectable aromatic emissions but to emit carbonyl compounds during cold-start and low-power demand operation such as extended idle and creep. Because TWC typically requires a slightly fuel-rich engine mixture, ammonia can be produced in the TWC. The ammonia emissions were approximately 1 g/mi\textsuperscript{e} (Thiruvengadam et al. 2016).

Compared to an earlier generation of buses with lean-burn CNG engines and oxidation catalysts, based on chassis dynamometer measurements, stoichiometric CNG buses with TWC reduced emissions of VOCs and carbonyls by 99% and PAH by 95%, while also substantially reducing emissions of NO\textsubscript{x}, PM, CO, and THC (Yoon et al. 2014). Another dynamometer-based study also found that a CNG bus with TWC had lower NO\textsubscript{x} and THC emissions than lean-burn CNG buses, but higher CO and NH\textsubscript{3} emissions (Hajbabaee et al. 2013). The contribution of methane to total exhaust gas GHG emissions varied from 1.4% to 5.9% depending on route, based on on-road measurements of a CNG truck tractor with TWC (Quiros et al. 2017).

Hybrid and plug-in hybrid electric vehicles

Although there have been many studies of the fuel economy of hybrid vehicles, there are few studies of their real-world emissions. Five HEVs and one PHEV that were Euro 5 compliant were measured on a chassis dynamometer on the World harmonized Light-duty Test Cycle (WLTC) at −7°C and 23°C. Emissions of CO, NO\textsubscript{x}, HC, and PN were lower than for conventional Euro 5 LDGVs. Their emissions of ammonia, ethanol, and acetaldehyde were similar to those of conventional LDGVs (Suarez-Bertoa and Astorga 2016). Two HEVs measured with PEMS in Macao were found to have lower CO\textsubscript{2} and NO\textsubscript{x} emission rates compared to conventional LDGVs, with CO\textsubscript{2} emission rates being less sensitive to changes in speed than for conventional vehicles (Wu et al. 2015). In general, HEVs appear to have lower CO\textsubscript{2}, NO\textsubscript{x}, HC, and PN emissions, on average, compared to comparable conventional vehicles.

Other light-duty vehicle powertrains

PFI bifuel gasoline–CNG taxis were found to have 54% to 83% lower black carbon emissions when operating on CNG (Wang et al. 2016). However, CO emissions were higher for high speed and high acceleration than for other driving modes, although there was not a significant average difference compared to conventional vehicles. NO\textsubscript{x} emissions were up to 20% higher than for gasoline vehicles. HC emissions were higher when bifuel vehicles burned gasoline compared to CNG (Huang et al. 2016). HC and NO\textsubscript{x} emissions were also found to be higher than for conventional vehicles in a PEMS study of Euro 2 and 3 bifueled taxis fueled with CNG. CO\textsubscript{2}, CO, and HC emissions rates were 60–300% higher for urban than freeway conditions, depending on the pollutant. Although average CO rates were below the level of the standard, HC and NO\textsubscript{x} emission rates were much higher than the standard (Yao et al. 2014). A Euro 5 bifueled passenger car operated on CNG on a chassis dynamometer comfortably met emission rates comparable to the Euro 6 standard (Bielaczyc, Woodburn, and Szczotka 2014).

Based on three LDDVs, NEDC and WLTC cycle average BC emission rates were typically one to two orders of magnitude higher than for GDI or PFI LDGVs, and BC accounted for 38% to 54% of total PM mass emitted (Zheng et al. 2017). These diesel vehicles had DOC but not DPF. Thus, later model year LDDVs with DPF are likely to have much lower BC emissions.

Diesel oxidation catalyst

The current state-of-practice in emissions controls for HDDVs certified under current standards in the United States and Europe, and similar standards in other parts of the world, is focused on control of NO\textsubscript{x} and PM emissions using DOC, DPF, SCR, and ammonia slip catalyst (ASC) (Walker 2016). The DOC oxidizes a portion of NO to NO\textsubscript{2}, 10% to 60% of CO, and 40% to 75% of HC. DO\textsubscript{C} oxidize 20% to 40% of engine-out PM, mostly related to the soluble organic fraction from unburned fuel and lubricating oil (EPA 2010b). Thus, DOCs are not effective in oxidizing BC. The DOC increases the exhaust gas temperature. Higher exhaust gas temperatures improve the performance of downstream emission control devices such as the DPF and SCR (Walker 2016).
Diesel particle filter

In a DPF, NO₂ in the exhaust reacts with deposited soot, which is comprised mainly of combustible material such as BC. In-use oxidation of soot with NO₂ is “passive regeneration” and enables removal of soot before it accumulates on filter surfaces. In case passive regeneration is not adequate to remove the soot layer, DPFs are designed with an option of active regeneration (Walker 2016). Active regeneration involves increasing exhaust temperature by running the engine with a more rich (lower air/fuel ratio) mixture or by directly injecting fuel into the exhaust. The DOC helps oxidize incomplete products of combustion or directly injected fuel to raise the exhaust temperature, which leads to more rapid oxidation of soot deposited in the DPF. Thus, unlike the DOC, the DPF is very effective at controlling BC emissions.

DPFs are capable of removing more than 95% of particle mass and 99% of particle number (Guan et al. 2015; Herner et al. 2009). DPF regeneration can lead to episodes of high emissions of some species depending on design. For example, DPF systems based on active regeneration appear to have much higher emissions of PAH during regeneration than systems based on passive regeneration (Hays et al. 2017). Noncombustible ash that accumulates in the DPF needs to be cleaned every 6–12 months (EPA 2010c), introducing a need to ensure that this maintenance is done by vehicle owners. There can be substantial variability in emissions during regeneration. For example, measured regeneration emissions ranged from 0.4 g to 37 g of PM₂.₅ for a 2007 truck and 0.2 g to 2.5 g for a 2010 truck. The particle size distribution for both trucks was dominated by nucleation mode particles less than 50 nm in diameter, particularly for the newer truck (Yoon et al. 2015). Particle number emissions of a 2013 truck during active regeneration were approximately 10 times higher than during passive regeneration (Wang et al. 2017a).

PN emissions of a 2013 truck were higher for highway driving than lower engine load driving situations, most likely because volatile nucleation mode particles were formed at DPF outlet temperatures over 310°F. Original equipment manufacturer (OEM) DPFs appear to be more effective than retrofitted DPFs, with substantially lower postcontrol emissions (Wang et al. 2017a).

Using a mobile measurement laboratory that sampled on-road pollutant concentrations (Westerdahl et al. 2005), fuel-based emission rates for NOₓ and BC were 40% lower for 2011 versus 2009 vehicles, but approximately the same for ultrafine particles (UFP; typically less than 0.1 μm) (Kozawa et al. 2014). The latter is consistent with findings in other studies that DPF-equipped vehicles emit a large fraction of nucleation mode particles. Furthermore, the NO₂/NOₓ ratios measured on the road increased from 0.23 to 0.30. For comparison, the NO₂/NOₓ ratio for LDGVs with TWC is typically around 0.05 (Carslaw and Rhys-Tyler 2013; Wild et al. 2017).

