# Non-Energy Sector Report

*A Technical Report of the Massachusetts 2050 Decarbonization Roadmap Study*

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# Acknowledgements

This report was prepared by Dr. Jonathan Krones, Visiting Assistant Professor of Earth and Environmental Sciences at Boston College, for the Commonwealth of Massachusetts as part of the Decarbonization Roadmap Study.

**Cover Photo Credit**: Photo of Deer Island Wastewater Treatment Plant courtesy of the Massachusetts Water Resources Authority.

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# List of Abbreviations

| | |
|---|---|
| AC | Air conditioning |
| AD | Anaerobic digestion |
| BOD | Biochemical oxygen demand |
| CARB | California Air Resources Board |
| CFC | Chlorofluorocarbon |
| $CH_4$ | Methane |
| CMR | Code of Massachusetts Regulations |
| $CO_2$ | Carbon dioxide |
| EEA | [Massachusetts Executive Office of] Energy and Environmental Affairs |
| EF | Emissions factor |
| EIA | [U.S.] Energy Information Administration |
| EPA | [U.S.] Environmental Protection Agency |
| F-gas | Fluorinated gas |
| FLIGHT | Facility Level Information on GHGs Tool |
| GHG | Greenhouse gas |
| GHGI | GHG inventory |
| GHGRP | GHG Reporting Program |
| GIS | Gas-insulated [electrical] switchgear |
| GRI | Gas Research Institute |
| GWP | Global warming potential |
| GWSA | Global Warming Solutions Act |
| HCFC | Hydrochlorofluorocarbon |
| HFC | Hydrofluorocarbon |
| HFO | Hydrofluoroolefin |
| IEA | International Energy Agency |
| IPCC | Intergovernmental Panel on Climate Change |
| LDV | Light-duty vehicle |
| LNG | Liquefied natural gas |
| MassDEP | Massachusetts Department of Environmental Protection |
| $MMTCO_2e$ | Million metric tons $CO_2$-equivalent |
| $MTCH_4$ | Metric tons of methane |
| SNAP | Significant New Alternatives Policy |
| MOVES | Motor Vehicle Emissions Simulator |
| MSW | Municipal solid waste |
| MVAC | Motor vehicle air conditioning |
| MWC | Municipal waste combustor |
| $N_2O$ | Nitrous oxide |
| $NF_3$ | Nitrogen trifluoride |
| ODS | Ozone-depleting substance |
| PFC | Perfluorocarbon |
| PHMSA | Pipeline and Hazardous Materials Safety Administration |
| RMP | Refrigerant Management Program |
| $SF_6$ | Sulfur hexafluoride |
| SIT | State (Greenhouse Gas) Inventory Tool |
| SLCP | Short-lived climate pollutants |
| SWMP | Draft 2030 Solid Waste Master Plan |
| tpy | Tons per year |
| UMDI | University of Massachusetts Donahue Institute |
| UNEP | United Nations Environment Programme |
| USCA | U.S. Climate Alliance |
| WMO | World Meteorological Organization |
| WSU | Washington State University |
| WWTP | Wastewater treatment plant |

# 1 Executive Summary

Non-energy-related greenhouse gas (GHG) emissions (or “non-energy emissions”) are a relatively minor but difficult-to-reduce contributor to global climate change. According to the Massachusetts Greenhouse Gas Emissions Inventory maintained by the Massachusetts Department of Environmental Protection (MassDEP), non-energy emissions totaled 5.6 million metric tons of $CO_2$-equivalent ($MMTCO_2e$) in 2017, representing 7.6% of state-wide emissions. Between 2010 and 2017, emissions from non-energy sources in Massachusetts have declined just 1.7%, or roughly 0.2% per year. The pace of reduction is significantly less than in the overall inventory (12.2% reduction over the same period). Even though the current non-energy emissions are relatively small, establishing realistic and cost-effective pathways for emissions reductions in this sector is important to achieving net-zero emissions by mid-century. This Technical Report details efforts to model non-energy emissions, forecast a reference emissions case to 2050, and evaluate possible reduction pathways.

This report is one component of a study led by the Massachusetts Executive Office of Energy and Environmental Affairs (EEA) that explores options for deep reductions in GHG emissions. This non-energy-specific report discusses six subsectors within the non-energy sector, each of which is addressed independently in this report:

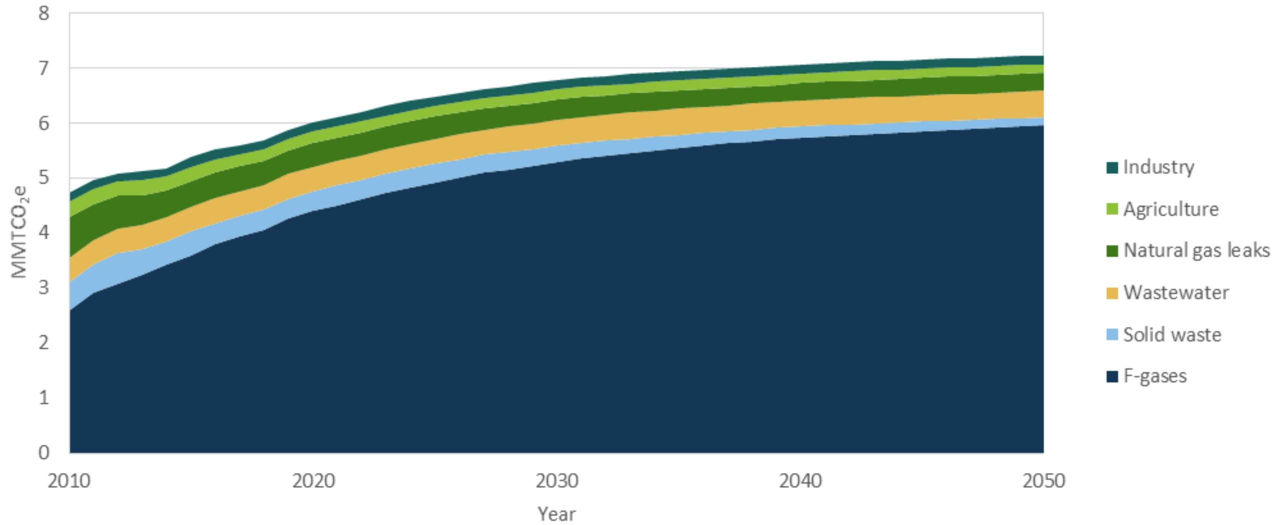
- Fluorinated greenhouse gases (F-gases): associated predominantly with refrigeration, cooling, and electrical switchgear;
- Solid waste management (including landfills): composting and anaerobic digestion, and municipal waste combustion;
- Wastewater treatment (including septic tanks): wastewater treatment plants, and effluent management;
- Natural gas transmission and distribution;
- Agriculture; and
- Industrial processes.

Generally, this report addresses non-$CO_2$ greenhouse gases such as methane ($CH_4$), nitrous oxide ($N_2O$), and F-gases like hydrofluorocarbons (HFCs) and sulfur hexafluoride ($SF_6$). According to MassDEP, the non-energy sector is a major source for all of these GHGs in the Massachusetts GHG Inventory. $CH_4$ is generated from biological processes (e.g., anaerobic decomposition of organic waste in landfills, compost facilities, and wastewater treatment) and directly leaked from natural gas infrastructure. $N_2O$ is produced in agricultural soil management and wastewater treatment. F-gases are emitted from leaks in refrigeration systems, heat pumps, and air conditioners as well as insulation foam, aerosols, semiconductor manufacturing, and electricity transmission equipment.

The modeling approach used for this study built on best practices from the MassDEP GHG emissions inventory, the U.S. Environmental Protection Agency’s (EPA) State GHG Inventory Tool (SIT) and other EPA resources, the California Air Resources Board’s (CARB) modeling and GHG accounting efforts, guidance from the Intergovernmental Panel on Climate Change (IPCC), and academic literature. Each of the six subsector models is intended for use in evaluating the relative importance of the non-energy emissions sources to Massachusetts’ decarbonization goals and the potential effects of different policy and technological interventions on emissions pathways.

The reference case developed for this study forecasts that with no mitigation policies in place, non-energy emissions would reach 7.23 $MMTCO_2e$ by 2050. This is a 53% increase from the modeled 2010 emissions level of 4.73 $MMTCO_2e$.[1] As can be seen in Figure 1, the largest source of non-energy emissions is expected to be F-gases, growing from 2.60 $MMTCO_2e$ in 2010 to 5.96 $MMTCO_2e$ in 2050—an increase of 129%. Emissions from wastewater are also expected to grow moderately over the next three decades, while the other four sources are anticipated to either stay constant or shrink.

*Figure 1. Reference case of total Massachusetts non-energy GHG emissions, 2010-2050 (excluding MWCs)[2]*




A range of technology and policy interventions was evaluated to assess the feasibility and magnitude of possible emissions reductions in each of the six subsectors.

## Fluorinated greenhouse gases

F-gas emissions were modeled using a tool developed by CARB and modified by the research team. The model forecasts annual F-gas emissions from 16 end-use categories, including refrigeration, residential heat pumps, foam blowing agents, and $SF_6$ leakage from gas-insulated switchgear. Forecasted emissions are calculated using emissions intensity factors derived from California's GHG inventory; emissions drivers such as population, number of households, and number of light-duty vehicles; and reduction factors tailored to different policy scenarios.

The reference case for F-gas emissions assumes no F-gas mitigation policies except for those already in place: a reduction in HFC emissions from light-duty vehicle air conditioning systems and a reduction in $SF_6$ emissions from gas-insulated electrical switchgear. The large increase in F-gas emissions forecasted by the model in the

[1] The difference between modeled and official inventory values of the 2010-2017 non-energy emissions is due almost entirely to how F-gases are handled. MassDEP uses national data from the U.S. EPA downscaled by population and this study uses a model developed by CARB for the U.S. Climate Alliance states that is based on data from California.

[2] Although part of the Massachusetts solid waste system, waste incinerators, or municipal waste combustors (MWCs) also generate energy and are inventoried in other sectors. However, emissions from MWCs are also assessed here as they are directly related to waste generation and waste policy.

reference case is driven by a few key sources: rapid growth in commercial refrigeration and air conditioning as well as growth of residential heat pumps and central AC.[3]

The model evaluates a variety of emissions reduction policy scenarios. For example, implementing Significant New Alternatives Policy (SNAP) rules 20 and 21 at the state-level would slow growth in F-gas emissions through 2050, but not lead to any annual reductions. A stringent prohibition of high-global warming potential (GWP) F-gas use in new refrigeration, heat pump, and air conditioning equipment proposed by CARB in the "short-lived climate pollutant" strategy would lead to a reduction in annual emissions so that 2050 emissions would be roughly equivalent to that in 2010. Finally, a scenario that is compliant with the Kigali Amendment to the Montreal Protocol—a phasedown of HFC production and consumption—would lead to the maximum reduction in 2050 annual emissions: 62% below 2010 levels.

This analysis provides a number of lessons for effectively reducing F-gas emissions in the Commonwealth. First, F-gas emissions reduction policies have significant delays, as they depend on the turnover of refrigeration and air conditioning equipment. Therefore, it is important to have low-GWP refrigerants in use as soon as possible, because equipment installed today will be a leakage risk over its entire lifetime. MassDEP's proposed regulation 310 CMR 7.76 is already underway to prohibit the use of certain HFCs and HFC blends in a range of applications in concert with parallel regulatory action in over a dozen other states. The anticipated large deployment of residential and commercial heat pumps puts pressure on the timeline to implement mitigation policies, but further actions, likely at the federal level, are necessary to curb the projected growth of F-gas emissions in the Commonwealth. In addition, F-gas emissions from semiconductor manufacturing should be addressed: they are not subject to any of the standard HFC policies. Finally, although current regulations do limit the leakage of $SF_6$ from electrical switchgear, tightening of these regulations could mitigate emissions further.

### Solid waste

Solid waste system emissions include $CH_4$ and $N_2O$ generation from organic waste decomposition in both landfills and organics recycling facilities like composting and anaerobic digestion (AD). Emissions from the combustion of municipal solid waste for electricity generation are also evaluated in detail but are currently accounted for in MassDEP's energy sector inventory. Modeling emissions from solid waste involves forecasting quantity, composition, and disposal pathway of solid waste in the Commonwealth.

In the reference case, current trends observed in waste generation and disposal infrastructure are assumed to continue through 2050, including an annual waste generation rate of 0.66 tons/capita. All of the municipal solid waste (MSW) landfills in Massachusetts are already slated for closure by the early 2030s, which is responsible for the bulk of the forecasted emissions reductions from solid waste management. These sites will continue to produce $CH_4$ as a result of slowly decaying organic matter, and attention should be paid to the long-term maintenance of landfill gas capture. Compost facilities are also potential sources of $CH_4$ and $N_2O$, depending on how they are managed. There is considerable uncertainty regarding emissions factors from composting, but well-managed composting facilities can control emissions through proper nutrient

[3] Heat pumps, air conditioners, and refrigerators use a volume of refrigerant gas to operate. This gas, almost universally a high-GWP F-gas, can leak out into the atmosphere over the lifetime of the equipment.

management and aeration. Anaerobic digesters, which convert organic waste into $CH_4$ and $CO_2$, are assumed to have negligible GHG emissions.

The Draft 2030 Solid Waste Master Plan (SWMP) lays out an alternative future for the solid waste system in Massachusetts. The major climate-related effect of the Draft 2030 SWMP is to reduce the generation of waste and divert recoverable material from disposal to higher uses. The plan proposes a large-scale increase in organic waste diversion but does not determine if this material is processed in either compost or AD. The GHG emissions impact of organic waste recycling should be evaluated in the context of the potential beneficial uses of the products; compost can be used as a resource in avoiding potential emissions from fertilizer production, while the use of AD can result in avoided emissions by potentially replacing fossil fuel use.

The major source of current waste system emissions is the seven municipal waste combustors (MWCs) operating in the Commonwealth, which burn MSW to reduce the volume of disposed garbage while producing useful heat and electricity. Initial diversion of plastic, paper, and other incinerable materials from the waste stream as called for in the Draft 2030 SWMP is expected to reduce GHG emissions from MWCs but is also expected to affect the economic viability of the facilities. Two long-range trends will affect the volume of disposed-of garbage by MWC facilities operating in Massachusetts. First, the Massachusetts Draft 2030 SWMP articulates a commitment to the longer-term goal of reducing the Commonwealth's solid waste disposal by about 90%, to 570,000 tons by 2050. On such a trajectory, Massachusetts would require less than its current MWC capacity to meet its solid waste management needs. Second — consistent with its obligations to protect the environment and to help the Commonwealth comply with the Global Warming Solutions Act (GWSA), in the event MWCs seek to update or rebuild facilities — MassDEP would require tighter emissions standards and increased efficiency standards based on the latest technology. MassDEP will reassess progress towards this goal in a solid waste program review to be conducted in 2025.

### Wastewater

Emissions from wastewater treatment (0.4-0.5 $MMTCO_2e$ per year) arise from septic systems, wastewater treatment plants (WWTPs), and effluent and sludge management. The reference case forecasts moderate, 9% growth in wastewater emissions from 2010-2050. The largest contributors are $CH_4$ from septic systems and $N_2O$ from sludge management and effluent.

There are no clear pathways for significant and reliable emissions reductions from this subsector. However, model analysis does suggest a few changes to the wastewater system that might lead to modest reductions. First, bringing more people onto sewers and off stand-alone septic systems would likely reduce $CH_4$ emissions from septic tanks, as would encouraging (or requiring) septic system owners to follow best practices. Second, expanding the use of anaerobic digesters at WWTPs would avoid many of the $CH_4$ emissions from WWTPs and have the compound advantage of converting sewage sludge into usable fuel. Finally, reducing per capita protein consumption may reduce $N_2O$ emissions, as would avoiding flushing food waste directly to sewers.

### Natural gas leaks

Leaks from the natural gas transmission and distribution systems in Massachusetts have been a policy priority for years. Compliance with a regulation that requires utilities to replace old and leaky gas pipes has led to the reduction of leaks from the gas distribution system of 40% between 2010-2017, while transmission system leaks have been held roughly constant. The reference case developed for this system assumes gradual growth of the gas distribution network and no change to the transmission system. As a result of anticipated continued replacement of old distribution pipes and service lines with low-leak, plastic pipes through the late 2030s, the model suggests that 2050 emissions in the reference case will be 0.33 $MMTCO_2e$, another 27% below 2017 levels.

Gas meters are a source of leakage that will be difficult to mitigate. It is cost prohibitive if not impossible in practice to create mass-produced consumer meters that do not leak; there is also leakage from the fittings on and around meters. These sources, and the significant uncertainty associated with the leakage rate from hundreds of thousands of different gas meters and fittings deployed across the Commonwealth, may contribute to the disagreement between the model-based assessment of gas leaks and recent empirical studies of $CH_4$ concentrations in the atmosphere in the Greater Boston area. Further work is recommended to rationalize this difference.

The gas transmission system generates gas leaks from compressor stations, liquefied natural gas (LNG) facilities, and transmission itself. In consultation with MassDEP, this study uses an improved methodology over the MassDEP GHG inventory that suggests the transmission system is a relatively minor contributor to overall sector emissions. However, the assumption that emissions will remain constant through 2050 is uncertain.

There is an important dependency between gas leaks and the energy sector. As residential and commercial heating and industrial boilers are electrified, gas meters and associated service lines will likely be taken offline, eliminating their contribution to system emissions. Leaks from distribution pipes will continue, however, as long as the gas system, in whole or in parts, remains pressurized. If electrification were to proceed in a geographic-based strategy, it is conceivable that sections of the gas distribution network could be removed from service, eliminating related sources of leakage altogether. At this point, there is not enough resolution in the gas leaks model to be able to quantify the effect of any particular future electrification strategy. The reference case can, therefore, serve useful as an upper limit for this source of emissions.

### Agriculture

Agricultural emissions arise from three sources in Massachusetts: agricultural soils management, manure management, and enteric fermentation. Agricultural soils generate $N_2O$ as a result of the application of nutrients to the soil, including synthetic fertilizers, sewage sludge, and livestock manure. Manure management refers to the anaerobic digestion of manure on farms. Enteric fermentation is the digestive activity of ruminants, including cows and sheep. Agriculture is a very small sources of emissions in Massachusetts: 0.21 $MMTCO_2e$ in 2017, or .3% of total emissions. Due to the small size of this sector and the inherent link between most agriculture activity and emissions, emissions reduction potential in Massachusetts is highly uncertain and likely linked to the level of agriculture activity taking place in 2050. Improved agricultural practices may make sense to pursue for a variety of reasons in addition to their contribution to emissions reductions.

### Industrial processes

There are four industrial processes that emit GHG in Massachusetts: industrial lime manufacturing and the consumption of limestone, dolomite, and soda ash. All of these processes involve the reduction of a carbonate to an oxide, producing $CO_2$ in the process. No practical alternatives to these processes have been identified for Massachusetts industry, although this may represent the best future source of capturable $CO_2$ for the production of synthetic fuels in the Commonwealth, as discussed in the *Energy Pathways Report*. Emissions from this subsector are very small, less than 0.2 $MMTCO_2e$ annually. The forecast of industrial process emissions is highly uncertain, sensitive to change in individual facility operations and the future of industrial activity in Massachusetts more broadly.

The process-related origins of non-energy emissions generally introduce what are effectively practical limits to decarbonization. Biological sources of $CH_4$ and $N_2O$ (solid waste management, wastewater treatment, and agriculture) can be reduced somewhat but not eliminated. Some industrial processes emit GHGs as part of their fundamental chemistry—again, reductions are possible, but elimination is exceedingly difficult to achieve. As a result, in other sectors, emissions reductions that are challenging but physically possible may be needed to offset the irreducible fraction of non-energy emissions in order to achieve a net-zero goal.

# 2 Overview

Non-energy-related greenhouse gas (GHG) emissions are a relatively minor but persistent contributor to Massachusetts' GHG inventory and global climate change. While energy emissions generally involve fossil fuel combustion, non-energy emissions are generated from a variety of non-fossil fuel combusting anthropogenic activities. Gases emitted from these activities include methane ($CH_4$), nitrous oxide ($N_2O$), and fluorinated gases (F-gases). Although these gases are emitted and accumulate in the atmosphere at lower levels than $CO_2$, they are more potent greenhouse gases than $CO_2$. Consequently, their net impact on influencing global temperature rise is considerable. This report evaluates the generation of these gases and potential mitigation options focusing on the following areas:

- F-gases from refrigeration, air conditioning, industrial processes, and other sources, including some of the most potent GHGs in existence;
- Emissions from solid waste[4] and wastewater management, much of which involves the anaerobic decomposition of organic materials;
- Leaks from natural gas transmission and distribution systems; and
- Process emissions from agriculture and industry.

Non-energy emissions differ from energy emissions in several ways. First, while some emissions in this sector are related to energy systems (F-gas and natural gas leaks), these emissions are not attributable to an incremental consumption of energy. Second, as non-energy emissions are generated by a group of diverse and sometimes complex processes, accurately quantifying these emissions is often more difficult than those from energy generation and use. Finally, while some emissions in this sector can be readily mitigated—such as by using low-GWP refrigerants—other processes remain difficult and prohibitively costly to mitigate—such as the emissions associated with septic tanks and wastewater treatment. These difficult-to-mitigate emissions present a fundamental limit to the extent to which the Commonwealth can decarbonize.

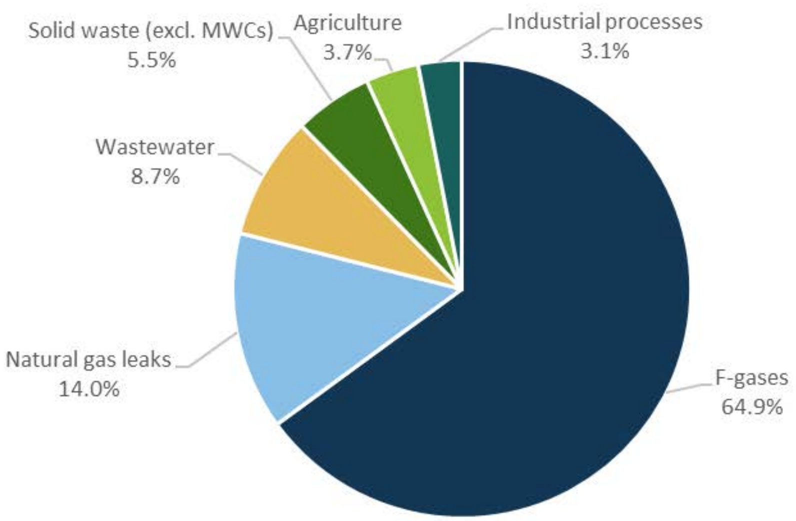
This report is one component of an analysis led by the Executive Office of Energy and Environmental Affairs (EEA) that explores options for reductions in GHG emissions. In companion reports, some of the topics analyzed here are further evaluated and discussed, including the electrification of buildings and a potential decline in the use of natural gas in the Commonwealth. Each of these transitions have implications for certain elements of this sector, specifically the use of refrigerants in building heating systems and fugitive emissions from natural gas systems. While these topics are also discussed in the *Energy System Report* and *Buildings Sector Technical Report,* this *Non-Energy Sector Technical Report* contributes distinct sensitivities associated with these transitions to capture potential non-energy emissions on the high and low realms of possibility. This report thus provides additional non-energy emissions bounds for consideration in these transition areas.

[4] The combustion of municipal solid waste (MSW) for energy recovery generates $CO_2$ emissions from the oxidization of petroleum-derived materials (e.g., plastics) to generate electricity. MassDEP tracks emissions from these facilities in the electricity sector of its state GHG Inventory. These facilities are a minor contributor to that sector but are incorporated in the electricity generation sector model used as the basis for the concurrent *Energy Pathways Report*. In this non-energy report, the dynamics of emissions from MSW energy recovery facilities are explored to compare with other non-energy waste treatment options and evaluate the impacts driven by the Commonwealth's waste management policies.

## 2.1 Massachusetts Non-Energy Emissions

In Massachusetts, non-energy sources emitted 5.6 million metric tons $CO_2$-equivalent ($MMTCO_2e$) of GHGs in 2017,[5] 7.6% of the total 73.4 $MMTCO_2e$ inventory in that year. Figure 2 illustrates the relative contributions of each non-energy sector activity. Within non-energy emissions, F-gas emissions (3.6 $MMTCO_2e$) make up more than three-fifths of the non-energy total, followed by 14% from natural gas leaks (0.78 $MMTCO_2e$), 8.7% from wastewater management, 5.5% from solid waste management (0.31 $MMTCO_2e$, excluding emissions from municipal waste combustors, which are accounted for in the energy sector), 3.7% from agriculture (0.21 $MMTCO_2e$), and 3.1% from industrial processes (0.17 $MMTCO_2e$).

*Figure 2. Massachusetts inventory of non-energy GHG emissions from MassDEP, 2017, in $MMTCO_2e$[6]*




Emissions from non-energy sources have been relatively stable in Massachusetts. From 1990 to 2017, non-energy emissions have declined 10.5%, or roughly 0.4% per year (Figure 3). The pace of reduction is less than half of that in the overall inventory (22.3% reduction from 1990-2017). Within the non-energy sector, however, there have been significant changes. Emissions from natural gas leaks and solid waste management have both declined substantially since 1990, but nearly all of the reductions have been offset by rapid growth in F-gas emissions.[7] The other three subsectors—wastewater, agriculture, and industrial processes—have remained largely stable.

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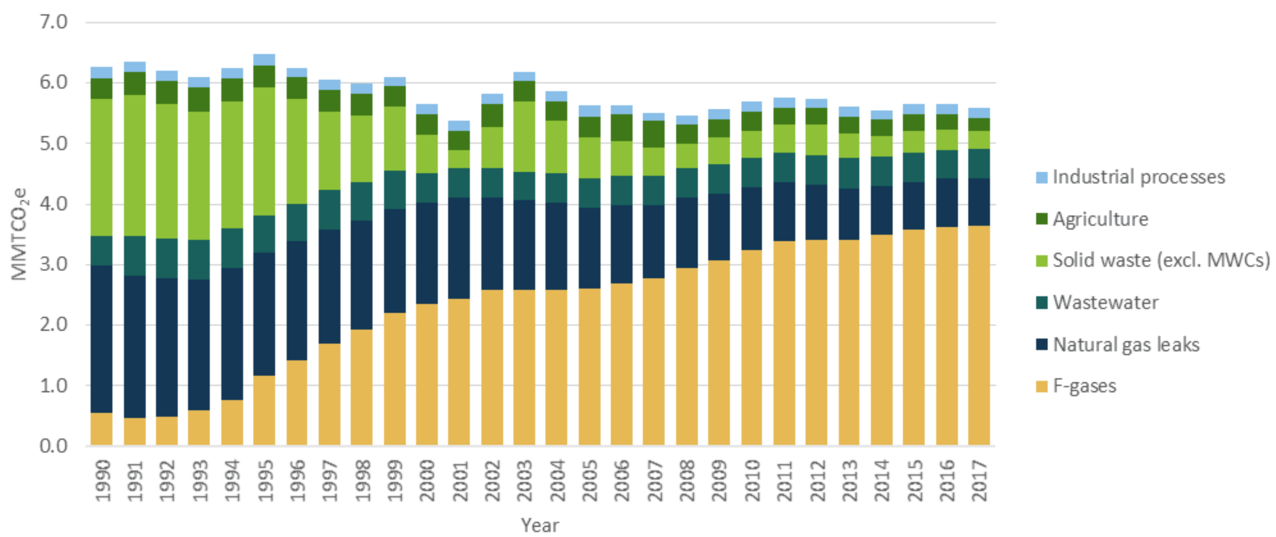
[5] These values exclude emissions from municipal waste combustors (MWCs), which are both solid waste management facilities and power plants. Emissions from these facilities are accounted for in energy sectors, but are also addressed in this report because of their relationship with the waste system. MWC emissions add an additional 1.2 $MMTCO_2e$ GHG emissions in 2017.