Solid UFP emissions were reduced by 99.8% for older urban and interurban buses retrofitted with a DPF. CO emissions were also lowered by as much as 80% because of the oxidation catalyst coating in the DPF, whereas there was little change in total NOₓ emissions (Tartakovsky et al. 2015).

Nitrous acid (HONO) emissions may be enhanced by DPF equipped vehicles. HONO is an important source of ambient hydroxyl radicals that affect production of photochemical oxidants. Based on a fixed-site air quality monitor in Hong Kong, ambient concentrations of HONO were found to vary seasonally and the measured HONO/NOₓ ratio was strongly correlated with BC, which is a marker for diesel particle emissions (Xu et al. 2015). Based on a tunnel study in Hong Kong by a different group of investigators, a similarly high ratio of HONO to NOₓ correlated with BC was observed (Liang et al. 2017).

Euro 6 LDDVs equipped with DPF measured on a dynamometer were found to emit fewer particles but more NOₓ and carbonyl compounds than comparable LDGVs (Martinet et al. 2017).

GHG exhaust emissions from diesel vehicles without aftertreatment were mostly CO₂ with up to 0.11% CH₄ and up to 0.27% N₂O on a CO₂-equivalent basis. DOC and DPF aftertreatment increased N₂O emissions but did not significantly change CH₄ emissions (Graham et al. 2008). For diesel vehicles equipped with DOC, DPF, and SCR, N₂O emissions were found to account for 2.6% to 8.3% of the tailpipe CO₂ equivalent tailpipe emissions (Quiros et al. 2017). DPFs substantially reduce BC emissions, which mitigates short-term climate effects (EPA 2012).

Selective catalytic reduction

Higher NO₂/NOₓ ratios favor lower temperature SCR conversion of NOₓ to N₂. A urea solution, referred to as “diesel exhaust fluid” (DEF), is injected upstream of the SCR reactor, which releases ammonia (and some CO₂) that mixes with the exhaust gas. Ammonia reacts with NO and NO₂ to form N₂ and H₂O. The ASC is designed to selectively oxidize ammonia to N₂ without creating excessive amounts of N₂O or NOₓ (Walker 2016).

Particle emissions from a HDDV with SCR but without DPF included volatile and nonvolatile nucleation mode particles of 23 nm or less. The nucleation particle emissions more than doubled for uphill driving and when exhaust temperature exceeded 350°C (Saari et al. 2016).

Based on “sniffing” exhaust plumes from trucks moving beneath an overpass, the NOₓ, BC, and particle number emission rates for 2010–2013 model year
trucks averaged 69%, 92%, and 66% lower compared to 2004–2006 model year trucks. The newer trucks were equipped with DPF and SCR, whereas the older trucks were not. Tailpipe NO\textsubscript{2} emissions were higher for the DPF- and SCR-equipped trucks (Preble et al. 2015).

Based on comparisons of measured emissions from an engine with no controls, with DPF, and with SCR, SCR and DPF were each found to help reduce emissions of CO, HC, and PM mass, with different control efficiencies. For example, SCR reduced PM emissions by 19%, whereas DPF reduced PM emissions by 98%. The DPF was found to control particle emissions of elemental carbon by 99.7%, organic matter by 93%, and trace metals by 94%. Both SCR and DPF reduced PAH emissions by 85% or more, and reduced emissions of other species such as aldehydes (Liu et al. 2008).

Although SCR is very effective in controlling exhaust NO\textsubscript{x} emissions when the exhaust temperature is high enough, it is less effective at relatively low exhaust temperatures under cold start, extended idling, and low-load, low-speed driving (Herner et al. 2009; Misra et al. 2013; Thiruvengadam et al. 2015).

In a field study of 11 SCR-equipped buses in Beijing, China, three were not injecting urea. Furthermore, there was no indication of failure of the urea injection system based on the on-board diagnostic (OBD) system (Guo et al. 2014). This raises concerns about both the efficacy of OBD checks as a way to identify failures and the possible frequency of failures of postcombustion controls.

The effectiveness of SCR on diesel hybrid buses may be less than for conventional buses, related to lower exhaust temperatures. Based on PEMS measurements of two hybrid and two conventional diesel transit buses, the hybrid buses were found to have lower engine-out NO\textsubscript{x} emissions but higher tailpipe exhaust NO\textsubscript{x} emissions, on a mass per time basis. The hybrid buses emitted less CO on a mass per time basis (Guo et al. 2015).

**Other heavy-duty vehicles**

PN emissions from a diesel-hybrid bus with SCR were seven times higher during acceleration than during steady-speed driving (Soylu 2015). Based on curb-side measurements of plumes of passing buses, UFP emissions were compared for CNG and diesel. The UFP mass emissions for CNG were only 9% of those from the diesel buses, but the particle number concentration was 6 times higher and 82% of the CNG emitted particles were volatile, whereas only 38% of the diesel emitted particles were volatile. The CNG emissions were most pronounced during acceleration. The CNG UFP were mainly in the nanoparticle size (Jayaratne et al. 2008).

CNG buses without after treatment had higher CO, HC, NMHC, VOC, and carbonyl compound emissions compared to diesel buses. Diesel vehicles without after-treatment had higher PM and PAH emissions. With after-treatment, NO\textsubscript{x} and other pollutant emissions were more similar between vehicle fuel types. However, stoichiometric CNG engines with TWC had lower NO\textsubscript{x} emissions. CO\textsubscript{2} emissions were typically lower for CNG vehicles than for diesel, related to the lower carbon content of natural gas, even though diesel engines are typically more energy efficient (Hesterberg, Lapin, and Bunn 2008).

**Cold start**

As hot stabilized emissions decrease as a result of improved technology and increasingly stringent exhaust emission standards, the relative contribution of cold starts to trip emissions could increase, perhaps to the point at which cold start-emissions are the dominant source of on-road tailpipe emissions (Drozd et al. 2016). Cold-start emissions typically occur within the first 200 seconds of certification tests such as the FTP (Koltsakis and Stamatelos 1997). A cold start is an engine start that takes place after the engine, including the engine block, engine coolant, and engine lubricating oil, has equilibrated to ambient temperature. Postcombustion controls, such as catalytic converters and SCR, are not fully effective until they achieve a “light-off” temperature.

The duration and intensity of the cold-start effect are more pronounced for colder ambient temperatures. Engines may be commanded by the electronic control unit to run more fuel-rich to achieve engine starts under cold temperatures, which can lead to a higher proportion of unburned fuel and other HC and VOC emissions. The sum of all VOC emissions increased for cold (−7°C) versus warm (24°C) ambient temperature cold starts by a factor of 7, 9, and 23 for E0, E10, and E85 fuels, respectively (George et al. 2015). Cold-start emissions of BC particles for PFI LDGVs occurred during the first 47 seconds to 94 seconds of driving cycles measured on a chassis dynamometer, and accounted for up to 76% of NEDC and 36% of WLTC cycle total BC emissions (Zheng et al. 2017). Cold starts are subject to increased SVOX exhaust emissions from light- and medium-heavy-duty diesel vehicles (Hays et al. 2017).

Cold starts can lead to high emissions of aromatic and carbonyl compounds, based on chassis dynamometer measurements of Euro 6 LDDVs and LDGVs (Martinet et al. 2017). MSAT emissions during real-world cold-start measurements for one LDGV were up to two
orders of magnitude higher than during hot stabilized operation (Sentoff, Robinson, and Holmén 2010).

Based on engine dynamometer measurements of a diesel engine without postcombustion control, the engine-out emissions during cold start were found to be lower by 15%, 48%, 44%, and 63% for NO$_x$, PN, PM$_1$, and PM$_{2.5}$ (Zare et al. 2017). However, for a medium- to heavy-duty diesel truck with DPF and LNT, cold ambient temperature cold-start BC and PNC tailpipe emissions were two to three times higher than for warm starts or higher ambient temperatures (Book et al. 2015).

**Effects of vehicle operation**

In the decades since the FTP was first adopted as the foundation of fuel economy and emissions certification testing for LDVs, there has been growing recognition that no single driving cycle can adequately account for variability in vehicle fuel use and emission rates. With the emergence of new measurement technologies, such as remote sensing and PEMS, the capability to measure vehicles under actual driving conditions has grown. This enables characterization of a myriad of factors that can affect “microscale” emissions on a second-by-second basis, as well as “mesoscale” trip-based emissions.

**Microscale emissions: speed, acceleration, and road grade**

Power demand related to vehicle tractive effort and, therefore, fuel consumption can be well approximated by vehicle specific power (VSP) (Jiménez-Palacios 1999):

$$VSP = v(a(1 + \varepsilon) + gr + gC_R) + \rho v^3 \left( \frac{C_D A}{m} \right)$$

where $a$ is vehicle acceleration (m/sec$^2$), $A$ is vehicle frontal area (m$^2$), $C_D$ is aerodynamic drag coefficient (dimensionless), $C_R$ is rolling resistance coefficient (dimensionless, ~ 0.0135), $g$ is acceleration of gravity (9.8 m/sec$^2$), $m$ is vehicle mass (in metric tons), $r$ is road grade, $v$ is vehicle speed (m/sec), VSP is vehicle specific power (kw/ton), $\varepsilon$ is a factor accounting for rotational masses (~ 0.1), and $\rho$ is ambient air density (1.207 kg/m$^3$ at 20°C).

The vehicle activity parameters for VSP are speed, acceleration, and grade, and are typically measured at 1 Hz frequency. The product of speed and acceleration is related to the change in kinetic energy, and is adjusted to account for energy related to rotational masses in the engine and powertrain. The product of speed and grade is related to the change in potential energy as the vehicle climbs or descends hills. The product of speed and rolling resistance coefficient is related to frictional losses between the tires and pavements and energy losses from tire flexing. Speed to the third power is related to the power needed to overcome aerodynamic drag and can be significant at high speed. Although not included in the preceding, horizontal curves that do not meet design standards for minimum radius can lead to excess energy consumption (Ko 2015).

For carbonaceous fuels burned at high combustion efficiency, the relative trend in CO$_2$ emission rate is similar to the relative trend in fuel use. The trend for CO$_2$ emission rate is given in Figure 17a, based on VSP modal definitions given in Table 3 and PEMS measurements of 50 Tier 2 passenger cars. These data are based on methods given in earlier work (Frey, Zhang, and Roupail 2010; 2008; Sandhu and Frey 2013). Fuel use and CO$_2$ emission rates typically increase linearly with positive VSP, are the lowest at idle, and may be slightly higher than the idle rate during braking and deceleration. The latter is typically because most engines are engaged to the drive wheels via the transmission and are operating at elevated RPM compared to idle. The trends in emission rates of other pollutants vary. For example, CO emission rates for Tier 2 LDGVs tend to be very low except at high power demand, especially for the largest VSP mode, because of fuel-enrichment and open-loop operation (Figure 17b). HC emission rates tend to

![Figure 17](image-url). Vehicle specific power modal average rates and 95% confidence intervals based on the average of 50 Tier 2 gasoline passenger cars: (a) CO$_2$ and (b) CO emission rates.
increase approximately linearly with increasing VSP. NO\textsubscript{x} emission rates tend to increase nonlinearly with increasing VSP. For all of these pollutants, the highest time-based average emission rates occur at the highest VSP mode.

There are many impacts of variation in VSP. For example, based on chassis dynamometer measurements of an LDDV, PM and PN emissions increased with acceleration and simulated road grade (Kim et al. 2017). Based on chassis dynamometer measurements of eight HDDVs, NO\textsubscript{x} emissions from engines with cooled EGR were 239\% higher for stop-and-go driving compared to steady speed (Dixit et al. 2017). Based on measurements of 78 LDVs in Bogota, Colombia, using PEMS, VSP was found to explain a substantial portion of variability in measured emission rates (Rodríguez et al. 2016).

The optimal cruising speed for conventional diesel bus fuel economy was estimated from a model that was calibrated to real-world data and found to be between 40 km/hr and 50 km/hr, which is lower than the typical optimal range for LDGVs of 60 km/hr to 80 km/hr (Wang and Rakha 2016). There is not much reported, however, on optimal cruising speeds for various pollutants based on measurements.

**Mesoscale emissions: cycle average rates**

Cycle average rates are sensitive to many factors, including intervehicle variability, interroute variability, variability in cycle average speed, and variability in the distribution of VSP. These interactions are illustrated in Figure 18 for intervehicle and interroute variability, and in Figure 19 for variability related to cycle average speed. These data are for the same vehicles shown in Figure 17. Driving cycles on these same routes were measured for more than 150 other vehicles. LDGVs of various body types (e.g., sedan, SUV) and engine sizes typically produce similar distributions of speed and acceleration on these four routes (Liu and Frey 2015a). Therefore, cycle average rates were estimated for an average vehicle based on the average VSP modal rates given in Figure 17 and each of 806 measured real-world cycles. These cycles were measured on typical commuting routes in the Raleigh, NC, area (Frey, Zhang, and Rouphail 2008). The routes are comprised of varying mixes of minor arterial, major arterial, freeway, and ramp road types with speed limits.

![Figure 18](image-url)
Cycle average rates versus cycle average speed for 543 and CO emission rates vary by approximately a factor of 2 from just over 200 g/mile to over 92 g/mile, depending on the distribution of VSP. This range of variation at a given average speed illustrates that the lowest fuel use rate can be approximately 13% lower than the highest fuel use rate, which is typical of estimates of potential energy savings achievable via eco-driving. At 50 mph cycle average speed, the lowest estimated cycle average CO emission rate is 56% lower than the highest estimated rate. CO emission rates are very sensitive to the distribution of VSP, especially high values of VSP, related to the frequency of fuel enrichment. Real-world driving cycles may vary with driving behavior (Tanvir, Frey, and Rouphail 2018). Measurements of naturalistic driving (e.g., Shrestha, Lovell, and Tripodis 2017) are needed from which to assess spatial and temporal variability in vehicle energy use and emissions, including the potential for differences between study areas related to factors such as terrain, fleet mix, road types, and driving style.