[6] [MassDEP, "MA GHG Inventory," 2020]. Massachusetts Department of Environmental Protection, "Massachusetts Annual Greenhouse Gas Inventory: 1990-2017," 2020. Available at https://www.mass.gov/lists/massdep-emissions-inventories (Retrieved 19 August 2020).

[7] The F-gas emissions subsector excludes ozone-depleting substances like CFCs and HCFCs. This internationally agreed upon data convention means that F-gas GHG emissions inventories are artificially low through the 1990s during the CFC phase-out, since those gases were also GHGs.

*Figure 3. Massachusetts inventory of non-energy GHG emissions from MassDEP, 1990-2017, in MMTCO$_2$e*[8]




## 2.2 Approach

This technical report details an effort to forecast GHG emissions from 2020-2050 and evaluate the potential for deep decarbonization in each non-energy sector activity. For each of the six subsectors, a reference case is defined based largely on current emission inventories, current trends, and current policies. Each subsector is modeled independently following methodological guidance from the Massachusetts GHG Inventory methodology, the State GHG Inventory Tool (SIT) produced by the U.S. Environmental Protection Agency (EPA), the Intergovernmental Panel on Climate Change (IPCC), and the California Air Resources Board (CARB). Potential reductions from each subsector are calculated by representing various technology and policy interventions in the models and comparing decarbonization results to the reference case. In addition, a number of variations are explored to test uncertainty in the models.

## 2.3 Overview of Mitigation Strategies

Even though the current emissions levels for these non-energy activities are relatively small, identifying and implementing strategies to stabilize and reduce them is important to achieving the Commonwealth's GHG emission reduction goals by 2050. As energy decarbonization strategies (e.g., electrification, renewables) continue to drive down emissions from electricity, buildings, and transportation, non-energy emissions will come represent a larger and larger fraction of the remaining total inventory. To achieve maximum decarbonization by 2050, this persistent fraction must also be reduced. Although many of the non-energy emission sources are challenging or even practically impossible to reduce with existing technologies, some approaches for mitigating non-energy emissions include:

- Phasing out high-GWP F-gases and replacing with low-GWP alternatives like hydrofluoroolefins (HFOs) and propane;

[8] MassDEP, "MA GHG Inventory," 2020.

- Comprehensive waste reduction policies that would enable landfills, key sources of methane leakage, to be closed and minimize the quantity of fossil-derived plastic that gets sent to incinerators; and
- Leak-prone natural gas distribution pipes can be replaced or retired from service.

## 2.4 Synthesis

While this report is part of a suite of technical reports explaining analysis conducted for the Commonwealth as part of a broader decarbonization study, it is intended to stand alone. This means that the important interdependencies between non-energy sector activities and other key sectors are unable to be fully characterized. Some of these interdependencies include:

- Tradeoffs between F-gas emissions and building energy efficiency, in which high-efficiency air conditioners and electric heat pumps, as well as high-R insulation all depend on increasing amounts of refrigerant that can leak.
- Potential synergies between solid waste and wastewater and biogas. Organic waste and wastewater sludge could be disposed in anaerobic digesters to create biogas for use in heating and electricity production.
- Potential redundancies between natural gas leak repairs and gas system retreat. Mitigating gas leaks involves costly, ongoing repairs to the gas distribution system which may be an unnecessary expense as electrification becomes the dominant heating technology.

There are a number of uncertainties and limitations associated with the non-energy sector and the modeling and analysis performed here. Some of these uncertainties stem from the relatively small contribution of non-energy emissions to the overall state-wide inventory. This means that data and other informational resources specific to Massachusetts are limited and analysis must rely on more general models that might miss details unique to the Commonwealth. Also, in some cases, what would be considered a minor change (e.g., the opening or closure of a single facility) can have significant consequences on non-energy emissions.

Beyond the modeling uncertainty, the process-related origins of non-energy emissions introduce what are effectively practical limits to decarbonization in this sector; in other sectors, emissions reduction that are challenging but physically possible may be needed to balance the non-energy emissions that are not possible to abate in order to achieve a net-zero goal. Biological sources of $CH_4$ and $N_2O$ (solid waste management, wastewater treatment, and agriculture) can be reduced somewhat but not eliminated. Some industrial processes emit GHGs as part of their fundamental chemistry—again, reductions are possible but elimination is exceedingly difficult to achieve.

# 3 Fluorinated Greenhouse Gases

## 3.1 Introduction

Fluorinated gases (F-gases) are a class of synthetic greenhouse gases (GHGs) that includes hydrofluorocarbons (HFCs), perfluorocarbons (PFCs), nitrogen trifluoride ($NF_3$), and sulfur hexafluoride ($SF_6$).[9] F-gases are widely used throughout the economy (Table 1). They are refrigerants in chillers, air conditioners, and heat pumps; blowing agents for foam insulation; propellants for aerosols and fire suppression systems; industrial solvents used in metals and electronics manufacturing; and dielectric substances used in electrical switchgear. Emissions from these sources, as well as other types of manufacturing that emit F-gases as byproducts, are potent contributors to greenhouse gas emissions and increasingly problematic to the global climate.

*Table 1. Types of F-gases and characteristic uses/applications.*

| Chemical type | Chemical formula/ abbreviation | Characteristic application(s) |
|---|---|---|
| **Hydrofluorocarbon** | HFC | Refrigeration, air conditioning, blowing agents, aerosols |
| **Perfluorocarbon** | PFC | Industrial solvents |
| **Nitrogen trifluoride** | $NF_3$ | Industrial etchants |
| **Sulfur hexafluoride** | $SF_6$ | Dielectric medium in gas-insulated electric switchgear |

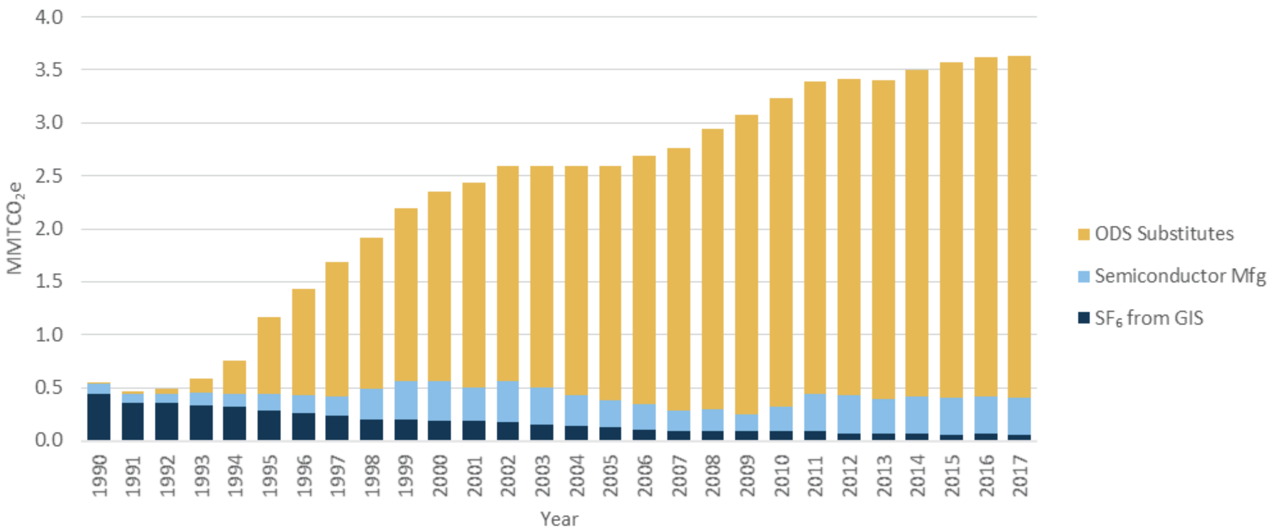
As a category, F-gases are extremely powerful GHGs. Even small quantities of F-gases can have major effects on the climate, as commonly-used F-gases have global warming potentials (GWP) thousands of times that of $CO_2$.[10]

The Massachusetts greenhouse gas inventory accounts for 3.6 million metric tons $CO_2$-equivalent (MMT$CO_2$e) of F-gases in 2017, an increase of 12% from 2010 (see Figure 4).[11] The inventory includes three sources of F-gas emissions:

- ozone-depleting substances (ODS) substitutes, which refers to the mainly HFCs that replaced chlorofluorocarbon (CFCs) and hydrochlorofluorocarbons (HCFCs) after those chemicals were banned in the Montreal Protocol;
- semiconductor manufacturing, which includes solvents and other process gases; and
- electric power transmission and distribution systems, which refers to $SF_6$ from gas-insulated switchgear (GIS) and other electrical equipment.

[9] There are other fluorinated compounds in use, including chlorofluorocarbons (CFCs) and hydrochlorofluorocarbons (HCFCs), manufacture of which have been banned by the Montreal Protocol.
[10] [WMO, 2018]. World Meteorological Organization, "Scientific Assessment of Ozone Depletion: 2018. Global Ozone Research and Monitoring Project—Report No. 58," Geneva, Switzerland: WMO, 2018. Available at https://csl.noaa.gov/assessments/ozone/2018/ (Retrieved 10 December 2020).
[11] MassDEP, "MA GHG Inventory," 2020.

*Figure 4. Massachusetts inventory of F-gas emissions from MassDEP, 1990-2017.*[12,13]




The inventory confirms that ODS substitutes are by far the largest contributor to GHG emissions from this subsector, comprising 88.7% of the subsector total in 2017. Emissions of ODS substitutes have grown at a pace of 1-3% per year since 2010, while emissions from semiconductor manufacturing (9.6% of total in 2017) are relatively stable and $SF_6$ emissions (1.7% of total in 2017) are declining gradually over time.

This chapter presents forecasts of F-gas emissions in Massachusetts through the year 2050, with the aim of identifying pathways for deep reductions in the emissions of these powerful GHGs. The chapter details the methodology used to develop a reference case as well as to evaluate various policy and technology options for reducing emissions. It closes with a discussion of uncertainty, model sensitivity, and policy implications of the model results.

For context, F-gases are a major climate concern worldwide, where rapid growth in F-gas emissions is expected to continue as availability of and demand for heat pumps, air conditioners, and other cooling appliances is increasing rapidly. According to a recent study from the UN Environment Programme (UNEP) and the International Energy Agency (IEA), without policy interventions, "HFC emissions are projected to raise global temperatures by 0.3-0.5°C by 2100,"[14] making action on F-gas emissions an urgent priority for climate mitigation.[15]

[12] By international convention, this inventory excludes the contributions of CFC and HCFC emissions in the 1990s and early 2000s, because a plan to reduce those emissions has been in place since the late 1980s.
[13] MassDEP, "MA GHG Inventory," 2020.
[14] HFCs were 88% of total global F-gas emissions in 2018, by $CO_2$-eq.
[15] [UNEP & IEA, 2020, p. 20]. United Nations Environment Programme & International Energy Agency, "Cooling Emissions and Policy Synthesis Report," Nairobi & Paris: UNEP & IEA, 2020. Available at https://www.iea.org/reports/cooling-emissions-and-policy-synthesis-report (Retrieved 19 August 2020).

### 3.1.1 Background: Types, Sources, and Characteristics of F-gases

F-gases considered in this model include four categories of synthetic chemicals: HFCs, PFCs, $NF_3$, and $SF_6$. The most important of these vis-à-vis climate change for both Massachusetts and the entire U.S. are HFCs, comprising 94% and 89% of total F-gas emissions (weighted by $CO_2$-eq.), respectively.[16,17]

#### 3.1.1.1 HFCs

HFCs are organic compounds made of chains of carbon atoms surrounded by hydrogen and fluorine atoms. There are numerous HFCs on the market, differing by length of the carbon chain and the placement, orientation, and ratio of hydrogen and fluorine atoms. The chemical structure of a given HFC compound determines its thermodynamic properties and functionality as well as its effect on the climate.

HFCs are classified as short-lived GHGs, with atmospheric lifetimes ranging from 27 days to 228 years.[18] The average atmospheric lifetime of HFCs currently in use, weighted by tonnage emitted, is 15 years.[19] Despite their relatively short lifetimes, HFCs are strong GHGs, although the ability of any given HFC compound to affect the climate depends in large part on its atmospheric lifetime: the longer it remains in the atmosphere, the larger the effect. The 100-year GWP values for HFCs range from 6 to 12,690, with a usage-weighted average value of 1,700.[20,21] Characteristics of major HFCs and other F-gases are presented in Table 2.

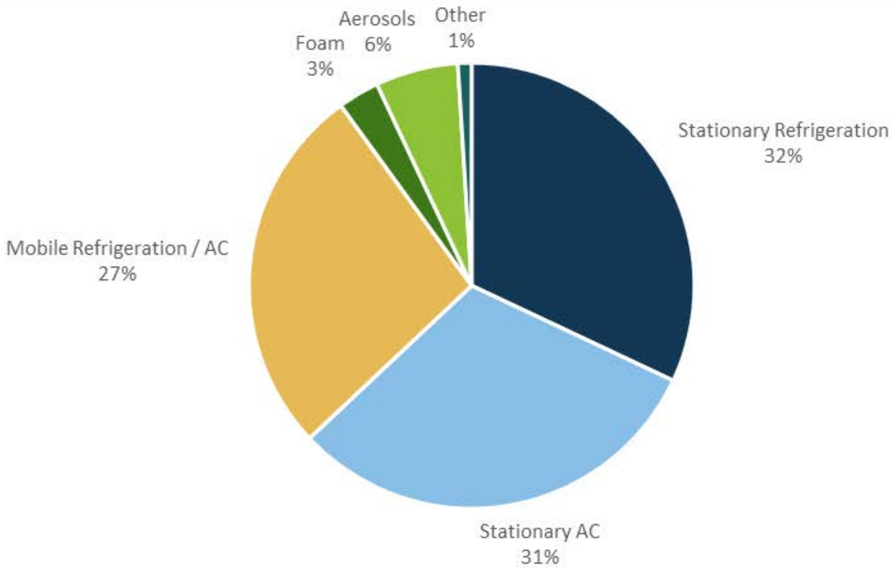
*Table 2. Atmospheric lifetimes and GWP values for a representative catalog of F-gases.*[22]

| Chemical name | Chemical formula | Atmospheric lifetime (years) | GWP-20 | GWP-100 |
|---|---|---|---|---|
| **HFC-32** | $CH_2F_2$ | 5.4 | 2,530 | 705 |
| **HFC-125** | $CHF_2CF_3$ | 30 | 6,280 | 3,450 |
| **HFC-134a** | $CH_2FCF_3$ | 14 | 3,810 | 1,360 |
| **HFC-143a** | $CH_3CF_3$ | 51 | 7.050 | 5.080 |
| **PFC-14** | $CF_4$ | 50,000 | 4,880 | 6,630 |
| **PFC-116** | $C_2F_6$ | 10,000 | 8,210 | 11,100 |
| **Nitrogen trifluoride** | $NF_3$ | 569 | 12,460 | 15,750 |
| **Sulfur hexafluoride** | $SF_6$ | 3,200 | 17,500 | 23,500 |

[16] [U.S. EPA, 2020]. U.S. Environmental Protection Agency, "Inventory of U.S. Greenhouse Gas Emissions and Sinks: 1990-2018," EPA 430-R-20-002, 2020. Available at https://www.epa.gov/ghgemissions/inventory-us-greenhouse-gas-emissions-and-sinks-1990-2018 (Retrieved 19 August 2020).
[17] MassDEP, "MA GHG Inventory," 2020.
[18] WMO, 2018.
[19] [CARB, 2018]. California Air Resources Board, "Current and Projected Emission Inventory and Methodology of HFC-Gases, Black Carbon, and Methane in California and Other U.S. Climate Alliance States," Draft Final, 2018.
[20] The short atmospheric lifetimes of these compounds make the choice of time horizon in GWP an important one. The HFC usage-weighted average GWP-20 is 3,800, 224% larger than the GWP-100 value. CARB calculated the average GWP and atmospheric lifetime of the mix of HFC emissions in use by taking an average weighted by the mass of each HFC emitted in California's 2018 GHG Inventory Report.
[21] There is a class of unsaturated HFCs known as hydrofluoroolefins (HFOs) with extremely short atmospheric lifetimes and negligible GWP (WMO, 2018). HFOs are promising, low-GWP substitutes for HFCs in many applications, as is discussed later in the report.
[22] WMO, 2018.

The main source of HFC emissions today is their use as refrigerants in chillers and air conditioners. These sources were responsible for 90% of the total U.S. HFC emissions inventory in 2018, as illustrated in Figure 5.[23] The remaining 10% of HFC emissions arises from aerosols, foams, solvents, and fire suppression systems. By 2018, these emissions were mainly composed of four compounds: HFC-32, HFC-125, HFC-134a, and HFC-143a.[24] Minor compounds include HFC-23[25] and HFC-236fa, among others. Most HFC emissions are the result of system leakage. Chillers, air conditioning units, and heat pumps can lose some refrigerant when they are first manufactured and charged. They can leak gradually over their lifetime through compressors and other leaky components as well as during servicing events or accidents. Finally, at end-of-life, this equipment can vent its remaining refrigerant charge to the atmosphere if not handled responsibly. Leakage rates from this equipment varies by model, installation and maintenance practices, and age, with some equipment not leaking at all and others much more prone to leaks and fugitive emissions. Closed-cell insulating foam can also leak HFC during manufacturing, over the foam's lifetime, and at disposal. Aerosols that utilize HFC propellants include medical inhalers and spray cans for both personal care and technical products. HFCs from these products are intentionally released in the course of the products' use. The same occurs with fire suppression systems, including both fire extinguishers and flooding systems. HFCs used as industrial solvents in semiconductor manufacturing are partially destroyed in their use and in industrial pollution control systems. Any HFC solvents that remain are emitted into the atmosphere.[26]

*Figure 5. U.S. HFC Emissions 2018: 191 MMTCO$_2$e[27]*




[23] CARB, 2018.
[24] U.S. EPA, 2020.
[25] HFC-23 is generated as a byproduct of HCFC-22 manufacturing, itself an ODS that should be phased out of production by 2020.
[26] More detailed descriptions of the sources of HFC emissions can be found in U.S. EPA (2020) and CARB (2016). [CARB, 2016]. CARB, "California's High Global Warming Potential Gases Emissions Inventory: Emissions Inventory Methodology Technical Support Document. 2015 Edition," 2016. Available at https://ww3.arb.ca.gov/cc/inventory/slcp/doc/hfc_inventory_tsd_20160411.pdf (Retrieved 18 August 2020)
[27] CARB, 2018.

#### 3.1.1.2 PFCs, $NF_3$, and $SF_6$

PFCs, $NF_3$, and $SF_6$ are all fully fluorinated compounds, which means they include no hydrogen in their chemical structure. This chemical property causes the compounds to be very stable, which results in extremely long life and large climate impact (see Table 2). These compounds are emitted mainly from industrial sources. PFC emissions result from aluminum and semiconductor manufacturing. In aluminum production, $CF_4$ and $C_2F_6$ are produced when the Hall-Héroult cell enters a state called the "anode effect." In semiconductor manufacturing, PFCs, along with HFCs, $NF_3$, and $SF_6$, are intentionally used as solvents, etchants, and cleaning gases. The portion of these gases that is neither consumed during the cleaning or etching process nor destroyed by pollution control systems is released to the environment. Semiconductor manufacturing is also the main source of $NF_3$ emissions.

In addition to semiconductor manufacturing applications, $SF_6$ is used in magnesium production and, most notably, as a dielectric and insulator in electric power distribution and transmission. In electricity sector applications, $SF_6$ emissions are the result of leaks from GIS and circuit breakers.

There is no aluminum or magnesium manufacturing in Massachusetts, limiting the potential uses—and emissions sources—of these fully fluorinated gases.

### 3.1.2 Strategies for reducing F-gas emissions

Strategies for reducing F-gas emissions include abatement, technology improvement, and chemical bans. Abatement refers to the mandatory recycling or destruction of the F-gas before it can be released to the environment. This approach is only applicable in certain contexts, namely, those for which it is possible to maintain a minimum degree of control over the emission, such as industrial facilities and refrigerant recovery and management programs. Technology improvement acknowledges that there are alternatives to nearly every technology that is a source of F-gas emissions. Emissions performance standards—which can tighten over time—and associated monitoring can help reduce F-gas usage and emissions. Finally, certain high-GWP gases can be banned, either with a phasedown of manufacturing or with an outright prohibition on use. The fact that most F-gas emissions occur as leaks over the lifetimes of various technologies and appliances means that there are inherent time delays associated with any of these approaches and inertia from existing inventory of technology. For example, a ban on the production and sale of a high-GWP refrigerant like HFC-227ea (GWP-100: 3,140) would not affect the gradual leakage of that gas from air conditioners, chillers, aerosols, etc. that had already been manufactured and sold. Even though the ban is in place, future emissions of that gas will occur until the time when that equipment is retired, which, in the case of some commercial refrigeration or residential air conditioning systems, could be 20 years or more. This time delay, combined with the short lifetime of many HFCs that concentrates climate forcing impacts in the next few decades, makes it especially urgent to address F-gas emissions in the face of the expected growth in refrigeration worldwide.

The effectiveness of F-gas emissions mitigation policies hinge in part on the availability of low-GWP alternatives. A number of alternatives to HFCs have been developed and are in various stages of implementation around the world. In general, there are four options for replacing conventional HFCs:[28]

- Low-GWP HFCs, like R-32 ($CH_2F_2$, GWP = 705);
- Hydrofluoroolefins (HFOs), which have GWP $\approx$ 1;
- HFC/HFO blends; and
- Natural refrigerants like propane, ammonia, and $CO_2$.

Each of these options has different thermodynamic and safety properties, which means that there is no single best option for all applications. The availability of these HFC alternatives also varies depending on geography, as does the availability of equipment designed to operate using low-GWP refrigerants.

F-gas emissions have been studied for several years and a number of policies are currently either in force or are being further explored at the state, federal, and international level. This section describes existing policies that could be applied in Massachusetts to reduce F-gas emissions by 2050.

THE IMPORTANCE OF REDUCING F-GAS USE TODAY:

Even if an HFC production and sale ban were place, future emissions will occur until the time when the HFC-using equipment is retired, which, in the case of some commercial refrigeration or residential air conditioning systems, could be 20 years or more. This time delay, combined with the short lifetime of many HFCs that concentrates climate forcing impacts in the next few decades, makes it especially urgent to address F-gas emissions in the face of the expected growth refrigerant use in heat pumps worldwide.

#### 3.1.2.1 Significant New Alternatives Policy (SNAP)

The U.S. Environmental Protection Agency's (EPA's) Significant New Alternatives Policy (SNAP) Program was established under Section 612 of the Clean Air Act to continually evaluate ODS substitutes in a number of end-use applications. As new alternatives became available for a given application, older, more environmentally impactful gases would be prohibited. In SNAP rules 20 (2015) and 21 (2016), the EPA began prohibiting the use of certain high-GWP HFCs in refrigeration and air conditioning, aerosols, foam blowing, and fire suppression and explosion protection. These rules were challenged in court and partially vacated in 2018. In 2020,

[28] [EC, 2020]. European Commission, "Climate-friendly alternatives to HFCs," EC, 2020. https://ec.europa.eu/clima/policies/f-gas/alternatives_en (Retrieved 27 September 2020).

MassDEP released a proposed regulation (310 CMR 7.76) that would adopt portions of these partially-vacated SNAP rules.[29]

### 3.1.2.2 Refrigerant Management Program (RMP)

Section 608 of the Clean Air Act sets rules for refrigerant management at the federal level to prohibit the intentional venting of refrigerant to the atmosphere. Section 608 regulations include licensing for technicians, refrigerant control during equipment servicing, restriction on sales of refrigerants, a leak repair requirement, and refrigerant recovery before equipment disposal. This suite of requirements was originally intended to mitigate the effects of refrigerants on the ozone layer, but in 2016 the U.S. EPA finalized a rule expanding its applicability to ODS substitutes as well, including HFCs. In February 2020, the EPA rescinded the extension of the leak repair requirements to ODS substitutes, meaning that stationary equipment using HFCs were no longer subject to that requirement. The rest of the 2016 rule remained in force, including the end-of-life recovery and refrigerant recycling rules.

In 2009, California implemented its own version of the RMP that includes "registration; refrigerant leak detection and monitoring; leak repair; reporting and recordkeeping; system retrofit or retirement planning; required service practices; and refrigerant distributor, wholesaler, and reclaimer prohibitions, recordkeeping, and reporting."[30] It goes beyond what the federal RMP would have done and applies to high-GWP refrigerants used in stationary, non-residential refrigeration systems.

### 3.1.2.3 Short-Lived Climate Pollutants Policies (SLCP)

The California Air Resources Board (CARB) has recommended a series of measures to reduce HFC emissions as part of its Short-Lived Climate Pollutants (SLCP) program.[31] These measures include prohibitions on high-GWP HFCs in stationary refrigeration and air conditioners that are more aggressive than those in the SNAP rules. Specifically, CARB proposed a rule in July 2020 to add GWP limits on new stationary refrigeration and air conditioning systems.[32] The draft rule would require new stationary refrigeration systems installed after January 1, 2022 to use a refrigerant with a GWP < 150 and new stationary air conditioning systems installed after January 1, 2023 to use a refrigerant with a GWP < 750.

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[29] [MassDEP, "310 CMR 7.76," 2020]. MassDEP, "310 CMR 7.76: Proposed Prohibitions on Use of Certain Hydrofluorocarbons," October 2020. Available at https://www.mass.gov/doc/310-cmr-776-proposed-prohibitions-on-certain-hydrofluorocarbons/download (Retrieved 10 December 2020).

[30] [CARB, "RMP," 2020]. CARB, "Refrigerant Management Program: About," 2020. Available at https://ww2.arb.ca.gov/our-work/programs/refrigerant-management-program/about (Retrieved 18 August 2020).

[31] [CARB, 2017]. CARB, "Short-Lived Climate Pollutant Reduction Strategy," 2017. Available at https://ww2.arb.ca.gov/sites/default/files/2018-12/final_slcp_reduction_strategy_w_appx_march2017%20Final%202017.pdf (Retrieved 18 August 2020).

[32] [CARB, "HFC Prohibitions," 2020]. CARB, "Proposed Regulation Order: Prohibitions on Use of Certain Hydrofluorocarbons in Stationary Refrigeration, Stationary Air-conditioning, and Other End-Uses," 2020. Available at https://ww2.arb.ca.gov/sites/default/files/2020-07/DRAFT%20CA%20SNAP%20Amendments-Reg%20Text.pdf (Retrieved 18 August 2020).