Intervehicle variability among the 50 Tier 2 passenger cars is quite substantial for the cycle average CO2 and CO emission rates (Figure 18). For CO2 emission rates, intervehicle variability is inversely related to fuel economy. The cycle average CO2 emission rates vary by approximately a factor of 2 from just over 200 g/mile to over 400 g/mile. There is also variability between routes, with the higher speed Route 1 cycle, of which more than 90% of driving is on freeways, having mileage-based emission rates lower by an average of 18% than the lower speed Route A cycle, which is entirely on arterial roads.

For CO, the cycle average emission rates vary by more than two orders of magnitude. For Route 1, the mean CO emission rate is 1,010 mg/mile with a 95% confidence interval of 620 mg/mile to 1,400 mg/mile. For Route A, the mean CO emission rate is 630 mg/mile, with a 95% confidence interval of 400 mg/mile to 860 mg/mile. However, there is positive correlation between these two distributions. The difference in emission rates for Route 1 less that of Route A has a mean of 380 mg/mile with a standard error of only +90 mg/mile, for a 95% confidence interval on the mean difference of 200 mg/mile to 560 mg/mile. Because of the pairing of vehicles among the routes, there is more confidence in the estimated difference than would be implied by assuming that the sampling means were uncorrelated. Therefore, despite the large variability in the sample, Route 1 has a statistically significantly higher rate than Route A, with a mean difference of a 60% higher rate on Route 1 versus Route A. This example illustrates that relative comparisons can be more robust to intervehicle variability than estimates of absolute rates.

Factors affecting variability in driving cycles
As shown in Figure 19, the mean fuel use rate per mile decreases from the lowest observed cycle average speed of 15 mph to a minimum at approximately 50 mph, and then increases slightly with increasing cycle average speed. At 50 mph, fuel use rates vary from approximately 80 g/mile to 92 g/mile, depending on the distribution of VSP. This range of variation at a given average speed illustrates that the lowest fuel use rate can be approximately 13% lower than the highest fuel use rate, which is typical of estimates of potential energy savings achievable via eco-driving. At 50 mph cycle average speed, the lowest estimated cycle average CO emission rate is 56% lower than the highest estimated rate. CO emission rates are very sensitive to the distribution of VSP, especially high values of VSP, related to the frequency of fuel enrichment. Real-world driving cycles may vary with driving behavior (Tanvir, Frey, and Rouphail 2018). Measurements of naturalistic driving (e.g., Shrestha, Lovell, and Tripodis 2017) are needed from which to assess spatial and temporal variability in vehicle energy use and emissions, including the potential for differences between study areas related to factors such as terrain, fleet mix, road types, and driving style.

It is difficult to isolate the effect of driver behavior in real-world measurements because there can be simultaneous variability in traffic conditions. Differences in driving cycles among 82 drivers observed on low-traffic rural roads were interpreted as more representative of personal driving style than for urban roads, for which traffic may have constrained preferred driving styles (Sentoff, Aultman-Hall, and Holmén 2015). Other driver choices can affect emissions. These including cabin comfort, trip planning, load management, fueling, and maintenance (Sanguinetti, Kurani, and Davies 2017). Many of these factors are not well quantified based on scientifically conducted studies.

Inspection and enforcement
The purpose of inspection and maintenance (I/M) programs is to encourage and enforce compliance with applicable vehicle emission standards.

In the 1990s, seven heavy-duty diesel engine manufacturers used computer-based strategies to adjust fuel injection timing during real-world driving in a manner that
deviated from how the engines were computer controlled during certification testing. The U.S. government alleged that these strategies increased NO\textsubscript{x} emissions and constituted a defeat device prohibited under law (e.g., USA v. Cummins 1999). The U.S. government, the state of California, and the manufacturers entered into consent decrees that, among other things, imposed new procedures to verify “not to exceed” (NTE) emission limits for NO\textsubscript{x}, NO\textsubscript{x} + NMHC, and PM. A period of time was allowed to develop and implement new test procedures (California Air Resources Board [CARB] 2000). The new test procedures were finalized in 2005, and include the option of conducting NTE tests on an engine dynamometer or under field conditions using a PEMS in accordance with CFR 1065 Subpart J (EPA 2005).

In June 2016, the U.S. government and Volkswagen (VW) agreed to the first of several consent decrees in which VW admitted that software enabled the vehicle electronic control module (ECM) “to detect when the vehicles are being driven on the road, rather than undergoing Federal Test Procedures,” that the “software renders certain emission control systems . . . inoperative,” and that the software resulted in “emissions that exceed EPA-compliant and CARB-compliant levels when the vehicles are driven on the road” (Cruden, Van Eaton, and Engel 2016). VW sold 590,000 LDDVs in the United States with this “defeat device” during the 2009–2016 model years. The costs for recall and modification or replacement of affected vehicles exceed $10 billion, plus VW agreed to pay $2.7 billion to a mitigation fund to offset lifetime excess emissions, $2.0 billion for electric vehicle charging stations and related efforts, $1.45 billion in civil penalties, and $2.8 billion in criminal penalties (EPA 2018). The defeat device issue also occurred with 3.0-liter LDDVs, but the penalties were primarily associated with the larger number of affected 2.0-liter vehicles. Excess NO\textsubscript{x} emissions from VW LDDVs that were 10–40 times the level of the U.S. tailpipe emission standard were first identified based on PEMS testing of two vehicles (Thompson et al. 2014), after which the EPA and CARB confirmed the findings. The scope of the “VW scandal” was international and affected as many as 11 million vehicles in the United States, Europe, Asia, and Australia (Anenberg et al. 2017).

There is growing recognition, particularly with respect to LDDVs in Europe, that real-world emission rates are generally higher than the level of the applicable European “type approval” tests. European type approval regulations have been adopted in full or in large part by many countries, and more than 70% of new vehicles sold globally are subject to such regulations. Based on emission estimates for 11 vehicle markets, approximately one-third of on-road diesel NO\textsubscript{x} emissions are in excess of the level of the applicable emission standard (Anenberg et al. 2017). The European Commission has implemented a new requirement for “real driving emissions” (RDE) measurements with PEMS (EU 2018). China has adopted a PEMS-based standard to prevent excess NO\textsubscript{x} emissions from heavy-duty vehicles under the China 5 standard, which is modeled on the Euro 5 standard (ICCT 2017) and will include assessment of RDE for LDVs under the China 6 standard (Cui 2018; Yang 2018).