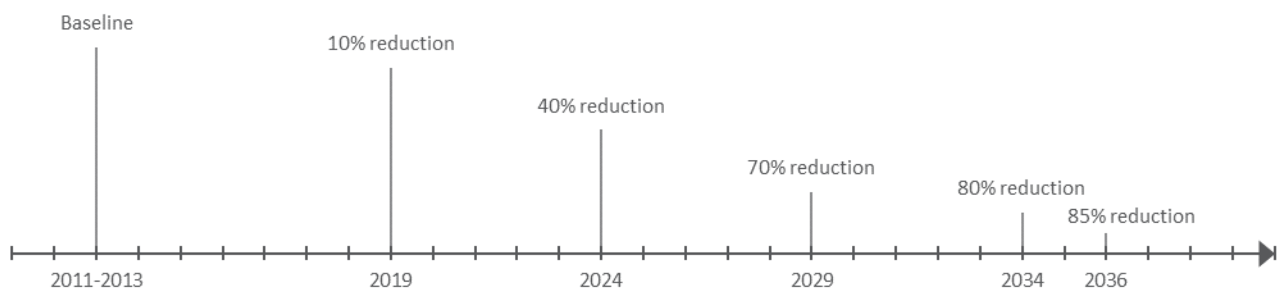
#### 3.1.2.4 Kigali Amendment

In 2016, the Parties to the Montreal Protocol adopted an amendment at the meeting in Kigali, Rwanda that established a schedule for the phasedown of the production of HFCs. This Kigali Amendment requires industrialized countries to reduce HFC production and consumption 85% below a baseline by 2036, with rapidly industrializing and developing countries following in the subsequent decade. As of January 1, 2019, the Kigali Amendment has come into force globally, but the U.S. has not ratified and has taken no actions at the Federal level to comply with the HFC phasedown schedule, presented in Figure 6.

#### 3.1.2.5 Reducing $SF_6$ Emissions from Gas-Insulated Switchgear

Massachusetts has a state-level rule focused on reducing $SF_6$ emissions from gas-insulated switchgear (310 CMR 7.72). This rule, finalized in 2014 and revised in 2017, required large utilities to ensure a maximum annual $SF_6$ leak rate of 1% from their GIS by 2020.

*Figure 6. Schedule of HFC phasedown in the Kigali Amendment for Non-Article 5 (main group) countries, including the United States*[33]




### 3.1.3 F-gas Emissions Modeling

Estimating and forecasting F-gas emissions and quantifying the potential effects of different mitigation policies requires a modeling approach that can accommodate the time delays and inertia described above.[34] The U.S. EPA maintains an inventory of national F-gas emissions using their Vintaging Model.[35] This model was originally developed to track the emission of ODS under the terms of the Montreal Protocol and the U.S. Clean Air Act. As ODSs are phased out, the model continues to track use and emission of ODS substitutes, including HFCs and other F-gases. The Vintaging Model is so named because it tracks the emissions performance and lifetime of distinct annual "vintages" of equipment produced and deployed each year. Each modeled piece of equipment of every vintage is assigned multiple emissions factors: leakage from manufacturing and installation is observed during its first year; leakage during operation is observed every year of its life; and leakage from servicing and retirement are observed in the estimated years that those activities occur. With 69 distinct end-

[33] [Kigali Amendment, 2019]. "Kigali Amendment to the Montreal Protocol on Substances that Deplete the Ozone Layer." Full text in English. Available at https://ozone.unep.org/sites/default/files/2019-04/Original_depositary_notification_english_version_with_corrections.pdf (Retrieved 16 November 2020).

[34] The IPCC publishes detailed guidance for constructing national F-gas emissions inventories (*Guidelines for National Greenhouse Gas Inventories*), but this does not necessarily translate into prospective models.

[35] U.S. EPA, 2020, Section 3.9.

use categories and decades of overlapping vintages, this model aggregates an enormous amount of data into total estimates of F-gas emission.[36]

The Vintaging Model has been used to forecast national F-gas emissions to 2050 by combining equipment lifetime estimates with forecasts of future equipment demand and performance.[37] It is limited, however, in three key areas:

- First, it is strictly a national model and does not break out its data or results at the state level. For the State Greenhouse Gas Inventory Tool (SIT) provided by the U.S. EPA to the states to assist them in managing their own GHG inventories, the default approach to estimating ODS substitutes (and semiconductor emissions) is to downscale national estimates by population.[38] This approach requires an assumption that per capita use of ODS substitutes is equal across the country. Significant regional differences in air conditioning and industrial activity demonstrate that this assumption is unideal.
- Second, the model is limited in its ability to test future emissions mitigation policies. The EPA notes that "the simulation is considered to be a 'business-as-usual' baseline case and does not incorporate measures to reduce or eliminate the emissions of these gases other than those regulated by U.S. law or otherwise common in the industry."[39]
- Finally, and most crucially for limiting the ability of a state to use the Vintaging Model to develop deep decarbonization plans, it is not an open model. This is due likely in part to the confidential business data that the EPA uses to refine its emissions factors and lifetime information.

As a result of these limitations of the national model, California created its own version: the CARB F-Gas Inventory Model.[40,41] CARB summarizes their model and methodology as follows:[42]

> Annual emissions of 76 F-gases, F-gas blends, and low-GWP substitutes are estimated from 33 end-use sectors (types of equipment and materials that use F-gases). The basis of CARB's Vintaging-type model is that emissions from a given production-year of equipment (fleet, or vintage) during its lifetime will overlap with the emissions from equipment produced in subsequent production-years. Emissions are aggregated for all equipment and material types that are in use longer than one year. F-gases in the following end-use sectors are assumed to be fully emitted the same year of production: aerosol propellants, fire suppressants, and solvents.

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[36] For more information about the Vintaging model, see U.S. EPA (2018) and U.S. EPA (2020, pp. A-267–A-297). [U.S. EPA, 2018]. U.S. EPA, "EPA'S Vintaging Model of ODS Substitutes: A Summary of the 2017 Peer Review," EPA-400-F-18-001, 2018. Available at https://www.epa.gov/sites/production/files/2018-09/documents/epas-vintaging-model-of-ods-substitutes-peer-review-factsheet.pdf (Retrieved 10 December 2020).

[37] U.S. EPA, 2018.

[38] [ICF, 2019, pp. 22-23]. ICF, "User's guide for estimating carbon dioxide, nitrous oxide, HFC, PFC, $NF_3$, and $SF_6$ emissions from industrial processes using the State Inventory Tool," Prepared for the U.S. EPA, November 2019.

[39] U.S. EPA, 2020, p. A-267.

[40] [Gallagher, et al., 2014]. Gallagher, G., T. Zhan, Y.-K. Hsu, P. Gupta, et al., "High-global warming potential F-gas emissions in California: Comparison of ambient-based versus inventory-based emissions estimates, and implications of refined estimates," *Environmental Science & Technology*, 48, 1084-1093, 2014. http://dx.doi.org/10.1021/es403447v.

[41] CARB, 2016.

[42] CARB, 2018, pp. 9-10.

The types of F-gases used in new equipment for each year must be determined, along with the relative percent of equipment using each type of F-gas. For example, commercial refrigeration equipment has traditionally used high-GWP refrigerants, such as the HFC blend R-404A, with a GWP of 3922. Due to existing or potential regulations, lower-GWP refrigerants have been developed, including HFC blend R 407A (GWP 2107), HFO-HFC blends such as R-448A or R-449A (GWPs of 1386 and 1396), and the natural refrigerants carbon dioxide, ammonia, and hydrocarbons, with GWPs of one, zero, and less than five, respectively. U.S. EPA, industry data, and refrigerant usage data reported to CARB are used to determine the F-gas profiles of each type of equipment.

CARB can use the model to forecast emissions into the future as well as test the effect of various policy regimes, as long as these interventions represent the effects of the policies in terms of emissions drivers, such as GWP of F-gas, lifetime of equipment, or deployment of technology.

In partnership with the U.S. Climate Alliance (USCA),[43] CARB used the parameters and results from their F-Gas Inventory Model to estimate F-gas emissions under a variety of policy scenarios for each of the states and territories in the USCA.[44] This is the version of the model used in this study. Methodological and data details are discussed in the next section.

## 3.2 Methodology & Data

The study's forecast and analysis of F-gas emissions in Massachusetts is based largely on the CARB/USCA model, with some modifications, including:

- expansion of the model's scope to include F-gases other than HFCs from semiconductor manufacturing and gas-insulated switchgear;
- extension of the time horizon of the model from 2030 to 2050;
- replacement of certain emission scaling factors with more refined, state-specific values; and
- adjustment of certain assumptions about the start years and sectoral applicability of policy scenarios.

This section describes the methodology and data used to construct the reference emissions case and the approach for testing the effects of various policy interventions on F-gas emission through 2050. The reference case is defined based on current trends and policy interventions already in force.

### 3.2.1 CARB/USCA Model Utilization

The CARB/USCA model used here is derived from the CARB F-Gas Inventory Model discussed in the previous section. In order to adapt California's inventory and the effects of different policy scenarios to F-gas emissions from other USCA states and territories, CARB derived emissions intensity factors for 14 end-use sectors

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[43] The U.S. Climate Alliance is a coalition of U.S. states and territories representing over half of the U.S. population committed to implementing decarbonization policies in line with the Paris Agreement targets. The USCA also serves as a forum for members to share and improve their climate action plans, coordinate regional action, and identify and promulgate best practices.

[44] CARB, 2018.

through the year 2030.[45] Assuming these emissions factors to be consistent across states, the model then applies state-specific scaling factors (population, number of households with particular air conditioning technologies, and number of light-duty vehicles) to calculate total state-wide F-gas emissions. The end-use categories and scaling factors implemented in this model are presented in Table 3.

*Table 3. F-gas emissions end-use sectors and scaling factors in the CARB/USCA model*

| End-use sector | Scaling factor (emissions per…) |
|---|---|
| ***Refrigeration*** | |
| Commercial refrigeration | Person |
| Industrial refrigeration | Person |
| Domestic refrigeration | Person |
| Transport refrigeration | Person |
| ***Air Conditioning, Commercial*** | |
| Stationary AC, > 50 lbs. | Person |
| Stationary AC, < 50 lbs. | Person |
| ***Air Conditioning, Residential*** | |
| Heat Pump | Household using heat pumps |
| Central AC | Household using central AC |
| Room Unit AC | Household using room units |
| ***Air Conditioning, Motor Vehicle*** | |
| Light-Duty MVAC | Light-duty vehicles |
| Heavy-Duty MVAC | Person |
| ***Other*** | |
| Foam | Person |
| Aerosol Propellants | Person |
| Solvents & Fire Suppressant | Person |

The first major modification made to the model was to extend its temporal scope from 2030 to 2050. This study did not go back to the original CARB F-Gas Inventory Model, which might have enabled the extension of emissions factors using equipment turnover and other lifetime parameters. Instead, this approach utilized exponential curve-fitting to develop reasonable forecasts of emissions per person, per household with AC, or per vehicle. The emissions intensity curves, including both the CARB-derived values from 2010-2030 and the study's forecasts from 2030-2050, are presented in Appendix A.

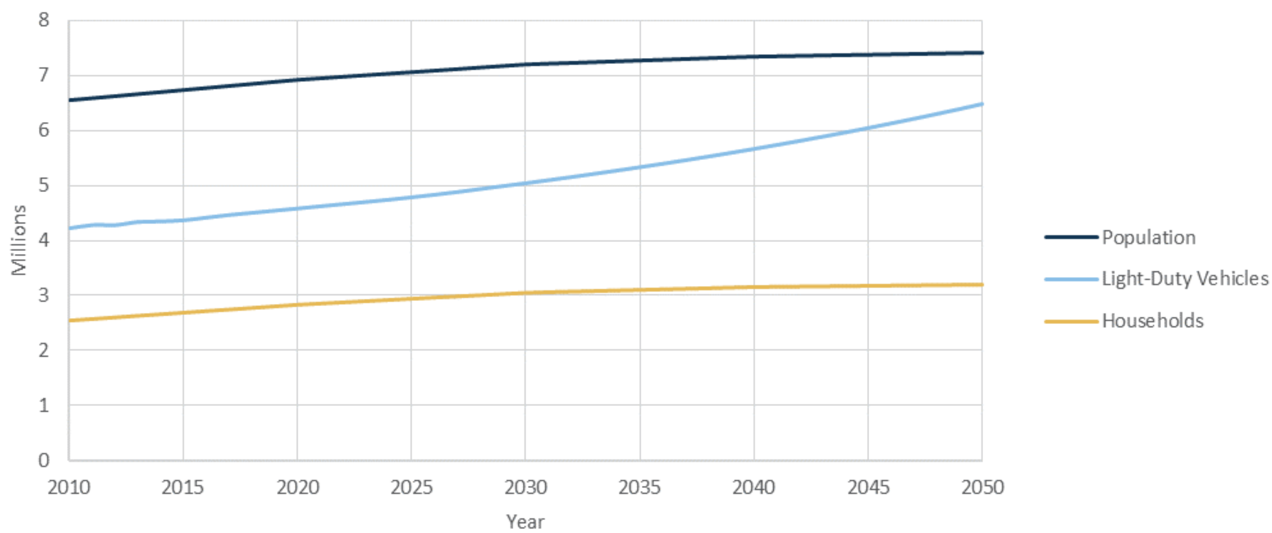
Most of the refrigeration and AC end-use sectors show an upward trajectory in their emissions intensity, reflecting two ongoing processes: a continued replacement of ODS with higher-GWP HFCs, and an increasing deployment of cooling technology. Of those sectors with upward emissions intensity trajectories, nearly all (except transport refrigeration and foam blowing) are shown to be leveling off, that is, the year-over-year

[45] The method for deriving these emissions intensity factors is very simple: CARB took the outputs from the F-Gas Inventory Model and divided each end-use sector by the California scaling factors. In essence, the CARB/USCA model does this operation in reverse but using other states' scaling factors. For more detail on the methodology and assumptions in this model, see CARB (2018).

increase in emissions intensity is decreasing. Emissions intensity factors for residential refrigeration, room unit AC, motor vehicle AC (both light-duty and heavy-duty), aerosol propellant, and solvents, are shown to be decreasing over time. Light-duty vehicle (LDV) AC reductions are particularly notable, decreasing from 220 kg $CO_2$e/LDV/year in 2010 to 30 kg $CO_2$e/LDV/year in 2030 and 3 kg $CO_2$e/LDV/year in 2050. This is because of trends already underway in the vehicle manufacturing industry to fully transition to low-GWP refrigerants like HFO-1234yf (GWP < 1) by the model year 2021.[46]

Scaling factors for Massachusetts were initially provided by CARB until 2030. The pre-existing population and household curves were replaced with data provided by the Massachusetts Executive Office of Energy and Environmental Affairs (EEA).[47] Statewide forecasts for population and households were provided every 10 years in baseline and high-growth scenarios. Forecasts for intervening years were established using linear interpolation. A forecast of the population of light-duty vehicles was provided by CARB, based on data from the U.S. EPA Motor Vehicle Emission Simulator (MOVES) model.[48] The forecast was extended to 2050 using an exponential growth model based on the data from 2010-2030. The forecasted population, households, and LDV curves are presented in Figure 7, with uncertainties discussed in section 3.5.2.1.3.

*Figure 7. Reference forecasts of population, number of households, and light-duty vehicles in Massachusetts, 2010-2050.*



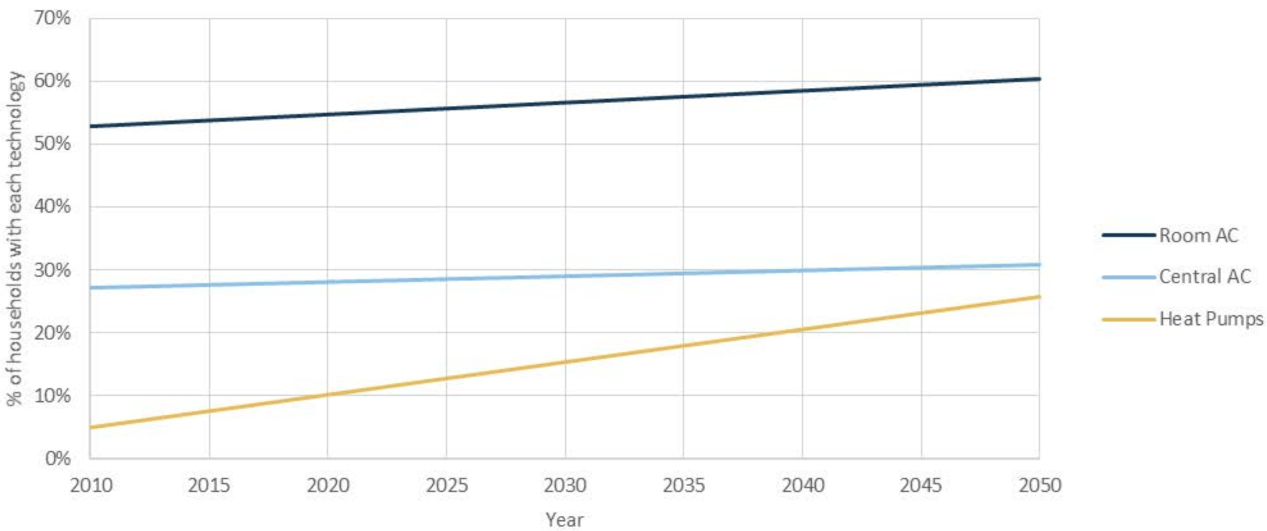

[46] [U.S. EPA, 2016]. U.S. EPA, "Proposed Determination on the Appropriateness of the Model Year 2022-2025 Light-Duty Vehicle Greenhouse Gas Emissions Standards under the Midterm Evaluation: Technical Support Document," EPA-420-R-16-021, 2016. Available at https://nepis.epa.gov/Exe/ZyPDF.cgi?Dockey=P100Q3L4.pdf (Retrieved 18 August 2020).

[47] The population projections used in this analysis were developed by the University of Massachusetts Donahue Institute (UMDI) and Metropolitan Area Planning Council (MAPC) for the Massachusetts Department of Transportation (MassDOT) Long-Range Transportation Plan (LRTP), included projections of population, households, and employment by municipality at decadal time-steps to 2040. MAPC and EEA contracted for UMDI to extend these projections to 2050 as part of their MetroCommons2050 planning effort and this report, respectively. UMDI and MAPC developed three different growth scenarios (baseline, high-, and low-growth); only the baseline and high population growth scenarios are evaluated in this report. Growth projections were generated prior to the emergence of the COVID-19 pandemic, the impacts of which, at current writing, are unknown regarding both population growth and building construction.

[48] CARB, 2018.

CARB calculated forecasts for deployment of residential heat pumps and air conditioning equipment using data from the 2005 and 2015 *Residential Energy Consumption Survey* conducted by the U.S. Energy Information Administration.[49] CARB assumed a linear extrapolation from these two data points to 2030, an assumption maintained here in extending the model to 2050. These deployment curves, defined as % of total households, are shown in Figure 8. The scaling factors used for residential heat pumps and ACs are calculated by multiplying each of these curves by the total # of households shown in Figure 7.

*Figure 8. Reference case forecasts of the deployment of residential heat pumps and AC equipment in Massachusetts, 2010-2050.*




The reference case for this subsector assumes no additional decarbonization policies are introduced in the future, including policies to encourage the adoption of residential heat pumps. The relatively low deployment of residential heat pumps by 2050 illustrated in Figure 8 is based on an extrapolation of current growth rates of this technology. The implications of aggressive residential electrification policies that would result in the widespread use of heat pumps are explored in the uncertainty and sensitivity analysis section 3.5.2.1.2 below.

### 3.2.2 Semiconductor & $SF_6$ Emissions Modeling

The CARB/USCA model is limited to short-lived F-gases, namely HFCs, excluding PFCs, $NF_3$, and $SF_6$ emissions. For this study, minor modifications and additions to the model were made to attempt to represent the admittedly small contributions of these long-lived F-gases to the state inventory. To do this, two new end-use categories were added to the model: semiconductor manufacturing and $SF_6$ from gas-insulated electrical switchgear.

The HFC emissions from semiconductor manufacturing is already accounted for in the CARB/USCA model "solvents and fire suppressants" end-use category. The modeled output values of this category were compared with the Massachusetts GHG Inventory, which accounts for semiconductor manufacturing F-gas emissions by downscaling national semiconductor manufacturing F-gas emissions by industrial output in the state. The model calculates that the entire "solvents and fire suppressants" end-use category generated 0.05 $MMTCO_2e$

[49] CARB, 2018.

of HFC emissions in 2017. The Massachusetts GHG Inventory, on the other hand, showed 0.35 $MMTCO_2e$ of F-gas emissions from semiconductor manufacturing in the same year.[50] In order to rationalize these two values, it is assumed that the bulk of the semiconductor emissions must be non-HFC F-gases.[51] Following the data distribution in the 2017 California F-Gas Inventory, the contribution of semiconductor emissions to the "solvents and fire suppressants" end-use sector is assumed to be 25% of the total, or 0.01 $MMTCO_2e$.[52] Here, that quantity is removed from the "solvents and fire suppressants" category and the data from the Massachusetts GHG Inventory is used to populate the new "semiconductor manufacturing" end-use category. Because of limited evidence substantiating whether emissions from this sector are likely to increase or decrease over the next three decades, this study assumes that the 2017 value remains constant until 2050.

Sulfur hexafluoride emissions from the electric power transmission and distribution system is excluded from the CARB/USCA model altogether. The Massachusetts GHG Inventory includes an estimate of this emission source: 0.06 $MMTCO_2e$ from gas-insulated switchgear and other $SF_6$ sources in 2017. The forecast of these emissions assumed a continuation of the $SF_6$ leak mitigation policy currently in force, which is expected to reduce emissions to 0.05 $MMTCO_2e$ by 2020 and then keep future emissions constant through 2050.

### 3.2.3 Policy scenarios

The CARB/USCA model implements policy scenarios by first calculating the effect of each policy on the California F-Gas Inventory. For each of the 14 end-use sectors, CARB then calculates the percentage deviation from the California business-as-usual scenario caused by each policy in each year. This deviation is then applied to the business-as-usual scenario calculated for each USCA state and territory.

Two changes were made to this method for evaluating the effect of F-gas mitigation policies on F-gas emission in Massachusetts. First, the starting year for any policy implementation scheduled to take effect before 2021 was pushed back to that year, acknowledging the reality of when F-gas mitigation policies could first be put in place in Massachusetts. Second, the analysis was extended from 2030 to 2050. Just as was done with the extrapolation of emissions intensities, exponential and logarithmic functions were used to extend the deviation curves. The main exception to this was in implementing the Kigali Amendment, where the deviation curve had to be shifted to meet the phasedown schedule shown in Figure 6. More detail on the ways that CARB implemented each policy scenario can be found in the CARB/USCA model methodology document.[53]

## 3.3 Reference Case

The reference case for Massachusetts F-gas emissions assumes emissions grow according to the forecasts presented above with no new emissions reduction policies. The resulting scenario is illustrated in Figure 9. In this reference case, F-gas emissions in Massachusetts increase from 2.56 $MMTCO_2e$ in 2010 to 5.93 $MMTCO_2e$ in 2050, a growth of over 130%. The growth is driven mainly by increases in HFC emissions from refrigeration

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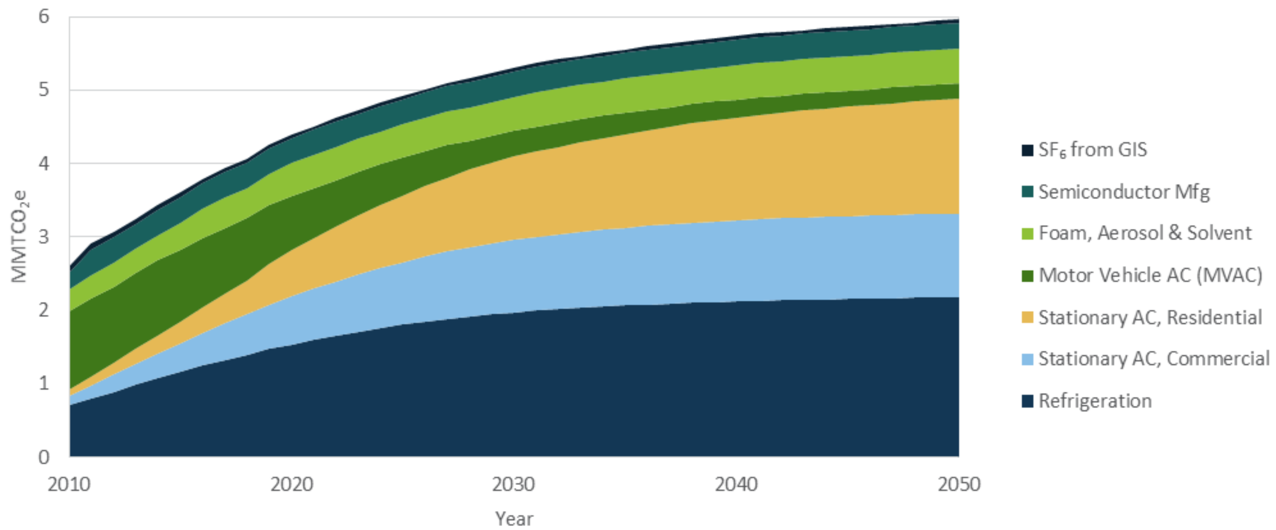
[50] MassDEP, "MA GHG Inventory," 2020.
[51] This assumption is borne out by data from the national GHG Inventory, which accounts for 4.8 $MMTCO_2e$ of F-gas emissions from the electronics industry in 2018, 4.4 $MMTCO_2e$ of which were non-HFCs (3.0 $MMTCO_2e$ PFC, 0.8 $MMTCO_2e$ $SF_6$, and 0.6 $MMTCO_2e$ $NF_3$) (U.S. EPA, 2020).
[52] CARB, 2018.
[53] CARB, 2018.

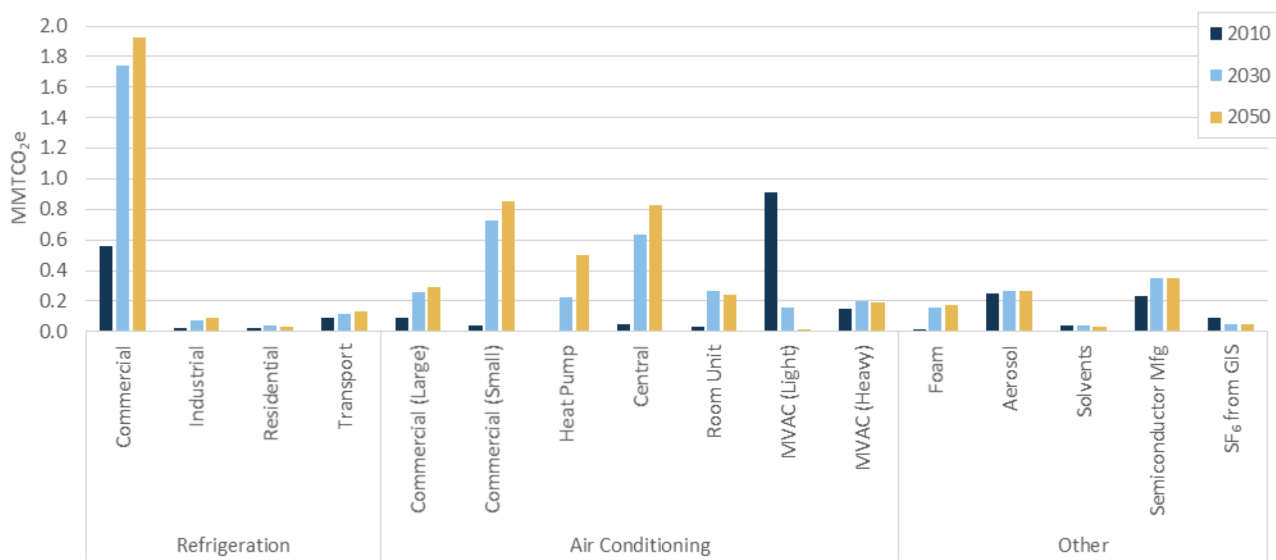
and stationary air conditioning. Motor vehicle AC is forecast to decrease significantly, while the remaining categories experience minor-to-no growth.

*Figure 9. Reference case of F-gas emissions in Massachusetts, 2010-2050*




Looking closer at the 16 end-use categories in this model (the 14 from the CARB/USCA model plus semiconductor manufacturing and $SF_6$ from GIS), one can see that responsibility for growth in the reference scenario is not evenly distributed across end-use. Figure 10 shows emissions from each of the 16 categories in 2010, 2030, and 2050. Over the modeling period, some end-use sectors experience emissions growth from nearly nothing: small commercial AC, residential AC (heat pumps, central AC, and room units), and foam. Commercial refrigeration is the largest single component of the inventory and also sees tremendous growth. The decrease in MVAC is driven by light-duty vehicles, as discussed above. Aerosols and semiconductors remain more or less stable, and the rest of the end-use sectors are relatively minor.

*Figure 10. Reference case forecast of F-gas emissions in Massachusetts by end-use sector, 2010, 2030 & 2050*



## 3.4 Reduction Pathways

CARB designed eight F-gas reduction scenarios in the CARB/USCA model that are adopted in this analysis: four scenarios each with a single, distinct policy, and four scenarios built from combinations of those policies. The resulting potential reduction pathways from this implementation of those eight scenarios applied in Massachusetts are illustrated in Figure 11. The combination scenarios are not simply additive, as two policies may target the same end-use sector. In these cases, CARB identified the specific mechanisms by which the policies operate and give preference to the more stringent policy operating in any given year. The policies explored were described in Section 1.2.1 and in CARB (2018).[54]

- A **Refrigerant Management Program (RMP),** based on California's policy, could reduce leaks from commercial and industrial refrigeration systems, resulting in a decrease in emissions by about 7% compared to the reference case by 2050. Even with an RMP, the analysis indicates that overall emissions will still increase from 2010 to 2050 by 113%.
- **Implementation of the SNAP rules 20 and 21 at the state level**, prohibiting the use of high-GWP HFCs in certain end-use sectors, would result in a 21% reduction in F-gas emissions below the reference case by 2050. The SNAP rules target key sources of growth of F-gas emissions, and if implemented in 2021 would limit the growth in overall emissions to 10% between 2021 and 2050, whereas the reference case sees 32% growth from 2021 to 2050. MassDEP's proposed regulation 310 CMR 7.76 which adopts portions of the partially-vacated SNAP rules has a prohibition of many HFCs and HFC-blends starting in 2021. Still, the analysis indicates an 80% growth in emissions from 2010 to 2050 even with the SNAP rules.
- **Combining SNAP with an RMP has an additive effect**: first, SNAP rules lower the GWP of F-gases used in chillers, foams, aerosols, etc. Then, the RMP reduces leaks from large refrigerators and air conditioners. This combination scenario is estimated to reduce emissions by 25% below the reference case by 2050. However, even with SNAP and RMP, emissions still increase 72% between 2010 and 2050.
- **The short-lived climate pollutants (SLCP) policy strategy** proposed by CARB bans high-GWP refrigerants from use in new stationary chillers and air conditioners. This aggressive policy is the first to show a reduction in emissions from 2010 levels: 60% below the reference case by 2050, 46% below the 2021 level, and 7% below the 2010 level.
- **Combining all three standalone policies** introduced so far—RMP, SNAP, and SLCP—has an even more significant effect on overall emissions and emissions trajectory. SNAP and SLCP are similar policies, but with different starting years and different cutoff rules. For example, some allowable SNAP refrigerants have GWP in excess of 2000, while SLCP requires they be below 750. For this study, whichever rule is more stringent for each end-use sector in each year was determined to be in force. The RMP layers on top of the GWP-reduction policies. This combination scenario results in emissions 65% below the reference case by 2050, a 51% reduction from 2021 levels and a 21% reduction from 2010 levels.
- The final three scenarios are all operating under a **Kigali global HFC phasedown** and assume that the United States becomes a party to the international agreement. These scenarios all result in 2050

[54] There are other approaches for reducing F-gas emissions, including improved training for refrigerant handlers, economic incentives and disincentives, and building code requirements for improved fittings and pipes. The list of policies discussed here are those that have been analyzed by the CARB/USCA model.

emissions levels between 0.9 and 1.0 $MMTCO_2e$. This reduction is roughly 85% below the reference case in 2050, 78% below 2021 levels, and 63% below 2010 levels. The assumptions made by CARB as to the way that Kigali rules would impact individual end-use sectors are complicated and discussion of them is out of scope for this report. For more information about the anticipated effect of Kigali on California's F-Gas Inventory, see CARB (2017) and CARB (2018). CARB modeled best- and worst-case outcomes from the implementation, a range that is explored in the Sensitivity Analysis section below. Because Kigali rules will only affect new production of HFC gases, it is most immediately felt in short-lifetime products like aerosols and may take years or decades to be felt in stationary chiller systems. Timely implementation of Kigali by federal actions is critical for significantly reducing F-gas emissions by 2050. By combining Kigali rules with policies like RMP, SNAP, and SLCP, F-gas emissions can be reduced even quicker than a standalone HFC phasedown could achieve, as shown in Figure 8.

Table 4 summarizes the quantitative results of the eight reduction scenarios tested in this study.

*Figure 11. F-gas emissions scenarios for Massachusetts, 2010-2050*

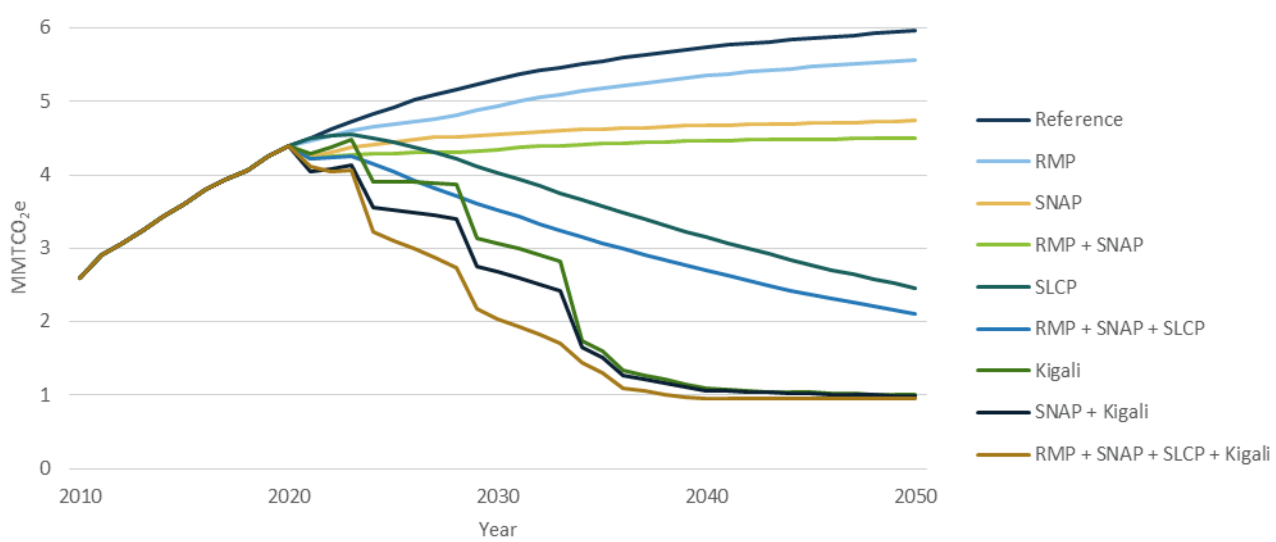



*Table 4. Changes to overall F-gas emissions in Massachusetts in different reduction scenarios*

| Reduction Scenario | % below reference in 2050 | % increase/decrease from 2021 | % increase/decrease from 2010 |
|---|---|---|---|
| ***Reference*** | 0% | 32% | 129% |
| ***RMP*** | 7% | 25% | 114% |
| ***SNAP*** | 21% | 11% | 82% |
| ***RMP + SNAP*** | 24% | 7% | 73% |
| ***SLCP*** | 59% | -45% | -5% |
| ***RMP + SNAP + SLCP*** | 65% | -50% | -19% |
| ***Kigali*** | 83% | -77% | -62% |
| ***SNAP + Kigali*** | 83% | -76% | -62% |
| ***RMP + SNAP + SLCP + Kigali*** | 84% | -77% | -64% |

## 3.5 Uncertainty & Sensitivity Analysis

### 3.5.1 Sources of uncertainty

There are three main sources of uncertainty in the modeling approach that have been taken here: representativeness of the data; accuracy of the forecast; and unknown unknowns. First, with the exception of non-HFC emissions, the empirical data used to construct the inventory and forecast is from California. This includes both the emissions data and the effects of different policies. The model hinges on an assumption "that the number of refrigeration, air-conditioning equipment and units using HFCs correlate highly with population," allowing California data to be used to derive emissions factors that are applicable in all states and territories.[55] This foundational assumption in CARB's model may not be accurate. Many characteristics are likely to affect the number and type of different HFC-generating activities in the state, including levels of urbanization, average income, population density, industrial makeup, climate differences, and the age of the building stock, among others. However, California, being a very large and industrially-diverse state, does offer a balanced inventory.

Second, the extension of the forecast from 2030-2050 was performed without access to the CARB F-gas Inventory, which is not publicly available. Therefore, the forecast may not accurately reflect all of the stock dynamics that determine emissions in a vintaging model.

Finally, and relatedly, there are aspects of future F-gas emissions that are not known well-enough to represent them in this model. One example is the rate of F-gas leakage from heat pumps deployed at large scales. These are relatively new technologies in the Massachusetts marketplace, and as they become more standard, it remains to be seen if best practices are adopted for service and end-of-life management. Federal policies regulating the use of high GWP refrigerants would help to mitigate future F-gas emissions from heat pumps. Another example is the future of semiconductor manufacturing. Ever-increasing purity requirements and smaller-scale etching may lead to notable changes in the demand for F-gases in an industrial context.

### 3.5.2 Uncertainty analysis & discussion

To analyze some of the sources of uncertainty in this model, a basic sensitivity analysis is performed, varying key input parameters and evaluating the effect on model output. Three areas of uncertainty are explored: scaling factors, non-HFC emissions, and policy scenarios.

#### 3.5.2.1 Scaling factors

The scaling factors used in this model are population; number of households; percentage of households with heat pumps, central air conditioners, and room-unit air conditioners; and number of light-duty vehicles. This subsection discusses effects of variability of each of these parameters.

[55] CARB, 2018, p. 21.

#### *3.5.2.1.1 Population & households*

Two scenarios of population and household growth were evaluated. All of the analysis performed above utilized the *baseline population growth* scenario, which forecasts 7.42 million people and 3.19 million households in Massachusetts by 2050. The *high population growth* scenario assumes total population and household numbers are 6.9% and 7.4% larger, respectively, by 2050. All else being equal, the *high population growth* scenario leads to similar increases in total annual emissions in 2050 under the reference case (6.6% increase from the *baseline population growth* scenario) and under a Kigali HFC phase-out policy scenario (6.5% increase from the *baseline population growth* scenario). This analysis shows that there do not seem to be strong non-linearities in the relationship between population (and household) growth and HFC emissions: if population grows faster than expected, emissions will too, and roughly proportionately. The same logic applies if population and households grow slower than expected.

#### *3.5.2.1.2 Residential heat pumps and AC equipment*

HFC emissions from residential heat pumps and AC systems are a relatively minor contributor to the F-gas inventory in 2020, but are anticipated to become the second most important source of emissions by 2050 in the reference case because of increased use of both central AC and heat pumps. It is therefore important to interrogate the validity of the relevant scaling factors that lead to this result. The CARB/USCA model tracks the annual emissions intensity of three residential system types: central AC, room-unit AC, and heat pumps; measured in $MMTCO_2e$/household. As explained above, for this model CARB extrapolated linear growth in technology deployment from two studies of residential energy use: one in 2005 and the second in 2015. There is no reason to necessarily expect linear growth for 35 more years; it is likely either that technology deployment will reach an asymptote at some point or that policies will encourage the acceleration of technology deployment. Further, the model does not limit households to a single system type, that is, the sum of the three percentages is not limited to 100%. In fact, while in 2015 89% of Massachusetts households were thought to have AC or heat pumps, by 2030 in the reference case this number exceeds 100%. It is not unreasonable that a single household may have more than one system: consider a house with two HVAC zones, one of which is covered by a central AC system and the other by window units. This nevertheless complicates the effort to project deployment rates into the future.

Two analyses are presented here to explore sensitivity and uncertainty in residential heat pump and AC equipment deployment rates. First, a sensitivity of the reference case assumptions looks at relatively minor (10 percentage point) variations. Second, an analysis of a scenario that achieves 100% deployment of heat pumps by 2050, which is more in line with pathways investigated elsewhere in the companion reports, including the *Energy Pathways Report, the Buildings Sector Technical Report,* and the *Roadmap Report,* than is the reference case.

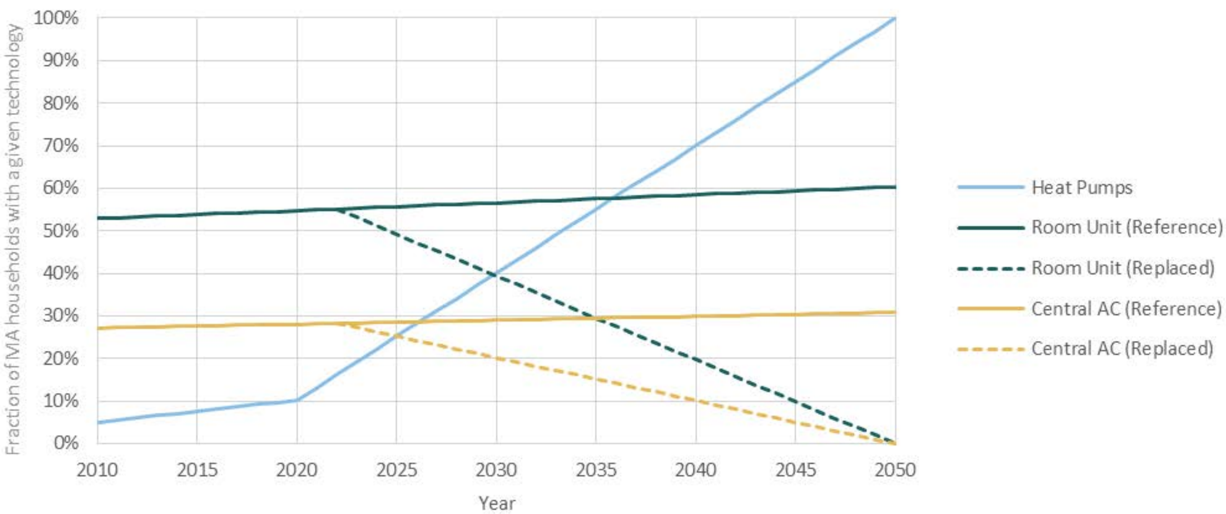
To evaluate the effect of a particular technology deployment rate on overall F-gas emissions, a simple sensitivity analysis is performed: varying the 2050 deployment percentages of each of the three residential system types by 10 percentage points up and down from the reference values and observing the effect on the overall F-gas emissions from residential AC. Results are presented in Table 5. This sensitivity analysis shows that in the reference case deployment of residential heat pumps and AC equipment, central AC and heat pumps have stronger effects on emissions than room AC. Varying central AC by 10 percentage points results in an emissions swing of 17% in the reference case and 14.5% in the Kigali reduction scenario. The same variation of heat pumps results in 12.4% and 10.6% variation, respectively. Varying room unit AC deployment has a larger effect under Kigali: 4.6% swing versus a 2.6% swing in the reference case.

*Table 5. Sensitivity analysis of residential AC deployment, varying by 10 percentage points up and down in 2050 under reference case and Kigali HFC phasedown scenario*

| Scenario | | Deployment % | Residential AC F-Gas Emissions ($MMTCO_2e$) | |
|---|---|---|---|---|
| | | | *Reference* | *Kigali phase-out* |
| **Central AC** | -10% | 20.8% | 1.30 | 0.24 |
| | 0 | 30.8% | 1.57 | 0.28 |
| | +10% | 40.8% | 1.84 | 0.32 |
| **Room AC** | -10% | 50.3% | 1.53 | 0.26 |
| | 0 | 60.3% | 1.57 | 0.28 |
| | +10% | 70.3% | 1.61 | 0.29 |
| **Heat Pumps** | -10% | 15.8% | 1.38 | 0.25 |
| | 0 | 25.8% | 1.57 | 0.28 |
| | +10% | 35.8% | 1.77 | 0.31 |

Heat pumps present a particular challenge to forecasting F-gas emissions, as they are widely seen as vital technologies for decarbonizing residential heating.[56] A second sensitivity analysis therefore considers two scenarios in which heat pump penetration reaches 100% of households in 2050: one in which growth in central AC and room-unit ACs continue as estimated by CARB—resulting in many households having both a heat pump and an AC unit—and another in which heat pumps fully replace all residential AC systems. These options represent boundary conditions to the heat pump adoption findings in the All Options pathway in the *Energy Pathways Report.* These deployment scenarios are illustrated in Figure 12.

*Figure 12. Residential heat pump and AC deployment scenarios in the case of 100% deployment of heat pumps by 2050.*




[56] [Kaufman, et al., 2019]. Kaufman, N., D. Sandalow, C. Rossi di Schio & J. Higdon, "Decarbonizing Space Heating with Air Source Heat Pumps," Columbia University Center on Global Energy Policy, 2019. Available at https://www.energypolicy.columbia.edu/research/report/decarbonizing-space-heating-air-source-heat-pumps (Retrieved 10 November 2020).

This 100% heat pumps sensitivity yielded several results and increased F-gas emissions from residential AC considerably:

- In the reference case, emissions nearly double from 1.57 $MMTCO_2e$ in 2050 to 3.02 $MMTCO_2e$, if heat pumps do not replace central and room unit AC systems. If non-heat pumps are fully replaced with heat pumps, F-gas emissions were anticipated to increase to 1.95 $MMTCO_2e$ in 2050.
- Under a Kigali phasedown, emissions increase from 0.28 $MMTCO_2e$ to 0.49 $MMTCO_2e$ without system replacement. The emissions stay nearly constant at 0.29 $MMTCO_2e$ if non-heat pumps are fully replaced.

The validity of emissions factors is a source of uncertainty, particularly around the future of space conditioning. Technology change is rapidly occurring; in a conventional residential central AC system, warm air is ducted to a centralized compressor where refrigerant is used to remove heat from the air, which is then recirculated into the house. In a ductless system, instead of the air being circulated, it is the refrigerant that is circulated through the house. This means that, depending on the length of the pipes, these systems may have a larger refrigerant charge size, different leak profile, and different lifetime than conventional central AC systems. The policy implications of heat pump adoption and uncertainty are addressed in section 6.

In commercial and industrial applications, the analogous technology is known as variable refrigerant flow. Like ductless mini-splits and heat pumps, these large-scale systems utilize long refrigerants lines with potentially more refrigerant—and more opportunity to leak—than the conventional systems they are increasingly replacing. The potential for technological lock-in, and associated emission lock-in, with both VRFs and ductless residential systems means it is especially important to adopt low-GWP refrigerants in these systems.

### *3.5.2.1.3 Light-duty vehicles*

HFC emissions from light-duty vehicle (LDV) air conditioning are represented as scaling with the size of the LDV population.[57] The baseline LDV population figures until 2030 were drawn by CARB from the U.S. EPA MOVES model, and the observed exponential growth is extended to 2050. In the reference case, HFC emissions from LDV AC are already expected to decline to 0.017 $MMTCO_2e$ by 2050, even as LDV population increases to 6.5 million vehicles. This is because there are already national-level policies and agreements under which high-GWP refrigerants in LDV AC systems will be phased out and replaced with low-GWP alternatives in order to reduce overall GHG emissions from the transportation sector.[58] Accordingly, none of the F-gas mitigation policy scenarios and combination scenarios examined in this report include any further cuts to emissions from LDV AC systems.

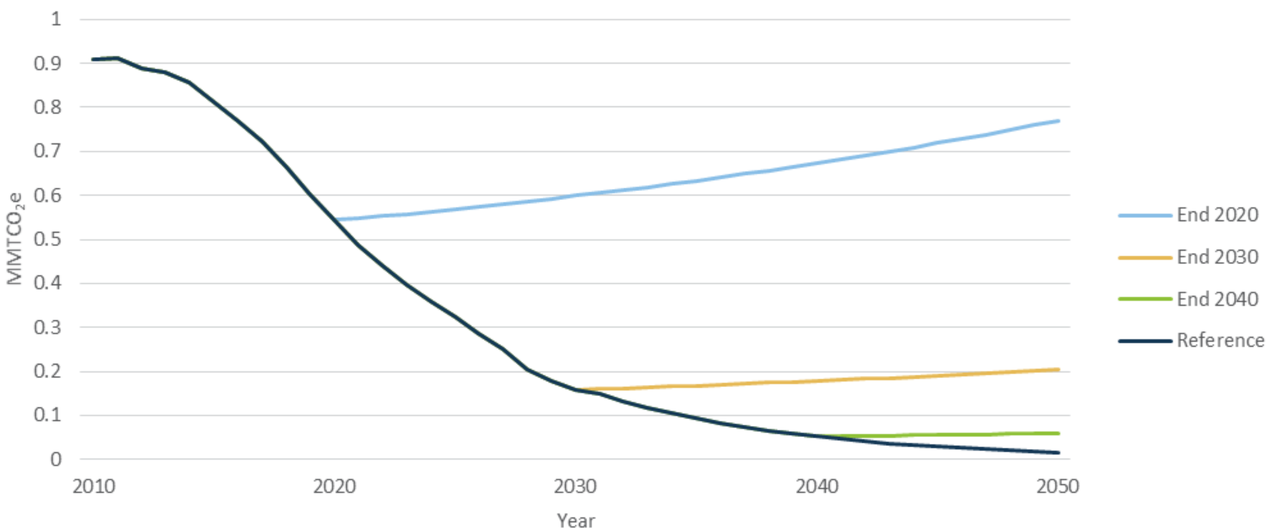
Two sources of uncertainty related to LDVs are explored. First, the effect of a 10% increase or decrease to the vehicle population by 2050 is tested, which produces a range between 5.83-7.13 million LDVs. Because the anticipated, ongoing substitution of LDV refrigerant yields an emissions factor of just $2.58x10^{-9}$ $MMTCO_2e/LDV$ by 2050, uncertainty in total number of vehicles has very little effect on the end result: a range of emissions

[57] This study assumes that refrigerant usage in LDVs does not vary with respect to drive-train (i.e., internal combustion, hybrid, or electric).
[58] U.S. EPA, 2016.

from 0.015-0.18 $MMTCO_2e$ in 2050, a change of roughly 0.3% of the total reference case emissions and less than 2% of the Kigali HFC phasedown scenario emissions.

The second source of uncertainty is whether the LDV refrigerant substitution will proceed as anticipated and continue all the way to 2050. To test this uncertainty, hypothetical scenarios are examined in which the emissions factor reduction halts at 2020, 2030, and 2040, representing cases in which any further refrigerant substitution after those years ceases. According to the model, LDV emissions are anticipated to reach $1.19 \times 10^{-7}$ $MMTCO_2e$/LDV by 2020, $3.15 \times 10^{-8}$ $MMTCO_2e$/LDV by 2030, and $9.29 \times 10^{-9}$ $MMTCO_2e$/LDV by 2040. The resulting emissions curves are presented in Figure 13. Upward trajectories in each of the emissions curves reflect a continual increase in the number of LDVs on the road until 2050. This analysis shows that LDV refrigerant substitution is important to overall decarbonization goals, although it is important for the high-GWP refrigerants to be phased out earlier than later, as illustrated in the 0.57 $MMTCO_2e$ difference between 2050 emissions if the 2020 emissions factor were locked in compared to if the 2030 factor were locked in.

*Figure 13. HFC emissions from LDV AC if anticipated reductions are ended in 2020, 2030, and 2040.*




### 3.5.2.2 Non-HFC emissions

In this analysis, roughly 6% of total F-gas emissions in both the reference case and Kigali scenario are contributed by non-HFC sources: semiconductor manufacturing and $SF_6$ leakage from GIS. These two sources are modeled independently of the CARB/USCA model and have their own sources of uncertainty that are addressed here.

#### *3.5.2.2.1 Semiconductor manufacturing*

Current F-gas emissions from semiconductor manufacturing arise from a small number of industrial facilities in Massachusetts. As a result, these emissions are highly sensitive to relatively small changes in the industrial makeup of the state. If Massachusetts attracts new electronics manufacturing, PFC and $NF_3$ emissions could rise significantly. On the other hand, if even one of the firms whose operations emit F-gases either changes their emissions control systems or closes down their operation in Massachusetts, those emissions could decline.

The policy scenarios explored in this study focus explicitly on HFC emissions reduction and are not intended to affect non-HFC emissions. In the analysis above, the only scenarios that affect semiconductor manufacturing emissions are those in line with the Kigali global HFC phasedown, which reaches 85% reductions from the baseline by 2050. The data from the Massachusetts GHG Inventory are not broken out by F-gas, so it could not be determined what fraction of the reported value would be affected by Kigali policies and what would not.

To address both uncertainties, semiconductor emissions are varied by 10% and scenarios are considered in which the Kigali reductions are both applied and not applied. Results are presented in Table 6. The main takeaway is the significance of applying the Kigali reduction assumptions to semiconductor emissions on the overall Kigali scenario. With the assumptions listed above, that scenario would result in annual emissions of just 0.96 $MMTCO_2e$ by 2050. If semiconductor emissions were excluded from the phase-out policies, the total would be 1.26 $MMTCO_2e$, an increase of 31%.

*Table 6. Results from sensitivity analysis of F-gas emissions from semiconductor manufacturing, 2050.*

| Emissions scenario | Emissions, 2050 ($MMTCO_2e$) | |
|---|---|---|
| | ***No reduction*** | ***Kigali reduction*** |
| ***-10%*** | 0.31 | 0.05 |
| ***0*** | 0.35 | 0.05 |
| ***+10%*** | 0.38 | 0.06 |

### *3.5.2.2.2 $SF_6$*

Sulfur hexafluoride has a GWP-100 of 23,500, making it the most powerful GHG evaluated by the IPCC. It is used in many electrical transmission and distribution system and industrial applications. The Massachusetts GHG inventory of $SF_6$ emissions is based on a downscaling of national data using electricity sales in each state as the scaling factor. This approach is incapable of accounting for any specific state-based policies that would yield a different emissions factor (kg $SF_6$/MWh electricity sold) than the rest of the country. On the other hand, the other source of data, EPA FLIGHT,[59] only accounts for a subset of $SF_6$ users (and therefore, emitters) in the state.