These recent examples of high real-world emissions illustrate the need for procedures by which to prevent and detect such emissions. In the United States, areas that have difficulty complying with air quality standards influenced at least in part by vehicle emissions are required to implement I/M programs (Eisinger and Wathern 2008). I/M programs can be effective in reducing real-world emissions. For example, the California Smog Check program was estimated to reduce tailpipe emissions of LDGVs by 26%, 34%, and 14% for HC, CO, and NO\textsubscript{x}, respectively, in 1999. California’s program required dynamometer testing on a transient driving cycle. Most of the benefits of the program were attributed to repair of vehicles that failed a test, as well as maintenance and repair prior to inspection. Most of the avoided emissions were for vehicles more than 10 years old (Singer and Wenzel 2003). As emissions of properly functioning vehicles decrease, the contribution of malfunctioning vehicles to total real-world emissions may increase, even if the fraction of malfunctioning vehicles decreases (Pang et al. 2015).

In 1992, EPA issued regulations that required all I/M programs to use dynamometer-based tests at test-only facilities. However, as a result of backlash, by 2001 EPA abandoned the test requirement in favor of less burdensome OBD system checks. The OBD-II standard went into effect with the 1996 model year. The EPA and several states conducted studies to compare the outcome of OBD system checks to that of emission tests for 1996 and newer model year vehicles. These tests revealed significant “lack of overlap” regarding which vehicles failed the OBD versus emissions test. For example, in Colorado, only 66 vehicles failed both tests even though nearly 3,000 failed the OBD test and nearly 400 failed the emissions test. Furthermore, because OBD system checks can be done only for 1996 and newer model year vehicles, I/M programs based on such checks exclude older vehicles. Older vehicles are more likely to be high emitters than newer vehicles (Eisinger and Wathern 2008). With the passage of time, the on-road fleet will include fewer pre-1996 vehicles, and in another decade or so the concern about not being able to test all model years will largely disappear.

Just as emission control systems can deteriorate and fail, so too might the OBD system. Based on comparison of remote sensing measurements and OBD tests,
the percentage of vehicles that had high real-world emissions but passed the OBD test increased from 16% for 3-year-old vehicles to 35% for 9-year-old vehicles. OBD system malfunction is implicated as a leading cause for this trend, although other factors are possible. Ineffective repairs of previously failed vehicles or tampering to defeat the emission control system, such as by clearing OBD system codes or installing an oxygen simulator that overrides the actual oxygen sensor, could lead to high emissions (Supnithadnaporn et al. 2011). Thus, some argue that OBD checks should supplement but not replace an emissions measurement-based inspection (Eisinger and Wathern 2008).

Remote sensing has been considered as a potentially useful tool for I/M programs, but there have been difficulties with it in practice. There can be substantial variability in measurements for a given vehicle that are not because of a failure. Remote sensing measurements are a snapshot of less than 1 sec. At such a short averaging time, emissions can be very sensitive to speed, acceleration, grade, and transients in operation. Although remote sensing sites, such as freeway on-ramps with positive grade, are typically selected so that there is some load on the engine, there is nonetheless variability in vehicle operation that contributes to variability in emissions. For example, remote sensing was used to measure CO and HC emission factors at Raleigh–Durham International (RDU) airport for transit buses that circulated between the terminal and parking areas. Measurements were made on a one-lane ramp with positive road grade. There were 25–28 measurements of the same bus for three buses. The CO emission rates varied by a factor of five or more, and the HC emission rates varied more than order of magnitude (Frey and Eichenberger 1997). Variability in emission rates for individual vehicles may lead to false positive or false negative findings in an I/M screening based on remote sensing (Wenzel et al. 2004). Based on analysis of remote sensing data, fuel-based emission factors for CO, NO, HC, and PM were found to be poorly correlated with each other, implying that I/M would be needed for each pollutant separately (Mazzoleni et al. 2004). Some proposals for remote sensing for I/M recognize the need to accumulate repeated measurements for more reliable identification of high emitters (Borken-Kleefeld and Dallmann 2018). Remote sensing has also been proposed as a clean screening tool to identify vehicles that could be exempted from an upcoming I/M test. Based on an analysis of data from the 1990s, simply exempting newer model year cars would be more effective than using remote sensing as a screening tool (Wenzel and Sawyer 1998).

Inspection of HDDV emissions in the United States and other countries has typically focused on quick tests that can detect excessive particulate matter emissions based on a free acceleration smoke (FAS) test. Under a “snap acceleration,” also referred to as “snap idle,” the throttle is held fully open for a few seconds while the vehicle is parked. Opacity is measured based on attenuation of a beam of light that passes through the exhaust gas. The opacity measurement is sensitive to BC from combustion and aerosols from oil and unburned fuel. Thus, the test is good at identifying gross failures of the fuel pump or fuel injectors and at identifying high emitters among older trucks. However, as of the 2007 engine model year for new U.S. trucks, which typically use DPF, BC particle emissions are substantially reduced, and the emitted particles tend to include a large share of small nucleation-mode particles that cannot be detected by an opacimeter. Thus, an opacity test is typically not capable of detecting whether a modern DPF-equipped truck is emitting higher than the low levels consistent with current standards. Replacing opacimeters with laser light-scattering photometers may enable improved detection of particles from modern heavy-duty vehicles. Repairs to HDDVs that reduce opacity or PM emissions could potentially lead to increased NOx emissions. Additional tests could be added for NOx, but many inspection agencies are concerned with increasing the expense or complexity of I/M tests. Remote sensing surveillance programs could also help screen potential high emitters (Denis and Lindner 2005; Houtte and Niemeier 2008; Posada, Yang, and Muncrief 2015).

Program evaluation: real-world versus standards

Based on PEMS measurements of 122 U.S. Tier 1 and Tier 2 LDGVs, real-world CO, HC, and NOx emission rates were typically higher than what was measured on the dynamometer certification test but lower, on average, than the level of U.S. standards (Khan and Frey 2017). This is in part because the measured levels from dynamometer certification tests are typically below the level of the standard by an average of 50% or more. PEMS measurements of heavy-duty vehicles provide further support for the efficacy of emission control systems in achieving real-world emission reductions. For example, in a comparison of PEMS measurements of U.S. diesel refuse trucks built in different model years, the emission rates of CO and HC decreased by a factor of approximately 10 when comparing the newest to oldest truck. NOx emissions decreased by 82% to 90% for the newest trucks with SCR compared to older trucks without SCR. PM emission decreased by approximately 97% for newer trucks with DPF compared to older trucks without DPF (Sandhu et al. 2015). Six U.S. diesel HDV tractors with DOC and DPF that pulled an instrumented flatbed trailer were found to have NOx emissions during highway driving that were lower than the level of the
applicable standard for some engines but not others. Gravimetrically measured PM emissions and THC and CO emissions were well below the levels of the applicable standard (Quiros et al. 2016).