The reference case assumes that the currently observed rate at which $SF_6$ emissions from GIS are decreasing in Massachusetts (an average reduction of 5.5% per year) continues through 2020 and then remains constant through 2050. The current $SF_6$ emissions mitigation regulation (310 CMR 7.72) sets a leakage cap on equipment owned by major utilities of 1% by 2020, controlling for any growth in the transmission network. As of now, there are no plans to further tighten this regulation, so the emissions level in 2020 (0.05 $MMTCO_2e$) is likely to be maintained through 2050. If emissions reductions were to continue at the same rate that they are being observed today, $SF_6$ emissions in 2050 would be less than 0.01 $MMTCO_2e$.

[59] EPA Facility Level Information on GreenHouse gases Tool (FLIGHT). https://ghgdata.epa.gov/ghgp/main.do?site_preference=normal

California regulators have proposed requiring utilities to phase out the use of $SF_6$ in GIS and other electrical equipment.[60] This regulation would require all gas-insulated electrical equipment to be replaced with technology that uses insulating gas with lower GWP than $SF_6$ or some alternative, zero-GWP technology. As of now, the market for these non-$SF_6$ equipment is immature, but as more jurisdictions set schedules for replacement, it is expected that new options will emerge.

The other main source of uncertainty for $SF_6$ emissions has to do with the size of the electricity transmission and distribution system in the future, and as a result the amount of $SF_6$-using equipment. Under a decarbonization strategy that involves mass electrification of heating and transportation, enormous amounts of new electrical infrastructure may have to be built, requiring new switchgear, potentially leading to new sources of $SF_6$ emissions that may not be subject to the current regulation.

#### 3.5.2.3 Policy scenarios

The policy scenarios explored in this report were developed using the California F-gas inventory. Their applicability to Massachusetts depends on the accuracy of the inventory itself and the responsiveness of HFC-emitting activities to policies in Massachusetts versus in California. Two sources of uncertainty have been identified regarding the policy scenarios that can help to offer insight into the reliability of the results of this study.

First, the original CARB/USCA model assumed the start dates for some policies as early as 2019. In this project, it is assumed that the earliest a policy can be implemented is 2021, with resulting changes in the reduction scenarios. It was found that there was no observable difference in the annual emissions by 2050 whether reductions started in 2019, 2020, or 2021. If reductions are postponed much further into the future, however, it is possible that the deep cuts to F-gas emissions necessary by 2050 will not be achievable. This is because increasing numbers of new, conventional equipment will be installed and will have to wait until its end-of-life to be replaced with a low-GWP substitute. This scenario could not be tested quantitatively for lack of access to the internal structure of the model, but some insight is available from the model documentation. The average lifetime of typical large, stationary refrigeration equipment is 15-20 years. That means a new commercial chiller installed today may be operating until 2035-2040, or potentially even longer. If that chiller is charged with a high-GWP refrigerant, leakage will continue over the entire lifetime of the appliance. If decarbonization policies are delayed to the point where there less than one generation of equipment between the onset of the policy and the target date, achieving F-gas emissions reductions would require other policy actions such as accelerated retirement of certain equipment and other aggressive interventions.

Second, there is uncertainty about the forecast of policy scenarios from 2030-2050, which, as mentioned above, were done by extrapolating the reduction curves that were defined for the period 2010-2030 using the California F-gas Inventory. The extrapolations were done as smoothly as possible, but there are possibly some asymptotes or other changes in the curve function that were not observable before 2030 but would come into

[60] [CARB, "$SF_6$," 2020]. CARB, "Staff report: Initial statement of reasons," Regulation for Reducing Sulfur Hexafluoride Emissions from Gas Insulated Switchgear, 2020. Available at https://ww2.arb.ca.gov/rulemaking/2020/sf6 (Retrieved 28 September 2020).

play before 2050. Again, this is not a reliably testable source of uncertainty, but a caveat that the policy scenarios may illustrate a lower limit of emissions by 2050 rather than an average.

One important instance of uncertainty in a forecasted policy scenario is the Kigali global HFC phasedown. CARB acknowledges that the Kigali scenario is much less straightforward than the other policies because it does not specify where and how HFC reductions are to occur, just that they are. CARB developed a "worst-case" and "best-case" reduction scenario for Kigali and estimated the average to be implemented in the USCA model. For example, in the worst case, old equipment would continue to be charged (and leak) high-GWP refrigerant that "could be supplied by the shrinking supply of new high-GWP refrigerant manufactured, from reclaimed and recycled refrigerant, through stockpiling of refrigerant, and from illegally imported refrigerants."[61] In the best case, manufacturers and operators of refrigerants would anticipate the restriction and voluntarily substitute low-GWP refrigerants for high-GWP refrigerants. The average case is somewhere in between, as defined by CARB:[62]

1. Most new equipment will use much lower GWP refrigerants than the required phasedown step,
2. Most existing equipment will continue to be serviced with the same high-GWP refrigerant it was originally designed to use, and
3. Some existing equipment will be retired early or retrofitted to use lower-GWP refrigerants.

The Kigali reduction schedule reaches its final step, 85% below baseline, in 2036. The original CARB/USCA model extends to 2030, which means that it only goes through the third-to-last reduction step in the Kigali schedule: 70% below baseline by 2029 (Figure 6). But the defined schedule is always quite different from the as-implemented schedule, since implementation depends on compliance with a set of policies, plus the inherent delays in reducing emissions from a sector that depends on equipment stock turnover. The CARB/USCA model shows, for instance, a reduction of just 15.1% below reference of HFC emissions from residential refrigerators by 2030, far from the target 70% reduction below the 2011-2013 average HFC emissions level (which is how Kigali defines the schedule). The implementation of Kigali in this study does achieve a reduction in HFC emissions of 85% below the reference case by 2050, but it gets there with some potentially unrealistic jumps. For example, commercial and industrial refrigeration jumps from a 50% reduction below baseline in 2033 to 73% reduction below baseline in 2034. This pattern follows one set by CARB in earlier years (for example, the jump from a 26% to a 42% reduction in those same end-use activities in 2028-2029), but it is a source of uncertainty. Ultimately, this analysis was intended to show the effect in Massachusetts if Kigali were implemented worldwide. The details of its implementation and the resulting market dynamics will require another, more in-depth study.

## 3.6 Discussion

### 3.6.1 Summary of policy conclusions

F-gas emissions in Massachusetts are estimated to be 4.40 MMT$CO_2$e in 2020. Without any policy intervention, this is expected to increase to 5.96 MMT$CO_2$e by 2050. The analysis in this report, based on previous work and

[61] CARB, 2018, p. 33.
[62] CARB, 2018, p. 35.

modeling by CARB, shows that it may be possible to achieve deep cuts to this emissions curve. By deploying a set of policies that are compliant with the Kigali Amendment to the Montreal Protocol, Massachusetts F-gas emission can be reduced to below 1.0 $MMTCO_2e$ by 2050.

The analysis reveals some challenges with this prescription. First, Kigali only addresses HFCs, whereas it can be seen that there are important non-HFC emissions that must be reduced as well. Massachusetts has a regulation focused on $SF_6$ emissions, and, in order to reduce emissions, would need to continue requiring absolute reductions in $SF_6$ leakage from electrical equipment, even as mass electrification increases the opportunities for leakage. Second, the Kigali Amendment is intended to be a national program, whereby a national government sets limits on HFC production and consumption. In the United States (which has not ratified the Amendment), individual states have very limited ability to phase-out HFCs holistically in this way. Instead, it requires policies that target specific compounds, end-uses, and gas management practices. The SNAP-style regulations that have been proposed in Massachusetts (310 CMR 7.76) are a first step, but the analysis demonstrates that they are insufficient to avoid continued growth of F-gas emissions, especially from expected growth of heat pumps which are not covered by the proposed regulation, and certainly insufficient to reduce emissions to minimum levels by 2050. The CARB SLCP strategy sets a much more stringent GWP limit on refrigerants in a wider range of applications, and while it does not achieve the level of deep decarbonization promised by a fully Kigali-compliant policy, it comes the closest. When combined with SNAP-style rules and a refrigerant management program, F-gas emissions in Massachusetts can be reduced to just about 2.0 $MMTCO_2e$ by 2050.

### 3.6.2 Relationship with other sectors

F-gas emissions do not occur in a vacuum, and in some cases there are important interdependencies between F-gas end-use sectors and other important GHG emitting activities. These interdependencies have not been explicitly modeled, but the F-gas analysis presents important constraints that should be considered in the other sectors.

There is an important relationship between building decarbonization and F-gas emissions.[63] Ductless mini-split air conditioners and heat pumps are more energy efficient than either conventional central AC or window units. Heat pumps also are essential to electrifying residential heating, a step towards reducing natural gas and oil combustion in homes and replacing it with renewable electricity. However, as has been seen, heat pumps are important sources of F-gas emissions as they are charged with a large quantity of refrigerant and have the ability to leak at various stages in their lifetime. If heat pumps are a key component to building decarbonization strategy, it is important to have policies in place as soon as possible to ensure they utilize low-GWP refrigerants (HFOs or natural refrigerants), or else they will be a locked-in source of F-gas leakage for decades. This is especially important given all the uncertainties surrounding the performance of heat pumps at scale and future practices in heat pump service and retirement.

A second component of building energy efficiency is weatherization. Spray foam is a useful, effective way to retrofit existing buildings with improved insulation, but the blowing agent used in closed-cell foam is often an F-gas that leaks during foam blowing, as the foam decays over its lifetime, and at building demolition. As with heat pumps, the expectation that insulation will be a key measure in improving energy efficiency on a large

[63] UNEP & IEA, 2020.

scale means that rules requiring the use of low-GWP spray foam blowing agents must be implemented before weatherization programs lock future F-gas emissions in for the lifetime of the foam.

There also may be factors that push up residential and commercial building energy use that are related to F-gas emissions. As the planet warms, demand for refrigeration and air conditioning will likely increase faster than population. While the urgency for ensuring low-GWP refrigerants in these appliances is not as present as in heat pumps and foam, it remains true that any new appliance deployed with the current, high-GWP HFC refrigerant is locking in future F-gas emissions for the lifetime of the product.

Finally, there is a dependency between electrification and $SF_6$ emissions. There are two likely scenarios of expansion of the electrical transmission and distribution system: 1) expansion of high-voltage transmission to bring renewable electricity into Massachusetts from out-of-state; and 2) expansion of distribution networks to connect distributed renewable power generation. In both cases, infrastructure expansion involves more than just stringing wires. It involves the construction of new switchgear, substations, transformers, and other facilities that utilize $SF_6$ as a dielectric substance. The current $SF_6$ management rules in Massachusetts (310 CMR 7.72) set both a 1% leak rate and absolute limits on $SF_6$ emissions for the two largest utilities. With large expansion in the electric grid, the mass-based (absolute) limits on emissions will likely require utilities to achieve leak rates somewhat below 1%, which may require the adoption of non-$SF_6$ equipment.

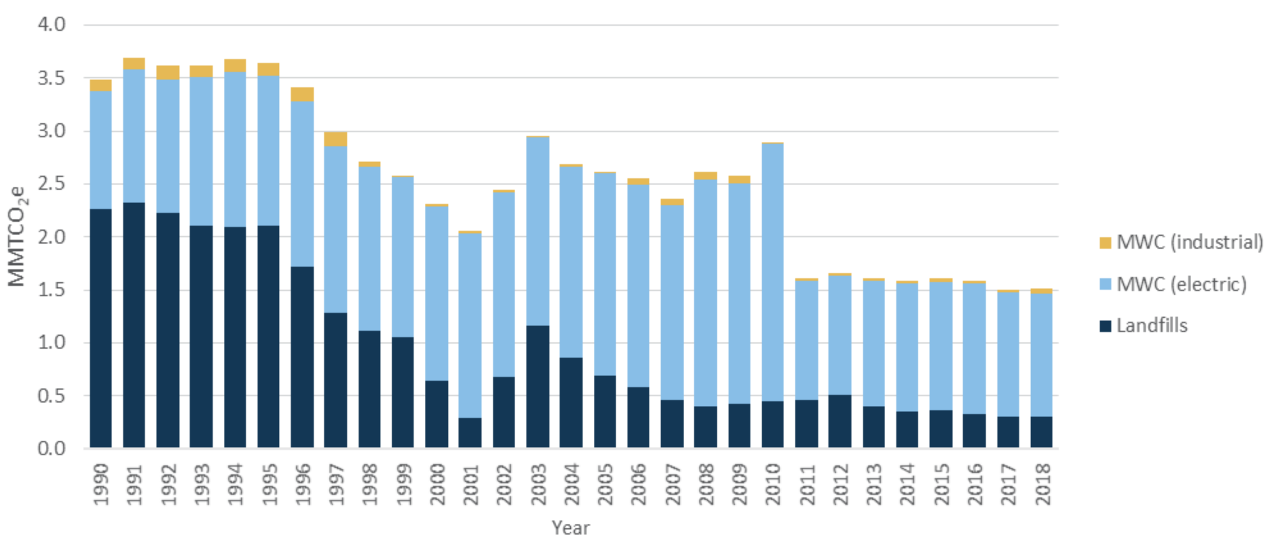
# 4 Solid Waste

## 4.1 Introduction

Emissions from solid waste management arise from 1) anaerobic decomposition of organic waste in landfills and organic waste facilities like compost and anerobic digesters and 2) the incineration of fossil fuel-derived materials (like plastics) in waste combustors. In 2018, emissions from the solid waste management sector comprised between 1-2% of total greenhouse gas (GHG) emissions in both Massachusetts and the entire US.[64,65]

The current inventory of GHG emissions from waste management in Massachusetts is shown in Figure 14. This inventory shows the effects of two important trends in the solid waste sector: the ongoing retirement of landfill capacity and the reliance on municipal waste combustors (MWCs) for municipal solid waste (MSW) management across the Commonwealth.

*Figure 14. Current inventory of solid waste sector GHG emissions from MassDEP, 1990-2018.[66]*




This chapter discusses the construction and analysis of emissions scenarios for the Commonwealth's solid waste sector from 2020-2050. In line with the Global Warming Solutions Act and the Massachusetts Department of Environmental Protection (MassDEP) GHG Inventory, the waste system includes the waste facilities physically located within the geographic boundaries of Massachusetts. There is an overarching modeling assumption that waste generated in-state is disposed of at in-state facilities up to the capacity of those facilities; any waste generated in excess of disposal capacity is exported. This assumption allows for the

[64] [U.S. EPA, 2020]. U.S. EPA, "Inventory of U.S. Greenhouse Gas Emissions and Sinks: 1990-2018," EPA 430-R-20-002, 2020. Available at https://www.epa.gov/ghgemissions/inventory-us-greenhouse-gas-emissions-and-sinks-1990-2018 (Retrieved 19 August 2020).
[65] [MassDEP, "MA GHG Inventory," 2020]. MassDEP, "Massachusetts Annual Greenhouse Gas Inventory: 1990-2017," 2020. Available at https://www.mass.gov/lists/massdep-emissions-inventories (Retrieved 19 August 2020).
[66] MassDEP, "MA GHG Inventory," 2020.

connection between policies that affect the composition (and tonnage) of waste generation and the emissions from waste facilities. This chapter is limited to emissions at waste facilities, as embodied and out-of-state emissions associated with the waste system are out of scope.

## 4.2 Methodology & Data

Modeling solid waste system emissions involves disaggregating the waste stream by destination and by composition, as materials in solid waste will be converted to GHGs differently depending on the disposal pathway. Three in-state disposal pathways are considered here:

- Landfill, where organic material is converted to methane ($CH_4$) following a first-order decay model, which is then either directly released to the atmosphere, partially oxidized by soil bacteria into biogenic carbon dioxide ($CO_2$), or captured and flared by landfill gas capture systems;
- MWCs, where carbonaceous wastes are burned and converted to $CO_2$ (emissions from combustion of paper, wood, food waste, and other organic materials are considered biogenic $CO_2$); and
- Organic waste recycling, where food waste and other organic material is converted to some mixture of $CO_2$, $CH_4$, and nitrous oxide ($N_2O$), depending on the type of facility.
    - Composting facilities operate aerobically, producing mainly biogenic $CO_2$ and some small but potentially important quantities of $CH_4$ and $N_2O$ (actual emissions factors depend largely on processing conditions)
    - Anaerobic digestion (AD) facilities convert organic waste in a closed, anaerobic vessel to biogas / digester gas, which has a much higher composition of $CH_4$, but which is then captured and either flared or burned for heat or power (releasing biogenic $CO_2$).

Forecasting emissions requires three modeling steps:

- First, estimating the tonnage, composition, and disposal pathways for MSW generated in Massachusetts each year, addressing changes in population and potential effects of policies on waste generation and disposal;
- Second, rationalizing the waste generation data in step one with the disposal capacity of Massachusetts waste facilities and the realities of inter-state waste trade, where some Massachusetts waste is disposed out-of-state (and therefore excluded from this model) and some out-of-state waste is imported and disposed in Massachusetts facilities; and
- Third, calculating the emissions from each type of facility based on composition and tonnage of waste handled.

Most of the data and assumptions on solid waste generation and disposal in Massachusetts used in this model come from two sources: the September 2019 draft of the "Massachusetts 2030 Solid Waste Master Plan"[67] (SWMP) and the February 2019 "Massachusetts Materials Management Capacity Study" prepared by MSW

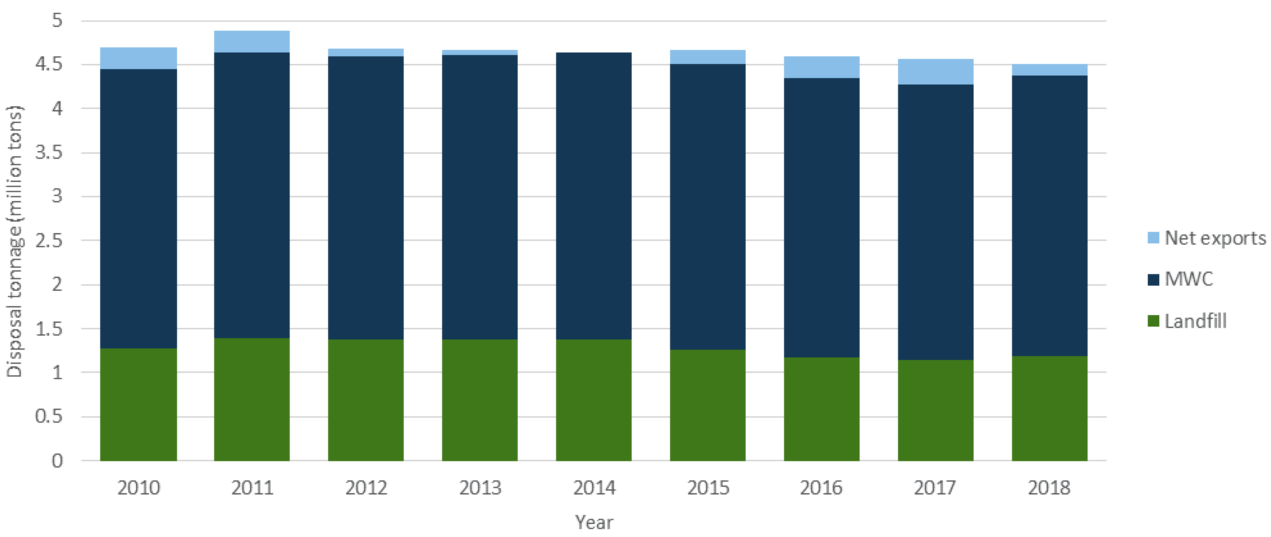
[67] [MassDEP, "Draft 2030 SWMP," 2019]. MassDEP, "Massachusetts 2030 Solid Waste Master Plan," Draft for public comment, 2019. Available at https://www.mass.gov/guides/solid-waste-master-plan (Retrieved 19 August 2020).

Consultants for MassDEP.[68] Other data sources are discussed below. This section reports waste quantities in short tons following the convention of these reports but emissions quantities are reported in metric units (million metric tons $CO_2$-equivalent, or $MMTCO_2e$).

### 4.2.1 Waste Generation & Disposal

The Draft 2030 SWMP publishes waste disposal figures for the years 2010-2018, including tonnage to landfill, tonnage to MWC, and net exports, as shown in Figure 15.[69] Over this time period, landfill disposal of MSW ranged between 1.14-1.39 million tons, MWC disposal ranged from 3.14-3.26 million tons, and net export ranged from 0-0.28 million tons (meaning that currently Massachusetts exports more waste than it imports). It should be noted that these figures give only partial information about waste generation, as they do not include any information about waste that is recycled or composted.

*Figure 15. Tons of MSW disposed in landfills and MWCs and MSW net export in Massachusetts, 2010-2018.*




Because Massachusetts is currently a net waste exporter, MassDEP calculates total MSW generated by Massachusetts residents bound for final disposal by adding landfill tonnage, MWC tonnage, and exports. In 2018, an estimated 6.85 million Massachusetts residents disposed a total of 4.51 million tons of MSW, or 0.66 tons/capita.

Composition of MSW is also included in the Draft 2030 SWMP, based on waste characterization data provided by MWCs. The characterization table is extremely detailed, but for the purposes of this model, MSW disposed in this time period is estimated to be 20.4% plastics, 63.6% organics (including paper, food waste, yard waste, and wood), and 16.0% inert materials (including metals, ceramics, and other inorganic materials).

[68] [MassDEP, "Capacity Study," 2019]. MassDEP, "Massachusetts Materials Management Capacity Study," prepared by MSW Consultants, 2019. Available at https://www.mass.gov/doc/massachusetts-materials-management-capacity-study-february-2019/download (Retrieved 19 August 2020)

[69] The data includes both MSW and non-MSW; only MSW is considered here.

Outside of the realm of the formal MSW management system detailed above, the 2018 Capacity Study estimated that approximately 400,000 tons of organic materials were composted[70] and an additional 92,000 tons were disposed in AD facilities. MassDEP estimated that in 2019, 41,000 tons of food waste was composted and 188,000 tons of food waste was sent to AD.[71] MassDEP also notes other destinations for this source-separated food waste: donation, processed into wastewater, animal feed, and on-site systems.

### 4.2.2 Disposal Capacity

As of 2019, Massachusetts had seven operational MWCs and nine MSW landfills, with total potential capacity of 3,518,225 tons and 842,245 tons, respectively (see Table 7). Six of the MWCs combust MSW and produce electricity; one (Covanta Pittsfield) produces steam for neighboring industrial facilities. According to MassDEP, landfills are operating at essentially maximum capacity while MWCs are operating at roughly 90% capacity.

*Table 7. MWC and landfill capacity (2019) and landfill expected closure date*

| MWC | Capacity (tons) | Landfill Name | Capacity (tons) | Closure |
|---|---|---|---|---|
| ***Haverhill*** | 602,250 | ***Bourne*** | 30,000[72] | 2024 |
| ***Pittsfield*** | 84,000 | ***Carver*** | 101,125 | 2020 |
| ***Rochester*** | 1,250,000 | ***Dartmouth*** | 115,000 | 2026 |
| ***Agawam*** | 131,400 | ***Middleborough*** | 60,000 | 2031 |
| ***Millbury*** | 529,575 | ***Nantucket*** | 26,000 | 2030 |
| ***North Andover*** | 460,500 | ***Taunton*** | 120,120 | 2020 |
| ***Saugus*** | 460,500 | ***Westminster*** | 390,000 | 2024 |
| **TOTAL** | 3,518,225 | **TOTAL** | 842,245 | |

Based on the anticipated closure dates shown in Table 7, the upcoming decade will see a significant reduction in in-state permitted landfill capacity. Figure 16 illustrates the anticipated reduction, showing most of the capacity gone by 2025, with the last landfill slated to close by 2032.[73]

According to the Capacity Study, many of the compost facilities surveyed were already at capacity, but there is the intention in the Draft 2030 SWMP to increase compost capacity in Massachusetts over the next decade. Similarly, the Capacity Study found AD capacity to process up to 275,000 tons per year in 2018, with an anticipated growth to 660,000 in the next few years, accounting for all expansion projects underway or awaiting permits at the time.

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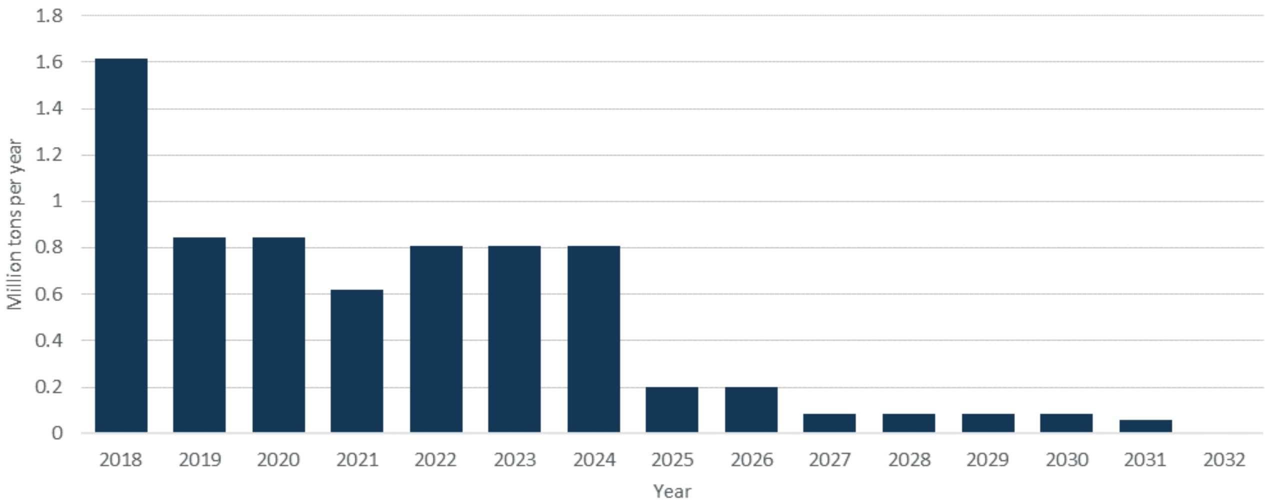
[70] This includes 129,925 tons received by large compost facilities (> 5,000 tpy), 248,827 ± 69,904 tons received by small facilities (< 5,000 tpy), and 21,219 ± 4,178 tons received by agricultural compost facilities. Composition of the organic material sent to compost is unknown.
[71] 2019 data provided directly from MassDEP.
[72] Capacity of the Town of Bourne Landfill is expected to increase to 219,000 tons per year in 2022.
[73] It is possible that some of these landfills may petition to extend their active life, but there is currently no expectation that landfills will remain open well after this time window.

*Figure 16. Anticipated permitted landfill capacity in Massachusetts, 2018-2032*[74]




### 4.2.3 Emissions factors

Calculating $CH_4$ emissions from landfills is more challenging than most other emissions calculations because organic matter deposited in a landfill will convert to methane over a period of years, rather than all at once. Accordingly, landfill emissions in 2020 are not only the result of waste disposed there in that year, but a function of waste disposed over the entire lifetime of the site. This also means that calculating contemporary emissions and forecasting potential future emissions requires the examination of now-closed landfills that may still be producing methane.

The standard method for calculating landfill emissions is a first-order decay function:[75]

$$A = \sum_{x=S}^{T-1} \left\{ W_x \times MCF \times DOC \times DOC_F \times F \times \frac{16}{12} \times \left( e^{-k(T-x-1)} - e^{-k(T-x)} \right) \right\}$$

where

$A =$ $CH_4$ generation (metric tons/year)
$x =$ Year in which waste was disposed
$S =$ Start year of inventory calculation
$T =$ Year for which emissions are calculated
$W_x =$ Quantity of waste disposed in landfill in year *x* (metric tons)
$MCF =$ Methane correction factor (assumed to be 1 for sanitary landfills)
$DOC =$ Degradable organic carbon fraction (function of the type of waste: 0.2 for bulk waste)
$DOC_F =$ Fraction of DOC decomposed (assumed to be 0.5)
$F =$ Fraction by volume of $CH_4$ in landfill gas (assumed to be 0.5)
$k =$ Decay rate constant ($yr^{-1}$) (function of type of waste: 0.057 for bulk waste)

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74 MassDEP, "Draft 2030 SWMP," 2019
75 [U.S. EPA, 2010]. U.S. EPA, "Greenhouse Gas Emissions Estimation Methodologies for Biogenic Emissions from Selected Source Categories: Solid Waste Disposal, Wastewater Treatment, Ethanol Fermentation," prepared by RTI International, 2010.

There are 22 landfills in Massachusetts, closed and active, that contribute to the GHG emissions inventory (see the list in Appendix B). Data on annual waste tonnage and composition for each of these landfills was drawn from the U.S. Environmental Protection Agency's Facility Level Information on GHGs Tool (EPA FLIGHT). The data reported by each facility also includes information about landfill gas collection and soil oxidation of methane, which the U.S. EPA and MassDEP use to construct the final landfill emissions inventory. Comparing the final emissions inventory with the results from the first-order decay calculation shows that across all MA landfills, landfill gas systems captured 86.8% of methane generated in 2018, allowing 13.2% to be released to the atmosphere. This factor is used to convert the results of the decay function for years 2019-2050 to forecasted methane emissions.

MWC emissions from 2010-2018 are also reported to EPA FLIGHT, following a complicated set of emissions accounting procedures that enable facilities to calculate fossil $CO_2$, $N_2O$, $CH_4$, and biogenic $CO_2$ generation. For this study, future emissions from MWCs are forecasted by scaling from the 2018 reported baseline according to tonnage and composition of waste disposed in MWCs. Fossil $CO_2$ scales by the amount of plastic disposed, $N_2O$ and $CH_4$ scale by total waste, and biogenic $CO_2$ scales by the amount of organic material disposed.

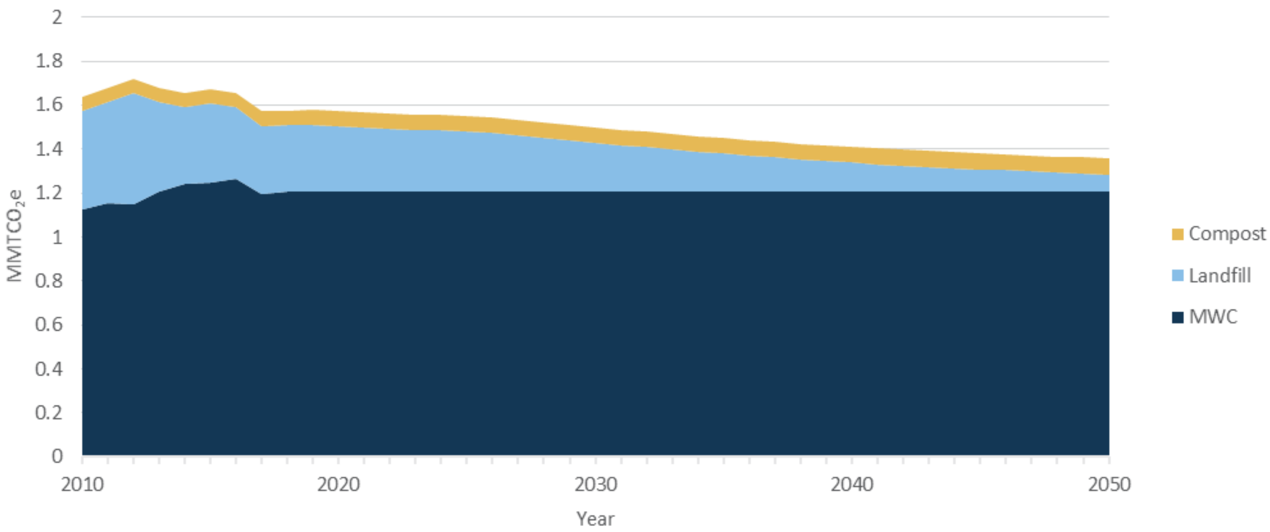
Calculating emissions from organic waste processors is comparatively simple: an emissions factor is multiplied by the quantity of organic waste disposed. There is a great deal of uncertainty as to the correct emissions factors for compost facilities. For this study, the baseline emissions factors from the IPCC are used: 4 g $CH_4$ / kg organic waste and 0.3 g $N_2O$ / kg organic waste. There is a similar lack of sufficient information about emissions from AD given that their design and maintenance can vary widely. Here, because of low prevalence of AD and the uncertainty in their emissions rates, it is assumed that there are no $CH_4$ emissions from these facilities. If AD becomes a significant treatment option in Massachusetts this assumption should be revisited in future work.

## 4.3 Reference Case

In the reference case for solid waste, changes to the waste system that are already in place will be expected to continue (e.g., closure of landfills during the next decade) but changes that would be the result of specific, new environmental policies (e.g., SWMP) are excluded. In this scenario, it is assumed that per capita waste generation and waste composition are stable from a 2018 baseline: 0.66 tons waste/capita and plastics/organics/inert fractions of 20.4%/63.6%/16.0%. MWC capacity is maintained at 3.19 million tons per year while landfill capacity is reduced according to the schedule presented in Figure 16. This scenario assumes no increase in organic waste diversion to compost beyond what would be expected from population growth, using the reported 400,000 tons of organic waste sent to compost in 2018 as a baseline. All waste generated in excess of in-state capacity is exported to neighboring states; the out-of-state waste will still be responsible for GHG emissions depending on how it is processed, but those emissions are not accounted for in this analysis and would be expected to be contained within the other state's GHG inventory. The reference emissions case for solid waste is presented in Figure 17.

In this scenario, in-state emissions are expected to decrease from 1.64 $MMTCO_2e$ in 2010 to 1.36 $MMTCO_2e$ in 2050, a reduction of 17% due entirely to the retirement of landfills in Massachusetts. Emissions from MWCs under this scenario remain stable and emissions from composting facilities increase very slightly. Because of the reduction in disposal capacity and increase in population, Massachusetts relies very heavily on waste exports, increasing from 250,000 tons in 2010 to 1.7 million tons in 2050.

*Figure 17. Reference case for solid waste emissions in Massachusetts, 2010-2050*




## 4.4 Uncertainty & Sensitivity Analysis

There are a number of factors that could be adjusted to examine the resulting effect on forecasted solid waste emissions. In this analysis, the sensitivities tested include changes to waste generation tonnage and composition, the landfill retirement schedule, the future of MWCs in Massachusetts, and the emissions factors associated with organics recycling. This sensitivity analysis is detailed and described in greater depth in the following sections.

### 4.4.1 MassDEP Draft 2030 Solid Waste Master Plan (SWMP)

The Draft 2030 SWMP[76] assumes a future very different from the one presented in the reference case. It sets an ambitious waste reduction goal of reaching roughly 30% below 2018 levels by 2030 and 90% below 2018 levels by 2050. On top of this goal, the plan prioritizes the diversion and/or source reduction of organic waste, setting a target of an additional 500,000 tons per year of avoided food waste disposal by 2030. The anticipated tonnage and composition of disposed waste (sent to landfills, MWCs, or export) from the Draft 2030 SWMP is shown in Figure 18. Overlaying this waste flow forecast onto the anticipated in-state disposal capacity reveals that landfills can continue to be utilized at capacity until they are retired.

Emissions from MWCs will depend on the amount and composition of in-state and imported waste. The reductions in Massachusetts-generated plastic waste being sent to disposal under the Draft 2030 SWMP may result in a reduction of fossil $CO_2$ emissions from MWCs, but at some point in the future, the decreasing calorific value of the waste being incinerated may require an increased consumption of fossil fuel to maintain sufficient combustion temperatures.[77] This emissions source is out of scope for the non-energy emissions

[76] MassDEP, "Draft 2030 SWMP," 2019.

[77] The primary job of an MWC is to dispose of waste. To do this, the waste must be burned at a hot-enough temperature both to destroy the waste and to avoid producing toxic pollutants. Achieving a high temperature in the MWC boilers

sector, but it is an uncertainty factor to consider. The expected composition for MSW in Massachusetts under the Draft 2030 SWMP, absent any change in regulation of in-state MWCs, is illustrated in Figure 19.

*Figure 18. MSW disposal composition (in million tons of waste) under the Massachusetts Draft 2030 SWMP, 2010-2050*

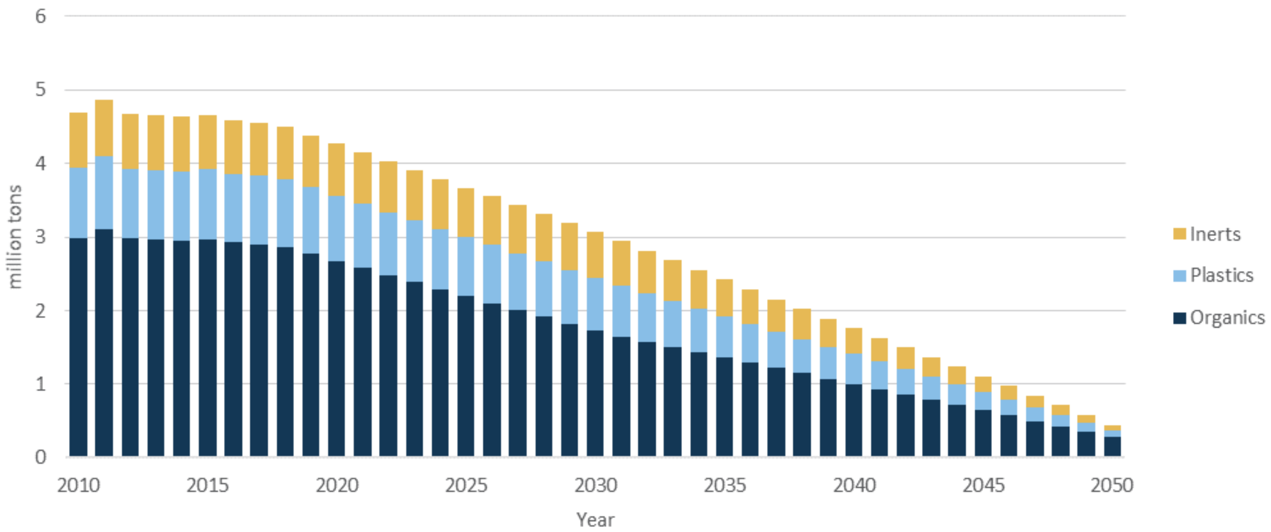
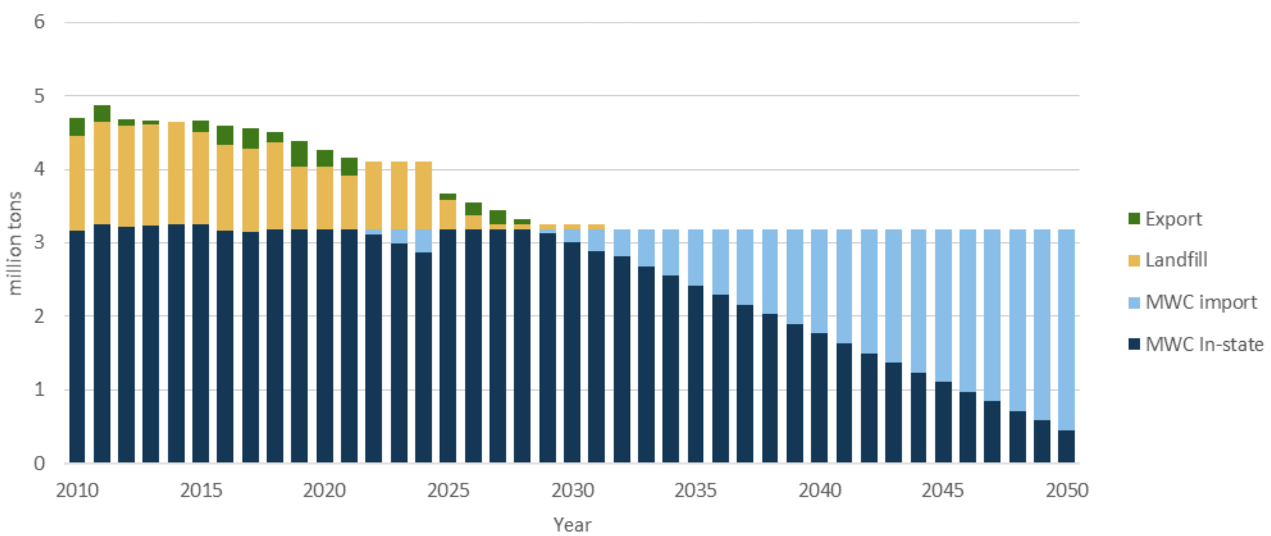



*Figure 19. MSW disposal pathway (in million tons of waste) and potential interstate trade under the Draft 2030 SWMP, 2010-2050*




The draft 2030 SWMP sets a goal of 500,000 tons of organic waste to be diverted by 2030. The draft 2030 SWMP does not specify the fraction of the organic waste diversion goal to be met through diversion (e.g., compost/AD) versus through source reduction. Assuming no source reduction, two scenarios are evaluated: a higher emissions scenario one in which all additional organics diversion is handled by compost and a lower

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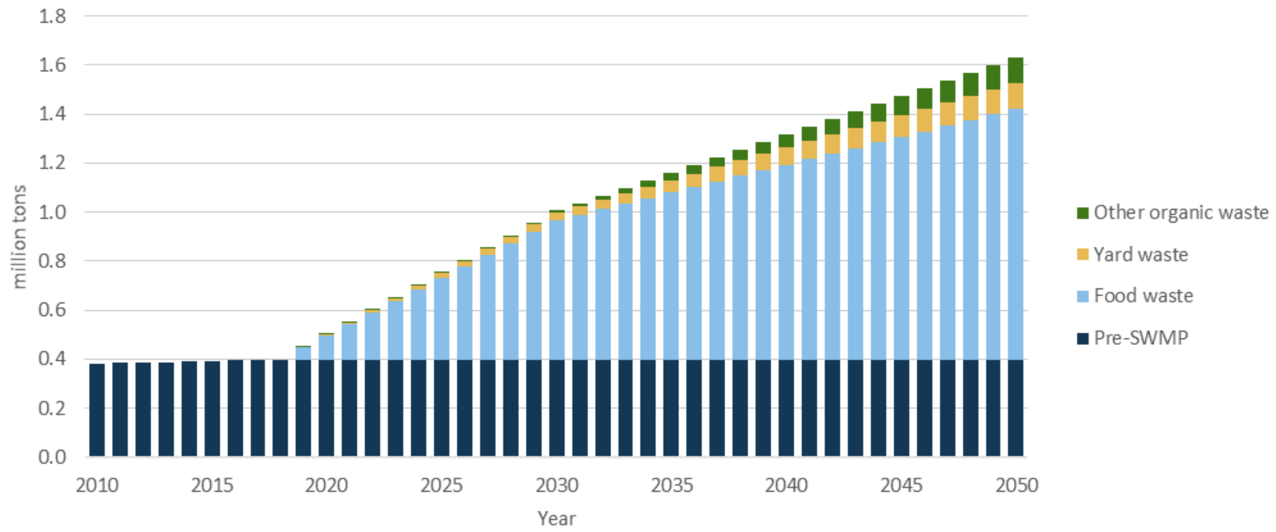
requires a high quality fuel, defined by the heating value, or calorific value, of the waste material. Some wastes, like paper and plastic, have high heating values and are useful to achieving the desired combustion temperature. Other wastes, like wet food waste and metals, have low heating values and pull down the temperature of the flame. In normal operation of MWCs, the operator may need to supplement with fuel to maintain hot-enough combustion temperatures. If plastics and paper are diverted from disposal faster than other materials, the average heating value of the waste stream decreases, potentially requiring an increased use of natural gas or oil to maintain proper combustion temperatures.

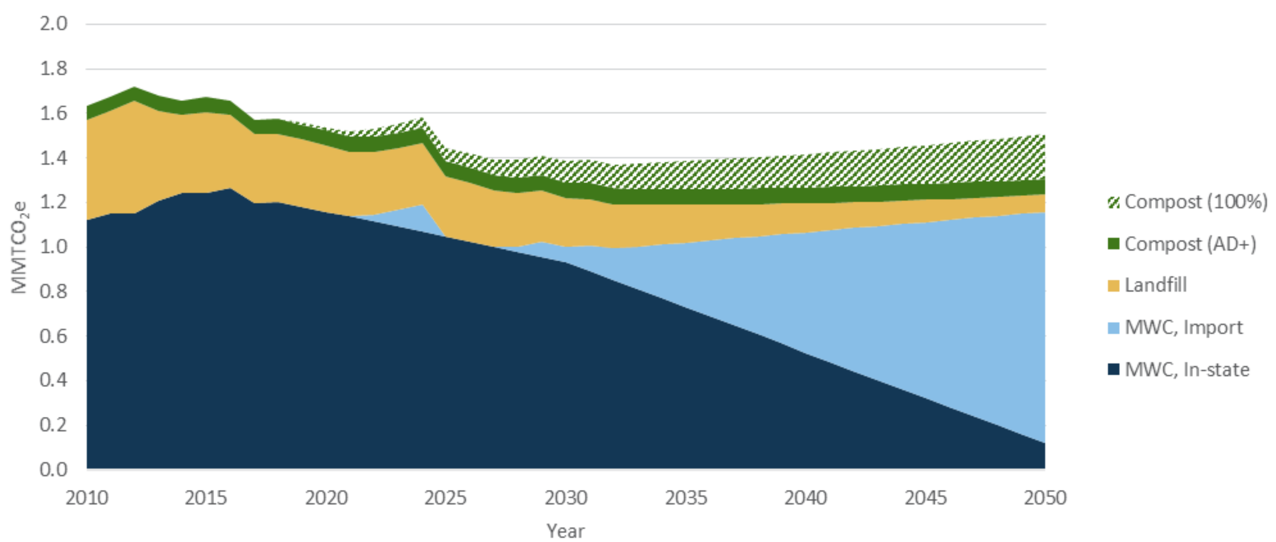
emissions scenario in which all additional food waste diversion is handled by AD with yard waste and other organic waste continuing to go to compost. The resulting waste flow is illustrated in Figure 20.

*Figure 20. Organic waste diversion (in million tons of waste) under the Draft 2030 SWMP, 2010-2050*




Bringing all these changes together results in a potential 2050 emissions pathway under the Draft 2030 SWMP shown in Figure 21. As discussed above, emissions from landfills are unaffected. MWCs observe a slight decrease in emissions as plastics are removed from the waste stream. If all organic waste diversion is assumed to be processed as compost (100% compost scenario—checkered wedge), emissions from compost also grow; under the AD scenario (where emissions stop at the solid green wedge), compost emissions stay essentially flat. Due to the reductions to waste generation under the Draft 2030 SWMP, there are no waste exports from Massachusetts anticipated after 2028, eliminating that source of emissions leakage. The small jump in imported waste in 2023-2024 is due to waste generation declining faster than in-state capacity. In 2025, 600,000 tons of landfill capacity is removed from the system, and all in-state capacity can be filled by in-state generation until 2029.

*Figure 21. GHG Emissions from the solid waste sector under the Draft 2030 SWMP, 2010-2050, two organic waste scenarios*



### 4.4.2 Landfill Gas Capture Sensitivity Analysis

The major source of uncertainty regarding landfills is whether landfill gas capture systems will continue operating at their current levels. Currently, the landfill gas capture systems operating in Massachusetts collect 86.8% of the methane generated in landfills, emitting just 13.2% to the atmosphere. Data from EPA FLIGHT suggests the leakage rate actually declined from about 20% in 2010 to 13.2% in 2018, yet there is not enough data to say definitively if this is a genuine trend or if it is an artifact of improved data collection and modeling on the parts of the facilities themselves and the U.S. EPA. If genuine, perhaps improvements are possible in the coming years. On the other hand, most of the landfills in the Commonwealth will reach the end of their post-closure care windows between now and 2050, and it is uncertain how these facilities will maintain their gas capture and flaring systems after that point. If the gas capture systems are not maintained, methane generated from organic waste decomposition may leak out at higher rates, even as the total amount of methane being generated will continue to decrease over time.

### 4.4.3 MWCs in the Commonwealth

The Massachusetts Draft 2030 Solid Waste Master Plan has a proposed goal to reduce disposal from 5.7 million tons in 2018 to 4.0 million tons by 2030. At that time, given anticipated landfill closures, MassDEP expects to continue to rely on existing MWC capacity to meet the Commonwealth's solid waste disposal needs. However, the Draft 2030 Solid Waste Master Plan has a longer term goal to reduce solid waste disposal to 570,000 tons by 2050. Contingent on this progress, Massachusetts would require lesser capacity to meet its solid waste management needs. At the same time, should MWCs seek to update or rebuild facilities, MassDEP would require tighter emissions standards and increased efficiency standards based on the latest technology. MassDEP will reassess progress toward this goal in a solid waste program review to be conducted in 2025.

### 4.4.4 Organics Recycling

Two pathways are available for future organic waste recycling: composting and AD. The modeling thus far has considered scenarios at the extremes of these pathways: 1) an assumption that most organic waste recycling occurs via composting and any growth in this waste stream will also be diverted to compost; and 2) an assumption that all future growth in food waste diversion is processed in AD. There is uncertainty in these pathways that stems from potential variation in:

- the balance in disposal pathway among compost (with a positive emissions factor), AD (with a negligible emissions factor and the possibility of biogas capture and use), and source reduction (with a zero emissions factor); and
- the composting system emissions factors themselves.

To the first point, any combination of organics diversion strategies that includes AD and source reduction would have a smaller GHG footprint than that calculated here. Although the real-world emissions from AD is likely nonzero, there is not enough evidence at scale to assign food waste digesters a reliable emissions factor. Further, there are potential emissions offsets available from the use of di-gas to replace the use of fossil fuels.

As for the emissions factors from composting, the IPCC-determined factors are based on a very high degree of generalization. The reality is made more complex by the specific nutrient composition of the waste (yard waste vs. food waste, for example), moisture levels, aeration, and other compost management characteristics. It was not within the scope of this study to develop more reliable emission factors for the Commonwealth's composting facilities, but it may be useful to examine the effect of these factors in future work.

Using the standard IPCC emissions factors (4 g $CH_4$ / kg organic waste and 0.3 g $N_2O$ / kg organic waste), the model estimates 0.07 $MMTCO_2e$ from composting in 2050 in the reference case and 0.27 $MMTCO_2e$ under the Draft 2030 SWMP scenario (assuming 100% organic waste diversion to compost). Because there is a linear relationship between compost emission factors and compost emissions, a range of 50% lower and higher emissions factors (2-6 g $CH_4$ / kg organic waste and 0.15-0.45 g $N_2O$ / kg organic waste) also produces a variation of 50% in compost emissions under both scenarios: 0.04-0.11 $MMTCO_2e$ (reference case) and 0.14-0.41 $MMTCO_2e$ (Draft 2030 SWMP). The effect on overall waste system emissions is more muted, however: 1.32-1.39 $MMTCO_2e$ (reference case) and 1.37-1.64 $MMTCO_2e$ (Draft 2030 SWMP).

## 4.5 Discussion

The modeling shows that in the reference case, emissions from the MSW system in the Commonwealth are expected to decrease 17% from 2010 to 2050 primarily due to reduced emissions from landfills. Under the mitigation scenario, defined by the aggressive waste diversion goals of the Draft 2030 Solid Waste Master Plan, 2050 emissions could actually be slightly higher than those under the reference case. There are a few reasons for this counterintuitive outcome.

1. The system boundaries of emissions accounting for the solid waste management sector are limited to strictly process (i.e., non-energy) emissions from solid waste management that occurs within the geographic boundaries of Massachusetts. Many of the environmental and climate benefits of waste reduction and diversion are realized over the life-cycle of waste materials.[78] With recycling, benefits from avoided disposal and additional impacts from secondary processing (such as $CH_4$ emissions from compost piles) are often dwarfed by the benefits from avoided production of new raw materials (depending on the material). These benefits are not observable given the definition of the system boundaries used for this study.
2. Emissions from landfills have inertia. Any changes made today to the quantity and composition of waste disposed in landfills or to the operation of landfills will take a long time to be observed because today's emissions are the result of decades of accumulated disposal.
3. The waste system transcends state (and to some degree international) boundaries. In 2018, Massachusetts exported over 750,000 short tons of MSW while importing 610,000 short tons of MSW from neighboring states.[79]
4. There is a lot of uncertainty around emissions from composting and other organic waste management systems. The modeling approach in this analysis adopted conservative assumptions so as to not be overly optimistic as to the climate benefits of compost. If operated correctly, composting can likely be much more climate friendly than the modeling currently suggests. Alternatively, an increased reliance on anerobic digestion would avoid some of the potential emissions from new composting capacity.

[78] A full accounting of waste system emissions would include embodied, life-cycle emissions of waste materials, so as to be able to evaluate the climate impacts of waste reduction and recycling. EPA WARM uses a simple life cycle model to be able to accomplish this task.
[79] MassDEP, "Draft 2030 SWMP," 2019.

# 5 Wastewater

## 5.1 Introduction

Wastewater systems are anthropogenic sources of all three major naturally-occurring greenhouse gases (GHGs)—carbon dioxide ($CO_2$), methane ($CH_4$), and nitrous oxide ($N_2O$)—but as $CO_2$ released during the decomposition of sewage and other organic matter in wastewater is biogenic, it is excluded from the GHG accounting for this sector.

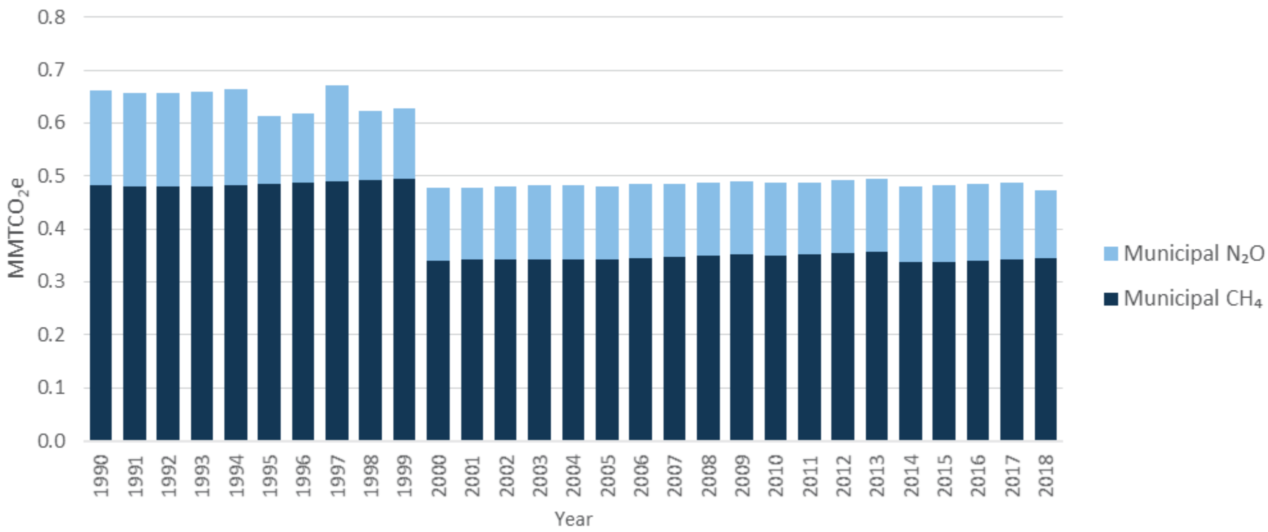
$CH_4$ is generated from the anaerobic decomposition of sewage and other organic matter in wastewater. Most wastewater treatment plants (WWTPs) are designed to operate aerobically, but methane can form in sewers before it reaches the plant and, in some process stages, in WWTPs where anaerobic bacteria can survive. WWTPs equipped with anaerobic digestion (AD) capacity are designed to capture the resulting methane as biogas (also known as digester gas or di-gas), preventing its release to the atmosphere. Standalone septic systems tend to operate much more anaerobically than their WWTP counterparts, which means that they generate more $CH_4$ per volume of wastewater than do WWTPs and do not have the capability to capture $CH_4$.