The situation is different, however, for LDVs in Europe and many other parts of the world that use the European regulatory scheme. In-use fuel consumption and CO₂ emissions of 924 passenger cars were found to be higher than the rated values by 11% for gasoline and 16% for diesel (Ntziachristos et al. 2014). Based on measurements of six LDDVs in Europe, NOₓ emissions were consistent with EU standards when tested on the NEDC but were higher than the level of the standard under real-world driving (Kousoulidou et al. 2013). Two Euro 4 diesel buses with SCR measured in China with PEMS were found to have NOₓ emission factors that were more than twice the level of the standard (Fu et al. 2013). Based on 39 Euro 6 vehicles measured with PEMS, real-world NOₓ emission rates averaged 4.5 times higher than the level of the standard (O’Driscol et al. 2016).

Differences in the U.S. and EU regulatory systems help explain differences in concordance of real-world and official values. The EU-type approval process has been based on the NEDC, which is a mild low-power-demand cycle that does not represent real-world driving. EPA has an in-use vehicle surveillance program that requires manufacturers to conduct measurements of vehicles that have been on the road to verify that there has not been substantial deterioration in emissions, whereas the EU does not. In the United States, manufacturers must disclose and obtain certification for any auxiliary devices that would affect emissions, whereas there is not a prior approval process in the EU (Nesbit et al. 2016). The EU approach to adopting RDE regulation is likely to mitigate the lack of conformity between real-world and type approval emission rates.

Measurement methods

Measurement methods are reviewed in more detail in the supplemental materials, including chassis dynamometers, engine dynamometers, tunnel studies, remote sensing, chase vehicles, portable emissions measurement systems, mobile emissions laboratories, automotive sensors, other techniques such as twin site ambient measurements, inverse modeling and methods for evaporative emissions, and low cost sensors (Bishop and Stedman 2008; Brimblecombe et al. 2015; Clark and McKain 2001; Cocker et al. 2004; Faiz, Weaver, and Walsh 1996; Frey et al. 2003, 2003; Harley et al. 2005; Haskew and Liberty 2010; Kelly and Groblicki 1993; Kittelson, Watts, and Johnson 2004; Kotz, Kittelson, and Northrop 2016; Krecl et al. 2017; Nine et al. 1999; Papapostolou et al. 2017; Singer et al. 1999; Stedman 1989; Suleiman, Tight, and Quinn 2016; Wang et al. 2017b). There are excellent critical reviews of vehicle emission measurement methods that provide more detailed treatment of this topic. Ropkins et al. (2009) compare and contrast exhaust measurement methods. Giechaskiel et al. (2014) provide a detailed treatment of particulate matter measurement methods. Franco et al. (2013) review commonly used exhaust emission measurement methods. El-Fadel and Hashisho (2001) review methods for tunnel studies. As part of a meta-analysis of HDDV emission measurements, Yanowitz, McCormick, and Graboski (2000) compare and contrast chassis dynamometer, remote sensing, and tunnel studies. A review of PM emission factors from tunnel, vehicle chasing, roadside, and dynamometer studies is given by Pant and Harrison (2013).

Dynamometer measurements are laboratory based and can be conducted under controlled conditions with high-quality instrumentation, but may not be representative of real-world operations. Remote sensing, tunnel studies, and PEMS-based measurements can be conducted under real-world conditions, which introduces sources of uncontrollable but potentially observable variability, such as road grade, traffic, and weather. Remote sensing and PEMS are more limited in terms of the suite of pollutants that can be measured. Remote sensing and PEMS can provide data for individual vehicles. Tunnel studies are typically based on fleet averages. Remote sensing and tunnel measurements are at a specific site, whereas PEMS measurements are trip based. Chase vehicles can “sniff” plumes of individual vehicles under real-world plume dilution processes. Measurements of near-road air quality can be used to infer vehicle emission rates. The increasing use of sensors to provide data to the vehicle electronic control unit, such as NOₓ sensors needed to control urea injection for SCR systems, opens opportunities to log data from the vehicle. The development of low-cost sensors may enable more ubiquitous measurement of vehicle activity, energy use, and emissions. More details are in the supplemental materials.

Impacts of transportation emissions on health and exposure

The supplemental materials discuss in detail the following topics: (a) evidence for and estimates of the health effects from traffic-related air pollution; (b) empirical evidence regarding traffic-related air pollutant concentrations near
roads, including SOA formation; (c) empirical evidence regarding traffic-related air pollutant concentrations in vehicles; and (d) methods for modeling human exposure to traffic-related air pollution. Criteria pollutants, GHGs, and MSATs all pose threats to public health (EPA 2009a; 2013b; 2010a; 2016b; 2007; Intergovernmental Panel on Climate Change [IPCC] 2014; U.S. Global Change Research Program [GCRP] 2017). Exposure to traffic-related air pollution is estimated to cause approximately 200,000 premature deaths globally each year (Bhalla et al. 2014; Chamberlin et al. 2014). Recent vehicle emission standards in the United States are estimated to avoid approximately 13,000 premature deaths annually (EPA 1999; 2000c; 2014c).

Air pollutant concentrations within 300 m of major roadways have been found to be elevated compared to upwind background concentrations for numerous traffic-related air pollutants (Beckerman et al. 2008; Ginzburg et al. 2015; Hagler, Thoma, and Baldauff 2010; Jeong et al. 2015; Patton et al. 2014; Zhu et al., 2002). SOA formation has also been observed within 300 m of roadways (Stroud et al. 2014). Pollutant concentrations on the roadway are typically higher than near the roadway (e.g., Fruin et al. 2008). The concentration of traffic-related air pollutants to which vehicle occupants are exposed depends primarily on whether windows are open and, if not, on whether the vehicle ventilation system is operating with fresh air intake or recirculated air, as well as the type of pollutant (e.g., gaseous, particle size) (Fruin et al. 2008; Fujita et al. 2016; Gong, Xu, and Zhu 2009; Hudda et al. 2012; Hudda and Fruin 2018; Jiao and Frey 2013; Liu and Frey 2011; Ott, Klepeis, and Switzer 2008; Riediker et al. 2003; Zielinska et al. 2012). Cabin air filters can help reduce in-cabin exposure to particles (Muha et al. 2014; Pui et al., 2008; Yu et al. 2017). For some vehicles and pollutants, there is evidence of self-pollution from crankcase and exhaust emissions that penetrate into the cabin (Harik et al. 2017; Ireson et al. 2011; Zielinska et al. 2008).

Methods for modeling exposure to traffic-related air pollution include stochastic population and microenvironmental models (e.g., Burke, Zufall, and Özkaynak 2001; EPA 2017a; Johnson and Paul 1981; Law et al. 1997), land use regression (e.g., Marshall 2008), street-level models (Zhu et al., 2016; Fu et al. 2017; Shi et al. 2018), and coupled traffic simulation, emissions, and dispersion models (e.g., Hatzopoulou and Miller 2010). Exposure models are useful for identifying high-end exposures, inequality in exposures, and ways to reduce exposures.

**Conclusion**

These conclusions are based on information given in the preceding, as well as in the supplemental materials.