$N_2O$ is generated through a series of chemical reactions involving the conversion of nitrogen known as simultaneous nitrification-denitrification. Generally speaking, nitrification is the oxidation of ammonia and other nitrogen-bearing compounds in wastewater to nitrate ($NO_3^-$)—which can generate $N_2O$ as a byproduct; denitrification is the subsequent reduction to $N_2O$ and nitrogen gas. These conversions occur naturally throughout the wastewater system: in sewers, in WWTPs, and in the aquatic environments that receive treated effluent. In locations sensitive to nitrogen pollution and eutrophication, WWTPs may facilitate nitrogen removal—reducing nitrogen in the effluent—through tertiary treatment steps that accelerate the nitrification-denitrification reactions, generating more $N_2O$ emissions than facilities without nutrient management systems. Septic tanks have not been observed to generate $N_2O$ themselves, but research does show $N_2O$ generation in the soil dispersal fields that receive septic effluent.

In Massachusetts, wastewater treatment accounted for 0.47 million metric tons $CO_2$-equivalent (MMT$CO_2$e), or 0.6% of statewide emissions, in 2018, three-quarters of which was from $CH_4$ and one-quarter from $N_2O$, as illustrated in Figure 22.[80] Although this quantity is less than 1% of the total state-wide GHG emissions inventory, wastewater treatment is responsible for nearly 20% of overall $CH_4$ emissions and 16% of overall $N_2O$ emissions. According to the Massachusetts Department of Environmental Protection (MassDEP), emissions from wastewater treatment have declined 28% since 1990, but have held largely stable since 2000, when the Deer Island WWTP became fully operational. The step change in the time series in that year represents the avoided $CH_4$ emissions from the use of Deer Island's anaerobic digesters. The observed stability in emissions before and after 2000 speaks to the limited opportunities to reduce emissions from this sector. Large infrastructural changes like the introduction of AD capacity can partially mitigate emissions, but because wastewater emissions are closely tied to the generation of human sewage, which is tied to population, this is a difficult-to-reduce source of GHG emissions.

[80] [MassDEP, "MA GHG Inventory," 2020]. MassDEP, "Massachusetts Annual Greenhouse Gas Inventory: 1990-2017," 2020. Available at https://www.mass.gov/lists/massdep-emissions-inventories (Retrieved 19 August 2020).

*Figure 22. Current inventory of wastewater GHG emissions from MassDEP, 1990-2018.*




Although wastewater emissions arise from both domestic[81] and industrial facilities, the current Massachusetts emissions inventory accounts only for domestic sources, a system boundary followed in this study as well. Industrial facilities wastewater treatment includes effluent from breweries, food production, hospitals, laboratories, fabrication plants and other activities.

This chapter presents forecasts of GHG emissions from wastewater and wastewater management in Massachusetts through 2050. No clear technological pathways exist for deep decarbonization in this subsector. The modeling approach utilized here quantifies the potential effects, both positive and negative, of varying key drivers shown to influence GHG emissions from wastewater. This approach also helps to ameliorate the significant uncertainty that challenges emissions accounting in this sector.

## 5.2 Methodology & Data

The methodology used to forecast GHG emissions from wastewater treatment is adapted from the Intergovernmental Panel on Climate Change (IPCC) and U.S. Environmental Protection Agency (EPA) emissions inventory guidance.[82] This approach adopts a simplified view of the wastewater system in the Commonwealth, in which domestic wastewater is handled by one of three systems: on-site septic, conventional WWTP, and WWTP with AD. Accordingly, four models were used to forecast GHG emissions from wastewater treatment: $CH_4$ from septic systems, $CH_4$ from WWTPs, $N_2O$ from WWTPs, and $N_2O$ from effluent. Each of these is a relatively simple model based on either IPCC or U.S. EPA emissions inventory guidance.

[81] There is some inconsistency in the literature regarding wastewater that comes from people, as opposed to that which comes from industrial sources. Names for the former category include residential, domestic, and municipal wastewater.
[82] [IPCC, 2019]. Intergovernmental Panel on Climate Change, "Wastewater Treatment and Discharge," Ch. 6 in *2019 Refinement to the 2006 IPCC Guidelines for National Greenhouse Gas Inventories, Volume 5: Waste*, Switzerland: IPCC, 2019. Available at https://www.ipcc-nggip.iges.or.jp/public/2019rf/vol5.html (Retrieved 18 August 2020).

### 5.2.1 Septic $CH_4$

Anaerobic bacteria in septic tanks are responsible for the partial conversion of sewage to methane. Annual $CH_4$ emissions from septic systems are calculated via the equation below, adapted from the IPCC guidance:

$$CH_4^{septic} = P_{septic} \times \left(1 - \frac{F}{2}\right) \times BOD \times B_o \times MCF$$

where

$CH_4^{septic}$ = Emission of $CH_4$ from septic systems (kg $CH_4$/year)
$P_{septic}$ = Population using septic systems
$F$ = Fraction of the septic system users who manage their systems using best practice, which leaves half of the organics in wastewater in the tank at any given time
$BOD$ = Per capita generation of organic material in wastewater, represented in terms of biochemical oxygen demand (kg BOD/person/year)
$B_o$ = Maximum $CH_4$-production capacity for domestic wastewater (kg $CH_4$/kg BOD)
$MCF$ = Methane correction factor, which indicates the extent to which $B_o$ is realized in septic systems

There is much uncertainty surrounding these factors. According to MassDEP estimates, 28% of Massachusetts residents rely on septic systems. The Mass Sludge Survey 2018 mentions the origins of this figure,[83] which appears to be based on a historical finding that 27% of the population used septic systems in 1990.[84] The U.S. EPA, using Census Bureau data, estimates that the national average is closer to 18-19%.[85] This uncertainty makes this factor an important candidate for sensitivity analysis below.

There is no data as to the rate of septic system owners complying with best practices. Regulation 310 CMR 15.000 states that septic systems "shall be pumped whenever necessary to ensure proper functioning of the system".[86] This equates to pumping generally every 1-3 years. IPCC suggests the use of a factor of 0.5 in jurisdictions lacking regulations for septic system operation. Here, 0.75 is used as a stand-in factor, and is tested in the sensitivity analysis.

The remaining factors are drawn from IPCC guidance.[87] Biochemical oxygen demand (BOD) for sewage in the United States is estimated to be 85 g/person/day = 31 kg/person/year, but potentially ranging between 50-120 g/person/day = 18-44 kg/person/year.[88] The IPCC estimates the maximum $CH_4$ production capacity of domestic

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[83] [NEBRA & MassCEC, 2019]. NEBRA & MassCEC, "The Mass Sludge Survey 2018, v 1.1: Wastewater Solids Generation and Management in Massachusetts," 2019.
[84] [Medalie, 1995]. Medalie, L., "Wastewater collection and return flow in New England, 1990," Water Resources Investigation Report 95-4144, USGS, 1995. Available at https://pubs.usgs.gov/wri/1995/4144/report.pdf (Retrieved 10 December 2020).
[85] [U.S. EPA, 2020]. U.S. EPA, "Inventory of U.S. Greenhouse Gas Emissions and Sinks: 1990-2018," EPA 430-R-20-002, 2020. Available at https://www.epa.gov/ghgemissions/inventory-us-greenhouse-gas-emissions-and-sinks-1990-2018 (Retrieved 19 August 2020).
[86] [MassDEP, "310 CMR 15.000," 2020]. MassDEP, "310 CMR 15.000: Septic Systems ("Title 5")," 2020. Available at https://www.mass.gov/regulations/310-CMR-15000-septic-systems-title-5 (Retrieved 18 August 2020).
[87] IPCC, 2019.
[88] [Metcalf & Eddy, 2003]. Metcalf & Eddy, Inc, *Wastewater Engineering: Treatment, Disposal, Reuse*, McGraw-Hill: New York, 2003.

sewage ($B_o$) to be 0.6 kg $CH_4$/kg BOD. Finally, the methane correction factor for septic tanks is assumed to be 0.5, with a range between 0.4-0.72. All factors with ranges are evaluated in the sensitivity analysis.

An alternative method is used by the U.S. EPA, which cites a 2010 report from the Water Environment Research Foundation that calculated a septic system emission factor in the U.S. to be 10.7 g $CH_4$/capita/day.[89] Results from the use of this factor are compared to the results from the more detailed model below.

### 5.2.2 WWTP $CH_4$

Methane production at WWTPs is estimated using a similar model to that for septic emissions:

$$CH_4^{WWTP} = P_{WWTP,aerobic} \times (1 - S_{WWTP}) \times BOD \times B_o \times MCF$$

where

$CH_4^{WWTP} =$ Emission of $CH_4$ from WWTPs (kg $CH_4$/year)
$P_{WWTP,aerobic} =$ Population using WWTPs without AD
$S_{WWTP} =$ % of BOD removed during primary treatment
$BOD =$ Per capita generation of organic material in wastewater, represented in terms of biochemical oxygen demand (kg BOD/person/year)
$B_o =$ Maximum $CH_4$-production capacity for domestic wastewater (kg $CH_4$/kg BOD)
$MCF =$ Methane correction factor, which indicates the extent to which $B_o$ is realized in WWTPs

The population that uses conventional, aerobic WWTPs without AD is estimated by subtracting from the total population the population using WWTPs with AD and the population relying on septic systems (28%, see above). Five WWTPs in Massachusetts have anaerobic digesters:

- Deer Island WWTP, which treats sewage from 43 communities in the greater Boston area (33% of total Massachusetts population in 2010);
- The Greater Lawrence Sanitary District, which treats sewage from Andover, Dracut, Lawrence, Methuen, and North Andover (and Salem, NH, which is excluded from the analysis);
- Pittsfield WWTP, serving Dalton, Hinsdale, Pittsfield, and parts of Lenox;
- Clinton WWTP, serving Clinton and Lancaster; and
- Rockland WWTP, serving Rockland.

Population forecasts for these communities were provided by the Massachusetts Executive Office of Energy and Environmental Affairs (EEA). Assuming 28% of the population uses septic (per MassDEP estimates), the fraction of Massachusetts residents using conventional WWTPs is found to be 34% in 2010 (2.20 million people), declining to 30% by 2050 (2.36 million people), under the baseline population growth scenario.

The U.S. EPA estimates that 32.5% of BOD is removed during primary treatment. BOD is the same as in septic systems: 31 kg/person/year (ranging between 18-44 kg/person/year). $B_o$ is also the same: 0.6 kg $CH_4$/kg BOD.

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[89] [Leverenz, Tchobanoglous & Darby, 2010]. Leverenz, H.L., G. Tchobanoglous & J.L. Darby. 2010. “Evaluation of Greenhouse Gas Emissions from Septic Systems.” DEC1R09. Alexandria, VA & London, UK: Water Environment Research Foundation (WERF) & IWA Publishing.

As WWTPs are designed to operate highly aerobically, the value of the methane correction factor (MCF) is much lower than that for septic systems: 0.03, ranging between 0.003-0.09.

Similar to septic systems, there remain significant uncertainties about the generation of $CH_4$ at WWTPs. Here, it is assumed that WWTPs with AD do not release any methane to the atmosphere. However, research shows that sewage can undergo anaerobic decomposition in sewers before it reaches the WWTP. It is also likely that AD facilities generate some emissions though leakage. Thus, facilities with AD would have some $CH_4$ footprint, but this is lower than those that do not have AD. Nevertheless, the general assumptions made here are largely in line with those made by MassDEP and the U.S. EPA, enabling comparisons between these results and the published inventories.

### 5.2.3 WWTP $N_2O$

Emissions of $N_2O$ from wastewater treatment and from effluent are also highly uncertain. The approach used here is based on the approach from the U.S. EPA.

For $N_2O$ emissions from WWTPs, a simple emissions intensity relationship is used:

$$N_2O^{WWTP} = P_{WWTP} \times EF_{N_2O}$$

where

$N_2O^{WWTP} =$ Emission of $N_2O$ from WWTPs (kg $N_2O$/year)
$P_{WWTP} =$ Population using WWTPs
$EF_{N_2O} =$ Per capita emission of $N_2O$ from WWTPs

Massachusetts population using WWTPs is found by subtracting from total population the population on septic (assumed to be 28%, per MassDEP). According to the U.S. EPA, WWTPs without nitrification/denitrification are assumed to generate 3.2 g $N_2O$/capita/year; those with nitrification/denitrification are assumed to generate 7 g $N_2O$/capita/year. The EPA State Greenhouse Gas Inventory Tool (SIT) uses a national average of 4 g $N_2O$/capita/year.

According to the Mass Sludge Survey, roughly half of surveyed WWTPs have some sort of nitrification/denitrification systems. It is unclear how this fraction translates into population served. Notably, the largest WWTP by far in Massachusetts—Deer Island WWTP—does not have nitrification/denitrification process steps.

### 5.2.4 Effluent $N_2O$

The other source of $N_2O$ is from effluent and biosolids management. Again, there is substantial uncertainty in the physical and biological processes that convert nitrogen in sewage solids and effluents into $N_2O$. The SIT provides a reasonable methodology based on IPCC guidance that can be forecasted into the future:

$$Biosolids = [P \times Protein \times F_{NPR} \times F_{NonCon}] - N^{WWTP}$$
$$N^{WWTP} = N_2O^{WWTP} \times {}^{28}/_{44}$$
$$N_2O^{Biosolids} = Biosolids \times (1 - F_{fertilizer}) \times EF_{biosolids} \times \frac{44}{28}$$

where

$Biosolids =$ Available N in biosolids (kg N)
$P =$ Population of Massachusetts
$Protein =$ Protein consumption (kg protein/person/year)
$F_{NPR} =$ Fraction of N in protein (kg N/kg protein)
$F_{NonCon} =$ Factor for nitrogen in non-consumed protein disposed in sewer system (kg N/kg N)
$N^{WWTP} =$ Direct N emissions from domestic wastewater (kg N/year)
$N_2O^{Biosolids} =$ $N_2O$ emissions from wastewater effluent and biosolids (kg $N_2O$/year)
$F_{fertilizer} =$ Percentage of biosolids used as fertilizer
$EF_{biosolids} =$ Emission factor for $N_2O$ from effluent (kg $N_2O$-N/kg sewage-N produced)
$44/28 =$ stoichiometric relationship between $N_2O$ and N (kg $N_2O$/kg N)

Data for these terms are drawn from the SIT, the IPCC guidance, and the Sludge Survey.

The U.S. EPA calculates that the average American consumes 44.7 kg protein/year. This factor is derived from U.S. Department of Agriculture (USDA) data. Protein is composed of approximately 16% N. For non-consumed protein, the SIT uses a default factor of 1.75. This differs from the default IPCC value of 1.13 for North America. The IPCC also includes a new factor $N_{HH}$, which incorporates N from household chemicals flushed down the drain into the model. It is unclear if the SIT factor already includes this or not.

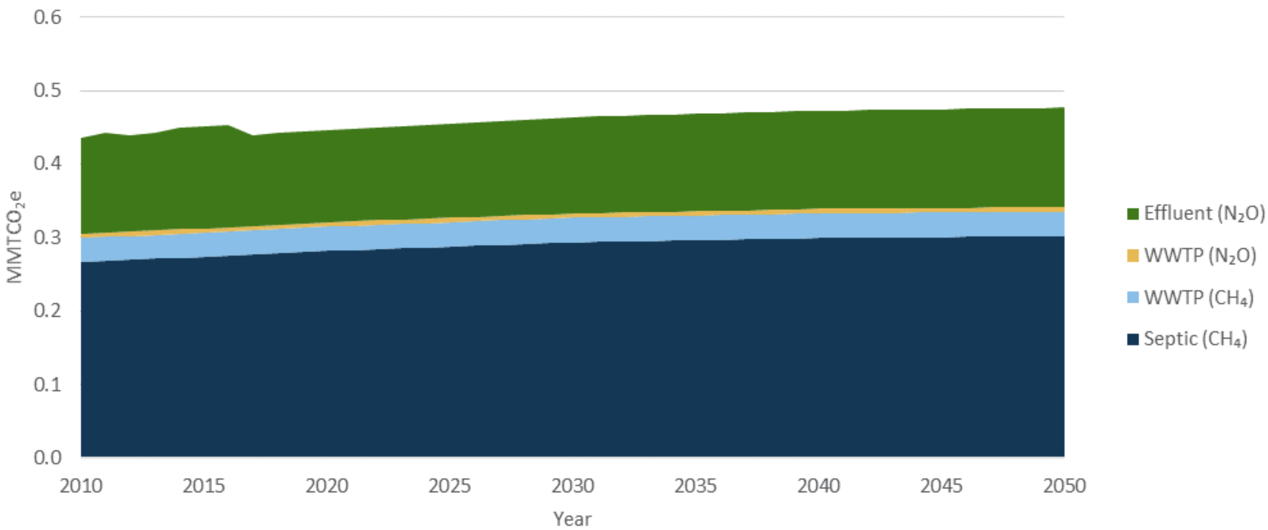
The model of $N_2O$ emissions from WWTPs (see above) is used to calculate direct N emissions from domestic wastewater. The Mass Sludge Survey reports that 38% of biosolids are land-applied. Finally, the default IPCC emissions factor for this activity is 0.005 kg $N_2O$-N/kg sewage-N produced. It is important to note that the use of biosolids as fertilizer does not prevent the release of $N_2O$ emissions; rather, the $N_2O$ emissions associated with biosolids that are land-applied as fertilizer are accounted for in the agriculture subsector rather than the wastewater subsector.

## 5.3 Reference Case

The reference case assumes baseline population growth and holds all other parameters constant from 2018, including the fraction of population using septic systems, the technologies employed by the various WWTPs, diet and food waste habits of Commonwealth residents, land application of sewage sludge, and environmental conditions. In addition, where multiple factors or ranges of factors are given above, the reference case uses the default or median value. The resulting reference case is presented in Figure 23.

The reference case shows emissions from wastewater treatment increasing gradually from 0.44 MMT$CO_2$e in 2010 to 0.48 MMT$CO_2$e in 2050, the result of population growth over that time. Septic system emissions are shown to be the largest fraction—two thirds of the total emissions are methane from septic tanks. Nitrous oxide emissions from effluent and biosolids are also a large fraction, with $CH_4$ and $N_2O$ from WWTPs contributing a relatively small amount to the total.

*Figure 23. Reference case for GHG emissions from wastewater in Massachusetts, 2010-2050*




## 5.4 Uncertainty & Sensitivity Analysis

As discussed above, nearly every equation and factor for this sector has significant uncertainty associated with it. The wastewater system is more complicated than represented by the models used here, as are the biochemical processes that convert sewage to GHGs. Each parameter in the model has multiple plausible values, ranges of which were provided in the previous sections. Here, key parameters are varied one-by-one to observe the overall effect on subsector emissions.

### 5.4.1 Population

Under the high-growth population scenario, population state-wide is expected to be 7% higher in 2050 than it would be under the baseline population scenario. This growth is not distributed evenly across all jurisdictions in the Commonwealth. Towns and cities that send their sewage to Deer Island are 9% larger under the high-growth scenario than the baseline scenario by 2050, whereas a town like Clinton (which also has a WWTP with AD) has only a 3% larger population. The effect of different population growth scenarios on overall emissions from wastewater treatment is minor: an additional 0.03 $MMTCO_2e$ in 2050 is due to growth in population on septic. This is an unlikely outcome, however, as population growth is likely to also bring new sewer systems rather than significantly more people using stand-alone septic tanks.

### 5.4.2 Emissions factors

Some of the emission drivers and factors used in the model are published with reasonable ranges; the median values are used in defining the reference case. Based on the model above, the emissions factor for $CH_4$ from septic systems can range from 12.8-23.0 g $CH_4$/capita/day (the median value of 15.9 g $CH_4$/capita/day was used in the reference case). Also, as mentioned, the U.S. EPA uses an emissions factor of 10.7 g $CH_4$/capita/day. This wide range of factors suggests that emissions from septic systems might be between 33% below and 44% above the reference case; the importance of septic system emissions to the overall sector inventory means that this septic emissions variation translates into a variation between 20% below and 27% above the reference case for the whole sector.

The emission factor for WWTP $CH_4$ emissions also has a very wide range of possible values: 0.16-4.86 g $CH_4$/capita/day—more than an order of magnitude. The small contribution of WWTP methane to the overall sector inventory dampens the effect of this uncertainty, however, to just 7% below to 15% above the reference case.

Ultimately, these variations in emissions factors affect the entire emissions forecast uniformly, so they do not give any additional insight into decarbonization strategy.

### 5.4.3 System improvements

Unlike some other subsectors, there are no well-established, widely-accepted methods for reducing GHG emissions from wastewater. However, the models provide direction for how to mitigate some emissions as well as insight into other trends may make emissions reduction more difficult.

#### 5.4.3.1 Septic systems

The model shows that septic systems are the largest contributors to sector emissions. The reference case assumes a large fraction (28%) of Massachusetts population uses stand-alone septic tanks rather than being connected to sewers. If this value is incorrect and it were closer to the national average of 19%, the overall sector emissions would be 0.36 $MMTCO_2e$ in 2010, growing to 0.39 $MMTCO_2e$. Whatever the current numbers, it is clear that connecting more of the population to septic systems could help to reduce wastewater sector emissions. For example, halving the percentage of the population on septic between 2010 (28%) and 2050 (14%) would result in a reduction of 0.09 $MMTCO_2e$ per year—a 22% improvement—over that time period.

Another source of uncertainty in septic systems, and potential area of improvement, is in their operation. The model assumes 75% of the septic systems are managed correctly, being pumped every 1-3 years. If this value were incorrect, and instead is closer to the IPCC default of 50%, overall sector emissions would be 12-13% higher than the reference case. On the other hand, if 100% of septic systems are managed correctly, emissions can be 12-13% lower than the reference case.

#### 5.4.3.2 WWTPs

The models include two technological levers for affecting emissions from WWTPs: increasing the population served by WWTPs with anaerobic digesters and increasing the deployment of nitrification/denitrification. Note that increasing AD reduces $CH_4$ emissions from WWTPs while increasing nitrification/denitrification to reduce nitrogen in effluent has the undesirable side-effect of increasing $N_2O$ emissions from WWTPs. There is no information available about trends, so extreme cases are examined here.

If all WWTPs in Massachusetts integrate AD into their process flow (while leaving the fraction on septic alone), $CH_4$ emissions from WWTPs are effectively eliminated, decreasing emissions from the wastewater sector by about 7%.[90]

If all WWTPs employed nitrification/denitrification, emissions of $N_2O$ from WWTPs would increase from 4 to 7 g $N_2O$/capita/year, following U.S. EPA assumptions. This large change to the emissions factor has a negligible effect on overall emissions, however, with an increase of less than 1%. In reality, deployment of tertiary treatment may not be necessary for all locations. For example, Deer Island releases its effluent nine miles offshore at 100 feet of depth where the impact of nutrient loading is minimal.

#### 5.4.3.3 Diet & food waste

The last factor to be explored that affects GHG emissions from wastewater is diet. $N_2O$ emission from biosolids and effluent is a function of the amount of protein eaten (and excreted) and food waste directly flushed to sewage. These two factors are examined in turn.

If Massachusetts residents reduced the amount of protein they consume, for example by adopting more of a plant-based diet, it would reduce the amount of nitrogen going into wastewater, and therefore reduce the potential for $N_2O$ emissions. The U.S. EPA derived a value of 44.7 kg protein/capita/year. If this were reduced by 5%, total wastewater emissions would decline by roughly 1%. Similarly, the EPA suggests that a significant amount of nitrogen is directly flushed to the sewer through food waste and household chemicals. If the value of the factor for non-consumed protein (including from household products) in the $N_2O$ model above were reduced from its default value of 1.75 to a value closer to that of Europe (1.18),[91] annual emissions could be reduced by nearly 10%.

The use of in-sink garbage disposal units complicates efforts to reduce nitrogen loading of wastewater. In cities with anaerobic digestion capacity, in-sink food waste disposal is sometimes advocated as an alternative to composting.[92] The relative effects that this practice has on increased $N_2O$ generation and decreased $CH_4$ generation (from avoided landfill and compost disposal of food waste) have not been calculated.

## 5.5 Discussion

GHG emissions from wastewater are relatively minor, but no clear policies or technologies exist to achieve deep decarbonization in this sector. The baseline data and models are extremely uncertain, but the generally accepted emissions models utilized here suggest 0.4-0.5 $MMTCO_2e$ annual GHG emissions from septic systems, WWTPs, and biosolids. Model analysis suggests a few changes to the wastewater system that could lead to overall reduction in emissions, although uncertainty in the models and model parameters prevents clear quantification of the reduction potential:

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[90] Anaerobic digesters do emit some quantity of $CH_4$, but it is not quantified in any of the models used here.
[91] IPCC, 2019, p. 6.41.
[92] For example, see the City of Boston Zero Waste Toolkit for Boston residents. Available at https://www.boston.gov/sites/default/files/file/document_files/2019/08/resident_zero_waste_toolkit.pdf (Retrieved 10 December 2020). The benefits from co-digestion of food scraps with sewage sludge must be balanced against the potential negative impacts of fats and greases accumulating in the sewer system.

- Bringing more people onto sewers and off stand-alone septic systems would likely reduce $CH_4$ emissions from septic tanks, as would encouraging (or requiring) septic system owners to follow best practices.
- Expanding the use of anaerobic digesters at WWTPs would avoid many of the $CH_4$ emissions from WWTPs and have the compound advantage of converting sewage sludge into usable fuel.
- Reducing per capita protein consumption may reduce $N_2O$ emissions, as would avoiding flushing food waste directly to sewers. Interestingly, this recommendation goes contrary to some suggestions that residents whose WWTPs have AD capacity use in-sink disposals for their food waste to increase the carbon content—and therefore the digester gas generation potential—of the sewage. The models used here are insufficient to reliably compare the conflicting effects of this practice.