**Trends in vehicle energy consumption**

In the 20th century, the vast majority of the exponential growth in global vehicles and VMT occurred in the United States and Europe. In the last few decades, however, most of the growth in vehicle stock and fuel consumption is shifting to outside of the United States and Europe, largely China and the rest of Asia. Petroleum-based fuels remain the predominant transportation fuels globally. Despite the emergence of a variety of alternative fuel and electric drive vehicles in the last two decades, the share of on-road transportation energy from fuels other than gasoline or diesel is currently trivial. Growth in demand for fossil fuels for transportation has been mitigated to an appreciable extent by improvements in vehicle fuel economy. In the United States, improvements in fuel economy came in two major spurts: (1) a period following two oil embargos in which energy efficiency was a shared national goal, after which fuel economy stagnated for approximately two decades as a result of low fuel prices and consumer interest in larger vehicles; and (2) regulations directed toward addressing climate change. In 2009, EPA found that six GHGs endangered public health and welfare, leading to more stringent fuel economy and new GHG standards effective starting in 2012. In the last decade, fuel economy of the LDV fleet in the United States has improved significantly. Fuel economy standards in many other countries are leading to concurrent improvements. The U.S. adoption of GHG emission standards for heavy-duty diesel vehicles motivates the widespread deployment of many technologies and practices that have been demonstrated at small scale.

**Trends in vehicle emissions**

Recognizing that traffic-related air pollutants, including criteria pollutants and their precursors, and MSATs, are a threat to public health and welfare, increasingly stringent U.S. regulations over the last four decades have led to dramatic reductions in emissions. Because measurements can be expensive and have limitations, many studies are essentially anecdotal, based on samples of the most readily available or easily accessed vehicles or the most accessible tunnel, road network, or a specific location or region. Lacking a more comprehensive surveillance program for real-world emissions, the best we can do at present is to examine multiple chassis dynamometer, remote sensing, tunnel, PEMS, mobile emissions laboratory, chase vehicle, and ambient air quality measurements over time. There, we can see consistent patterns of demonstrated emissions reductions at various locations throughout the United States and elsewhere. A meta-analysis of these multiple lines of evidence could help to draw quantitative
conclusions regarding these trends. Nevertheless, combining these observations with modeling and emissions, it is clear that emission standards have been highly effective in reducing emissions.

**Trends in technology and compliance**

Beginning with the 1970 Clean Air Act, the United States has implemented aggressive standards that forced the deployment of relatively new technologies. These technologies have proven to be effective and durable in real-world operation. The three-way catalyst is one of the most important emission control technologies of the 20th century, and is responsible for a greater than 10-fold decrease in emissions of CO, NO\textsubscript{x}, and VOCs. This success was not possible without an integrated set of successive regulations and technologies over decades. The combination of phasedowns for gasoline lead and sulfur content, requirement for a standard OBD data port, and tiered regulation of vehicle exhaust emissions, coupled with the technology response of adoption of improved computer-controlled fuel injection and feedback control of the air/fuel ratio, was critical to the success of the TWC. The public health benefits of reduced CO, NO\textsubscript{x}, and VOC emissions are indicated by the virtual elimination of CO and NO\textsubscript{x} non-attainment areas in the United States and a general trend of decreasing ambient O\textsubscript{3} ambient concentrations. Ongoing evolution of the three-way catalyst and its applicability to a wide variety of fuels used in stoichiometric engines worldwide promise to accrue ongoing health benefits into the foreseeable future.

The suite of postcombustion controls for diesel engines, including DOC, DPF, and SCR, is likely to be among the most important environmental technologies of the 21st century. The ability of these technologies to reduce real-world emissions has been validated empirically using many measurement methods. Some questions and issues remain regarding the long-term durability of these technologies, emissions from DPF regeneration, SCR effectiveness related to low exhaust temperature, and the potential for SCR ammonia slip. Similar to the story for the TWC, the success of these technologies depends on regulatory push and recognition of the coupling between fuel and vehicle technology. Decades ago, it was impossible to envision a future in which the characteristically high engine-out NO\textsubscript{x} and PM emissions of diesel engines could be reduced by two orders of magnitude, as they have been now. Diesel engines are the backbone of the on-road heavy-duty vehicle fleet, and are used on many types of vehicles, such as transit buses and refuse trucks, that circulate daily through populated communities. The public health benefits of reduced NO\textsubscript{x} and PM\textsubscript{2.5} emissions from diesel vehicles are substantial.

For a new individual vehicle, such as a gasoline-fueled car or a diesel-fueled truck, the emissions per mile of NO\textsubscript{x}, CO, VOC, and PM are typically at least an order of magnitude lower, if not even lower, compared to decades ago. However, GDI engines introduced commercially in the last 10 years clearly have higher exhaust particle emissions than PFI gasoline engines, which may require mitigation with GPF to achieve increasingly stringent particle mass emission standards in the United States and particle number standards in Europe. GPF appears to be a promising and relatively low-cost mitigation option. There is growing recognition that vehicle emissions of IVOCs and SVOCs contribute to the formation of SOA, including some rapid formation near roadways and additional formation at larger spatial scales over a period of days. Although emissions of SOA precursors appear to be decreasing, the amount of SOA formed depends on atmospheric conditions and may be much larger than primary particle emissions. Thus, it is important to continue research on SOA formation and development of targeted prevention options, most likely related to more efficient control of releases of SOA precursors, as well as consideration of control of ambient peroxy radical precursors.

Evaporative emissions from the crankcase and from fuel tank venting are effectively controlled with well-established technologies. Deterioration of components of the evaporative emissions control system, such as fuel filler caps, can be addressed via appropriately designed I/M programs. However, the transition from test-based to OBD check-based I/M programs may be leading to unacceptably high rates of both false positive and false negative outcomes. More work needs to be done to validate that OBD checks are a sufficiently robust indicator of either tampering or failure of emission control systems, and to identify and mitigate potentially innovative forms of tampering related to the OBD system. The institutional and technical approach to I/M for heavy-duty vehicles has generally not kept pace with the changes in HDV emission control technologies in any country, and requires incorporation of measurement methods relevant to the lower emissions of modern diesel vehicles.

**The need for continued improvement**

As older vehicles leave and cleaner, more fuel-efficient new vehicles enter the fleet, the average emission rate (g/mile) will decline substantially for many non-GHG pollutants. However, at some point, without other interventions, growth in vehicle stock, VMT, and fossil
fuel consumption could outpace the reduction in non-GHG emission rates and total emissions could increase.

With regard to GHG emissions, there is no clear panacea in the near future in the absence of strong policies, such as a push for vehicle electrification coupled with a renewable portfolio standard for power generation. Shifts from transportation fossil fuels to EDVs will only lead to substantial GHG emissions reductions if the power generation fuel mix is decarbonized. The attempt at the latter in the United States, the Clean Power Plan, is threatened with rollback under the current political administration. With the forecast continued use of fossil fuels, especially gasoline and diesel, CO₂ emissions will inevitably increase. Shifts from liquid fuels to natural gas may offer some marginal benefit of reduced GHG emissions only if upstream fugitive emissions from natural gas production, transport, and distribution are appropriately managed, and only if emissions of unburned fuel are not substantial. Shifts to hydrogen from steam reforming of methane would offer little marginal benefit. Scenarios in which there is plentiful low carbon energy to extract hydrogen are also ones for which decarbonized electricity is readily available. Thus, given infrastructure requirements for hydrogen fuel, it is difficult to envision such vehicles playing a major role in coming decades. Electrification is likely to be more promising.

A key issue that has emerged in the last decade in Europe and many other countries is a widening disparity regarding real driving emissions compared to laboratory-based measurements of emissions. Although manufacturer cheating could be a contributing factor, the European type approval procedure is clearly a key factor. In Europe, only one low-power-demand driving cycle (NEDC) has been used for type approval, which is not representative of real-world driving, and the replacement cycle (WLTC) is only a small improvement. In the United States, fuel economy rating and emissions certification are based on multiple cycles that represent a much wider range of vehicle operating conditions and therefore are more accurate with regard to real-world fuel economy and emissions. The planned use of PEMS in Europe and China to measure RDE should lead to improved conformity.

I/M programs have likely been very effective at preventing high emissions and remediating some of the vehicles that produce high emissions. However, as the VW scandal illustrates, typical I/M programs based on OBD tests cannot catch defeat devices installed by the OEM. VW was caught because of PEMS measurements made in a research program, not because of I/M programs. The U.S. regulatory framework can impose severe sanctions against those who cheat, which should deter future cheating. However, given that there was a NOₓ emissions scandal in the United States more than a decade prior to the VW scandal, it appears that the threat of penalties is not sufficient to deter cheating and capabilities must be developed to detect cheating. The EU approach to developing RDE regulations means that manufacturers will not know exactly under what conditions their vehicles will be tested. The United States may be an exemplary case study regarding legal aspects of enforcement after a scandal has been detected. The EU may turn out to be an exemplary case study for how to prevent such a scandal in the future.

Despite the limitations of existing assessment methods based on modeling approaches, and the relative sparsity of empirically based measurements of the real-world effectiveness of policies, there appear to be robust findings that land use patterns, demographic factors, economic factors, design and operational factors, and societal values affect demand for transportation, mode choice, energy use, and emissions. Some of these factors, individually, may provide only small benefits, but in combination, coupled with technology-based solutions, could be part of an integrated approach to energy and emissions management. Although technological solutions to reducing emission rates of vehicles per unit of travel (e.g., fuel, time, distance) continue to provide remarkable benefits, the demand for travel is an integral part of trends in vehicle energy use and emissions and a needed focus of research and policy to manage vehicle energy use and emissions.

Non-exhaust emissions are an important contributor to the total burden on the environment from motor vehicle operation. Unfortunately, relatively few data are available for quantifying these emissions. Given their complexity in terms of gaseous and particle species, these emissions need to be better characterized. Furthermore, with trends in vehicle technology, there will be an ongoing need to characterize and manage non-exhaust emissions, as illustrated by venting processes for CNG vehicles, challenges in dealing with venting for HEVs, and the effect of vehicle weight on wear emissions for BEVs.

MOVES does not currently account for some newer technology vehicles, such as HEVs and PHEVs. MOVES also does not explicitly account for some of the trends in conventional LDV technology, such as a transition from PFI to GDI. As emission rates generally decrease, the omission of such sources of variability may lead to biases in emission estimates. Methodologically, more work is needed to better quantify and communicate uncertainties in emission inventories.

As hot stabilized emissions decrease, the relative importance of cold start emissions is likely to increase.
In addition to the well-known cold-start effect for TWC, other catalyst-based systems such as SCR require a minimum exhaust temperature. Thus, technical and policy approaches to address cold-start emissions will be of continuing interest. Wear emissions are likely to persist into the indefinite future as long as vehicles require brakes and tires. Their relative proportion of total vehicle emissions is likely to increase with time as other emissions are reduced or eliminated. More detail is needed on the effect of materials and operational factors on the variability in these emissions.

Comparisons of real-world driving cycles illustrate that some driving patterns can lead to lower emission rates. The concept of eco-driving has been tested mainly with respect to fuel consumption and should be further evaluated with regard to emissions. Lessons learned from comparative analysis of driving cycles can help in designing advice for human drivers and algorithms for automated driving. The large range of inter-vehicle variability often found in measurement studies implies a need for representative sampling and sufficient sample size if the goal is to estimate fleet average emission rates. However, for comparative studies that assess differences in fuels, technologies, traffic controls, and so on, the data requirements may be less stringent, since there are often correlation structures that reduce the variability in differences. The effect of load has not been addressed adequately in the development of vehicle emission factors and requires further characterization. The roles of ambient conditions, and auxiliary loads such as air conditioning, are also not adequately characterized.

Emission factor models such as MOVES are typically based on a relatively small number of driving cycles. However, with the growing availability of data on real-world driving cycles, it is now possible to assess inter-cycle variability as a contributing factor to variability and trends in mean emission rates. Mean emission rates based on sampling of a large number of cycles are likely to be more accurate than emission rates estimated based on a limited number of default cycles.

Currently, it is difficult to predict how full adoption of autonomous vehicle technology will affect energy use, much less whether emissions of GHGs, criteria pollutants, and MSATs will be higher or lower. The net effect depends on a number of factors, including whether such vehicles ultimately will increase or decrease VMT. It is possible that adaptive management could mitigate against large increases in energy use via pricing or cordonning schemes. AVs can be EDVs, but electric drive is not necessary for AV implementation. Given the wide range of potential results, it is critical to promote integrated and iterative consideration of energy, emissions, exposure, and health impacts along the AV development and deployment path.

Improvements in existing methods and emergence of new methods could facilitate more widespread capability to measure vehicle emissions and their impact. Lower cost micro-PEMS, low-cost air quality sensors, and new open path measurement methods may make it easier for governments, communities, businesses, or individuals to monitor local air quality. Deployment of sensors on roving vehicles, such as Google Street View or mass transit vehicles, can enable extensive spatial and temporal data collection that can be used to develop, calibrate, and validate high-resolution exposure models.

**Health effects of vehicle emissions**

Vehicle emissions pose a significant threat to public health. Existing exposure and risk assessment methods are useful in estimating the health risk directly related to vehicles. However, there are key uncertainties and unresolved issues that require ongoing work. Current estimates of vehicle effects are based on models of vehicle emissions and use of damage functions that are derived from epidemiology studies that assess community-wide concentrations from all sources, not only vehicles. More work is needed to characterize spatial and temporal variability in exposures, especially in the on-road and near-road environments, and to determine whether some components of vehicle emissions present higher risk than the community-wide mixtures. Vehicle activity, energy use, emissions, exposure, and health effects are highly heterogeneous, which implies the need to develop effective sampling strategies to account for the myriad of factors that contribute to variability and to have large enough sample sizes to obtain robust findings. This implies a need for adequate commitment of resources for measurements, data collection, data analysis, model development, and data interpretation.

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