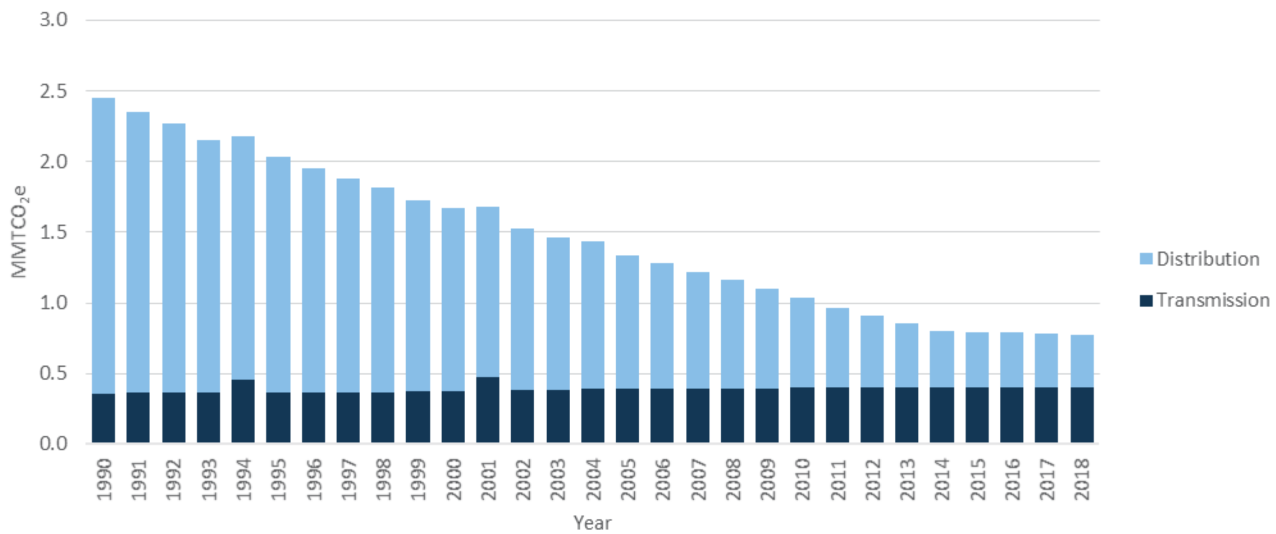
In a climate-constrained future, nutrient pollution from WWTP could have more detrimental effects on the inland and coastal waterways that receive WWTP effluent. It is possible that, in that future, more WWTPs would need to employ nitrification/denitrification processes to avoid eutrophication. The model results suggest that, although this process step does result in an increased $N_2O$ emission, it remains exceedingly minor and not a significant concern.

# 6 Natural Gas Leaks

## 6.1 Introduction

Massachusetts has some of the oldest natural gas distribution infrastructure in the country. Old pipes, joints, compressors, and meters leak methane ($CH_4$), resulting in gas leaks that are potentially hazardous to human health, pose risks of fire and explosion, and contribute to global climate change. According to the Massachusetts Department of Environmental Protection (MassDEP), the natural gas system in the Commonwealth emitted 0.77 million metric tons $CO_2$-equivalent ($MMTCO_2e$) of $CH_4$ from leaks and other fugitive emissions in the gas transmission and distribution in 2017, [93] a bit over 1% of the total state-wide emissions inventory. Looking at the trends within this sector in Figure 24, emissions from gas distribution have declined dramatically—by a factor of five—since 1990, while emissions from transmission have held largely stable.[94]

*Figure 24. Current inventory of natural gas leaks from MassDEP, 1990-2018[95].*




Independent research into natural gas leaks in Massachusetts suggest that the MassDEP inventory may be an underestimate, potentially significantly so. A study of atmospheric methane concentrations in the greater Boston area in 2012-2013 suggests that leaks may be as much as three times larger than the values in the

[93] Nationwide, the natural gas system also includes gas exploration, production, processing, and storage, all of which are also responsible for gas leaks.
[94] The decline in distribution emissions is due to long term trends away from leak prone pipe due in part to the Gas System Enhancement Plans program requiring gas companies to replace leak-prone natural gas infrastructure since the mid-2010s. Also, emissions per mile of pipeline have declined from the early 1990s to the mid-2010s, as reflected in the data sources MassDEP uses to calculate emissions. See the following section for more information.
[95] MassDEP is currently revising its methodology for the estimation of emissions from natural gas transmission, compressor stations, and LNG facilities, as described in Section 6.2.2. The new methodology results in an estimate that is 54% lower for 1990 emissions and 84% lower for 2018 emissions. The analysis conducted here uses the new methodology, but the existing inventory, presented in Figure 24, does not, at time of publication.

inventory. If true, methane emissions from the natural gas system in Massachusetts could be as high as 2 $MMTCO_2e$ in 2018, substantially higher than shown in Figure 24, though still a minor source in the overall state-wide inventory.[96] The authors of that study identify a few possible reasons for the discrepancy, including unaccounted-for emissions sources such as post-meter leaks and inaccurate emissions factors, but as yet there has not been work either to rationalize the MassDEP inventory with this bottom-up study or to replicate the research in more recent years. The project team chose to use the MassDEP inventory and top-down accounting method for the decarbonization study to ensure comparability with the current GHG Inventory methodology, acknowledging this potentially significant underestimate.

In this chapter, emissions from the Commonwealth's natural gas transmission and distribution system are forecasted to 2050. In all practical future scenarios that include gas transmission and distribution systems, the leakage and fugitive emission of natural gas is to be expected. Anything distributed in these systems, be it fracked, synthetic, or renewable natural gas—which are all substantially composed of methane, a potent GHG—therefore pose a climate risk. Current policies are encouraging the reduction of existing leaks,[97,98] but the future of gas leaks will depend largely on the future of natural gas in the Commonwealth. Accordingly, the reference case developed in this study assumes no changes to natural gas demand in Massachusetts beyond population growth and those changes already being observed in data about gas infrastructure.

The reference case is assumed to represent an upper bound on GHG emissions from this subsector. As part of this same Massachusetts Decarbonization Study, scenarios are being investigated that would shrink the gas distribution and transmission systems in Massachusetts. Specifically, widespread residential electrification would reduce the need for gas distribution, with associated reductions in gas leaks.

## 6.2 Methodology & Data

MassDEP has developed a detailed model of emissions from natural gas distribution systems based on a dynamic inventory of distribution mains, service lines, and meters. This approach is replicated below. Emissions from gas transmission are forecasted following a method developed by MassDEP, presented in Section 6.2.2.

### 6.2.1 Natural gas distribution

Natural gas distribution systems are networks of pipelines made of a variety of different materials, connected to residences, commercial buildings, and industrial facilities by service lines. At each consumer location, gas passes through a meter, and there are larger metering and regulation (M&R) stations located at key points in the network. Gas leaks occur in all parts of the system. Pipelines and service lines leak due to rust, cracking from temperature changes, and accidents like dig-ins during construction and intentional venting for cleaning

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[96] [McKain, et al., 2015]. McKain, K., et al., "Methane emissions from natural gas infrastructure and use in the urban region of Boston, Massachusetts," *PNAS,* 112(7), 1941-1946, 2015. https://doi.org/10.1073/pnas.1416261112.
[97] [DPU, 2020]. Massachusetts Department of Public Utilities, "GSEPs Pursuant to 2014 Gas Leaks Act," 2020. Available at https://www.mass.gov/lists/gseps-pursuant-to-2014-gas-leaks-act. (Retrieved 18 August 2020).
[98] [MassDEP, "310 CMR 7.73," 2020]. MassDEP, "Reducing Methane ($CH_4$) Emissions from Natural Gas Distribution Mains & Services (310 CMR 7.73)," 2020. Available at https://www.mass.gov/service-details/reducing-methane-ch4-emissions-from-natural-gas-distribution-mains-services-310-cmr (Retrieved 18 August 2020).

and safety purposes. Meters leak gas from their pressure regulators, their connection pipe joints, and other components depending on their age, physical integrity, design, and construction.

The MassDEP gas distribution emissions model follows:

$$A_x = \sum_i P_{i,x} \times EF^P_{i,x} + \sum_i S_{i,x} \times EF^S_{i,x} + \sum_j M_{j,x} \times EF^M_{j,x} + \sum_k R_{k,x} \times EF^R_{k,x}$$

where

$A_x =$ Emission of $CH_4$ from natural gas distribution systems in year *x* (kg $CH_4$/year)
$P_{i,x} =$ Miles of distribution pipelines of material *i* in year *x* (mi)
$EF^P_{i,x} =$ Emissions factor for distribution pipeline of material *i* in year *x* (kg $CH_4$/mi)
$S_{i,x} =$ Number of gas services of material *i* in year *x* (# services)
$EF^S_{i,x} =$ Emissions factor for service lines of material *i* in year *x* (kg $CH_4$/service)
$M_{j,x} =$ Number of customer meters of type *j* in year *x* (# meters)
$EF^M_{j,x} =$ Emissions factor for customer meters of type *j* in year *x* (kg $CH_4$/meter)
$R_{k,x} =$ Number of M&R stations of size/type *k* in year *x* (# stations)
$EF^R_{k,x} =$ Emissions factor for M&R stations of size/type *k* in year *x* (kg $CH_4$/station)

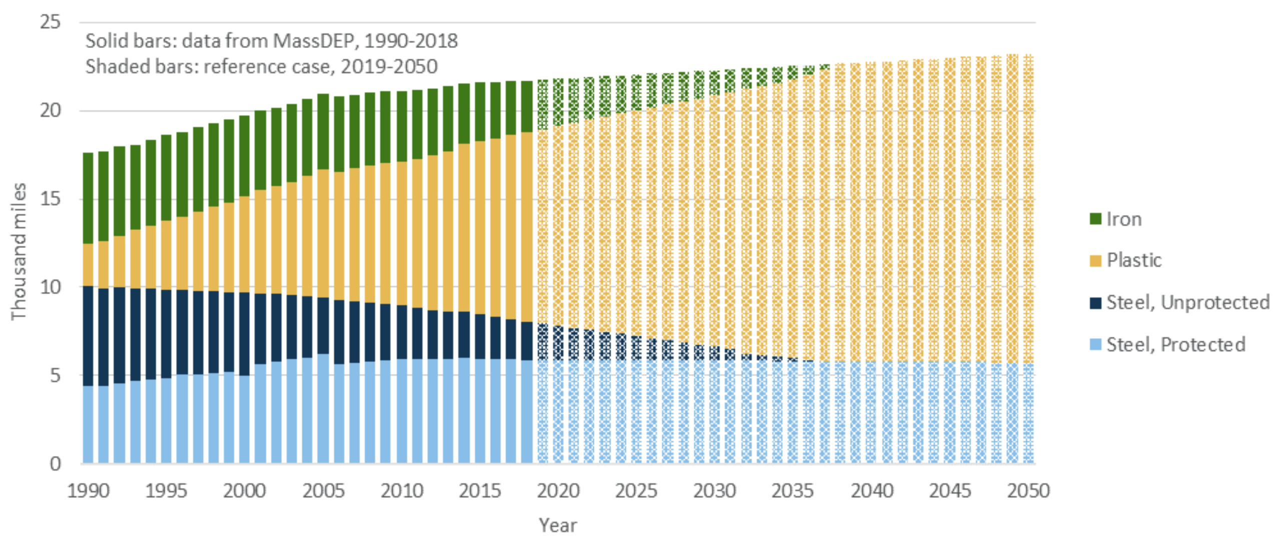
Data on miles of pipelines by material, number of services by material, and number and type of M&R stations have been collected by MassDEP. Data on the number of gas customers of different types are published by the U.S. Energy Information Administration (EIA). Emissions factors were established by a team from Washington State University (WSU) and published in 2015.[99,100]

Distribution pipelines in Massachusetts are made of steel (both cathodically-protected and not), plastic, and iron, with small amounts of copper and other materials. As seen in Figure 25, in 1990 gas pipelines were predominantly made of steel and iron. Over the subsequent two decades, the leakier iron and unprotected steel pipes have been replaced with more robust plastic, even as total distribution line mileage has grown from 17,600 miles in 1990 to 21,700 miles in 2018. The 2050 reference case forecast was established using a linear extrapolation of 2010-2018 rates of change of both total mileage and material replacement. According to the reference case forecast, by 2050, three-quarters of the 23,000 miles of distribution pipeline will be plastic, and one-quarter will be cathodically-protected steel. By 2038, all iron and unprotected steel distribution pipe will have been fully replaced, consistent with the current 310 CMR 7.73 regulation.

[99] [Lamb, et al. 2015]. Lamb, B.K., et al., "Direct Measurements Show Decreasing Methane Emissions from Natural Gas Local Distribution Systems in the United States," *Environmental Science & Technology, 49*: 5161-5169, 2015. https://doi.org/10.1021/es505116p

[100] This work replaced an earlier set of widely used emissions factors from a 1996 study by the U.S. EPA and the Gas Research Institute for which the base year was 1992.

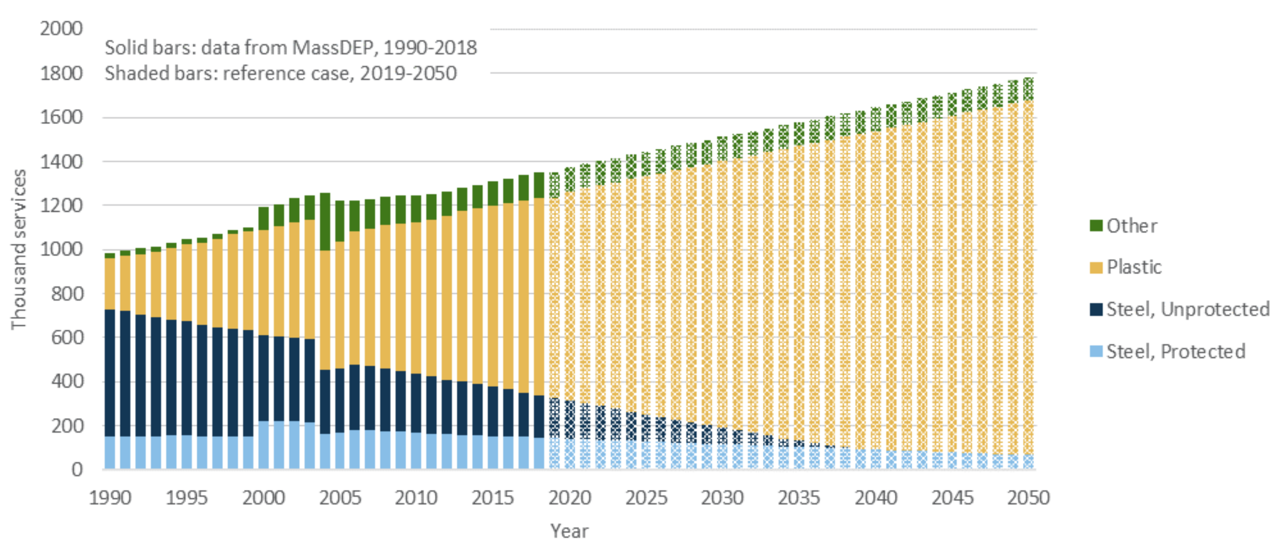
*Figure 25. Mileage and material of natural gas distribution pipelines in Massachusetts, 1990-2050 (reference case)*




The emissions factors calculated by WSU help to justify the turnover in pipeline material. Iron pipes were found to leak 1,147 kilograms of methane per mile-year (kg $CH_4$/mile-year), unprotected steel pipes leaked 811 kg $CH_4$/mile-year, cathodically protected pipes leaked 72 kg $CH_4$/mile-year, and plastic pipes leaked just 8.6 kg $CH_4$/mile-year. Additionally, MassDEP accounts for a variety of sources of periodic venting from all gas distribution pipes, such as pipeline blowdowns, pipeline dig-ins, and pressure relief valves, estimated to leak gas at a rate of 33.5 kg $CH_4$/mile-year.

Gas services, the smaller lines that connect distribution pipes to customers, are made of similar materials as pipelines. MassDEP data reproduced in Figure 26 show similar trends to pipelines as well: an overall increase in the number of services and an ongoing replacement of the older, leakier unprotected steel lines with newer, flexible, and robust plastic lines. The reference case continues these trends to 2050: an overall growth rate following a linear trend line based on 2010-2018 data and a constant replacement rate.

*Figure 26. Number and type of natural gas distribution services in Massachusetts, 1990-2050 (reference case)*

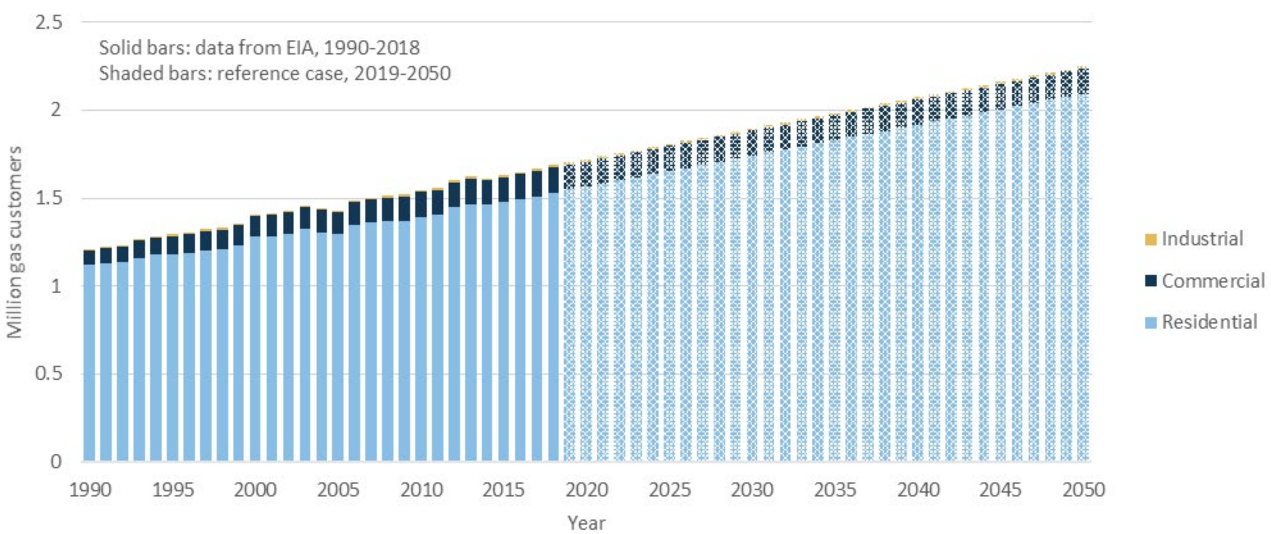


Emissions factors for services are established per service-year, irrespective of pipe length. Unprotected steel services were found to leak 5,180 kg $CH_4$/service-year, protected steel services 2,240 kg $CH_4$/service-year plastic services 210 kg $CH_4$/service-year, and copper services 4,880 kg $CH_4$/service-year.

Service lines connect gas distribution to residential, commercial, and industrial customers. The EIA collects and publishes data on the number of each type of gas customer in Massachusetts, seen here in Figure 27. A steady, constant growth of all three types of customers characterized the baseline years 1990-2018 (although there is an order of magnitude more residential customers than commercial customers, which is yet another order of magnitude larger than the number of industrial customers). The reference case forecast to 2050 followed the regular linear behavior of the historical data.

Emissions factors for the three customer types are provided by MassDEP, and are proportional to the expected size and throughput of each type of gas meter: residential meters leak 1.63 kg $CH_4$/meter-year, commercial meters leak 8.39 kg $CH_4$/meter-year, and industrial meters leak 231 kg $CH_4$/meter-year.

*Figure 27. Number and type of natural gas customers in Massachusetts, 1990-2050 (reference case)*




The last component of the natural gas distribution leaks model is the M&R stations. MassDEP and WSU account for many different types of M&R and regulating stations, varying by size, pressure, and location (above ground vs. vault). The data on the number of M&R and regulating stations are only available for years 1990 and 2014-2019. Data for 2019 are presented in Table 8. The brief time series (2014-2019) suggests a slight decrease in the total number of M&R stations, but there is little change in the number of the facilities with the highest emissions factors. The reference case therefore assumes the 2019 M&R stations values remain constant through 2050.

*Table 8. Number and emissions factors of M&R stations in Massachusetts, 2019*

| M&R Stations | # (2019) | Emissions factor (kg $CH_4$/station-year) |
|---|---|---|
| ***>300 psi (T-D)*** | 94 | 2,134 |
| ***100-300 psi (T-D)*** | 2 | 988 |
| ***>300 psi Above ground (not T-D)*** | 10 | 862 |
| ***>300 psi Vault (not T-D)*** | 4 | 52.6 |
| ***100-300 psi Above ground (not T-D)*** | 103 | 142 |
| ***100-300 psi Vault (not T-D)*** | 177 | 52.6 |
| ***40-<100 psi Above ground (not T-D)*** | 113 | 163 |
| ***40-<100 psi Vault (not T-D)*** | 441 | 52.6 |
| ***<40 psi (not T-D)*** | 215 | 0 |
| ***TOTAL*** | 1159 | |

### 6.2.2 Natural gas transmission

In its official inventory of Massachusetts GHG emissions associated with natural gas transmission, MassDEP uses the U.S. Environmental Protection Agency's (EPA) State Greenhouse Gas Inventory (SIT). The SIT offers an approach that applies simple emissions factors to annual data on miles of gathering and transmission pipeline and numbers of gas processing plants and liquefied natural gas (LNG) storage compressor stations.[101] Rather than using empirical data, the SIT projects numbers of gas transmission compressor stations and gas storage compressor stations based on miles of transmission pipeline. The emissions factors are based on 1992 data from a Gas Research Institute (GRI) study and do not account for the broad diversity of size, technology, and performance of gas compressors. The SIT aggregates emission factors across several activities to simplify data entry for states that use SIT.

MassDEP is currently updating the methodology for this sector, switching from using the SIT model to using updated emission factors from the U.S. EPA's annual national GHG inventory (GHGI) and actual annual emissions reported to U.S. EPA under the GHG Reporting Program (GHGRP) accessed through EPA's Facility Level Information on GHGs Tool (FLIGHT).

Annual facility-specific emissions data and other information became available through EPA FLIGHT for LNG import terminals in 2011, compressor stations in 2015 and pipelines in 2016. EPA's GHGI transmission sector methodology has undergone multiple updates in recent years. Additional GHGI component- and activity-specific emission factors have been developed for compressors, station flaring, venting blowdowns, pneumatic devices, LNG storage facilities and LNG import terminals. FLIGHT and GHGI also account for the carbon dioxide contained in natural gas. The GHGI methodology requires considerable knowledge of the state's various transmission system components. MassDEP has gathered such data from the U.S. Department of

[101] Massachusetts has no natural gas production (and therefore no gathering pipeline nor gas processing plants).

Transportation Pipeline and Hazardous Materials Safety Administration (PHMSA), FLIGHT, and natural gas transmission companies.

Estimates of methane emissions from natural gas transmission for 1990 and 2018 using the previous and updated approaches are shown in Table 9.

*Table 9. Estimated methane emissions from natural gas transmission in Massachusetts, 1990 and 2018, using two approaches*

| Emissions Source | Metric tons $CH_4$ | | | |
|---|---|---|---|---|
| | Previous SIT approach | | Updated GHGI/FLIGHT approach | |
| | 1990 | 2018 | 1990 | 2018 |
| ***Pipeline Leaks and Venting/Blowdowns*** | 574 | 703 | 574 | 59 |
| ***Compressor Stations*** | 6,762 | 8,290 | 3,296 | 824 |
| ***LNG Storage Facilities/Import Terminals*** | 7,110 | 7,110 | 2,833 | 1,700 |
| ***TOTAL*** | 14,446 | 16,103 | 6,703 | 2,583 |

Using the updated approach produces an estimate of emissions from natural gas transmission that is less than half of the SIT-based estimate for 1990 and less than one quarter of the SIT-based estimate for 2018. This is largely due to the aggregated nature of the SIT emission factors and the availability of more accurate factors from the GHGI. For example, SIT's LNG storage facility emission factor is about 13 times greater than the more accurate GHGI emission factor for an LNG storage facility. Similarly, the SIT compressor station emission factor is one third greater than the more accurate GHGI.

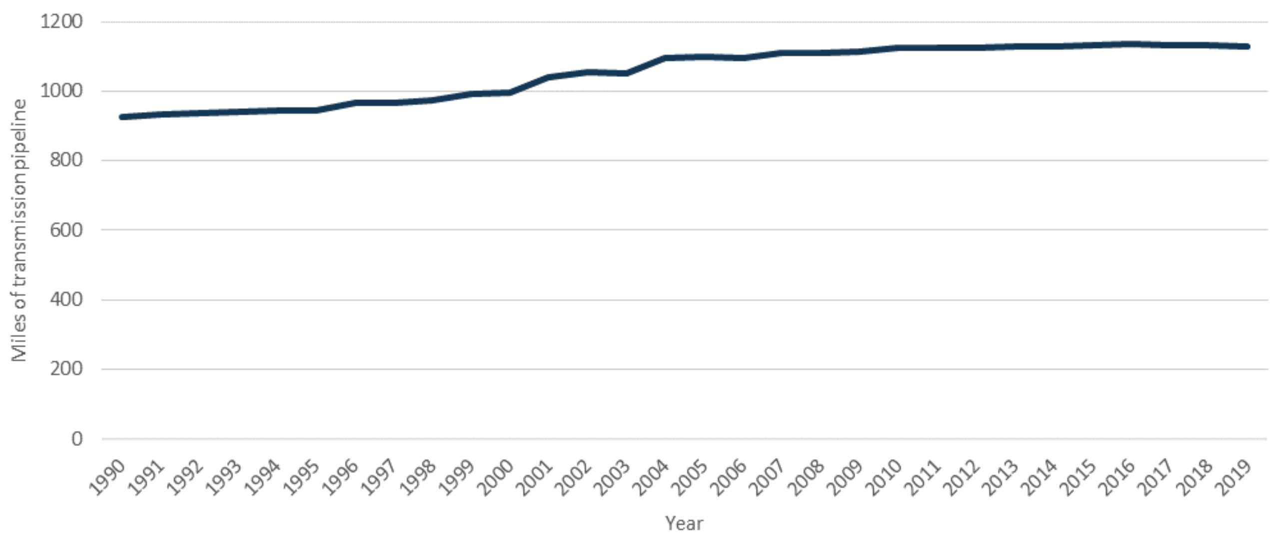
The updated approach discussed here is applicable only to this study's 2050 forecast at the time and not to the MassDEP emissions inventory presented in Figure 24. That inventory is only updated in concert with a public comment process.[102] MassDEP will consider proposing updates to the 1990 GHG emissions inventory for the transmission and other sectors in the future.

Transmission pipelines have been built around the Commonwealth over the past three decades. Data from PHMSA show 925 miles of transmission pipeline in 1990, growing to 1,134 miles in 2018. As can be seen in Figure 28, the pipeline growth effectively ceased between 2010 and 2019. In the reference case, it is assumed that transmission pipeline length is maintained at 2019 levels until 2050. Accordingly, the reference case also assumes no change to the number and operation of compressor stations and LNG storage facilities and import

[102] The *Statewide Greenhouse Gas Emissions Level: 1990 Baseline and 2020 Business as Usual Projection* (July 1, 2009) states: "The Department recognizes that the science and practice of determining GHG emissions is changing rapidly and that Massachusetts, being at the cutting edge of this work, should avail itself of advancements in the science to the extent possible. Therefore, MassDEP will reevaluate the 1990 Baseline as needed (e.g., significant new data becomes available). If amendment is necessary, a full public review process will be used." See p. 3 in https://www.mass.gov/doc/statewide-greenhouse-gas-emissions-level-1990-baseline-2020-business-as-usual-projection/download

terminals. Per these assumptions and the estimation approach discussed above, annual transmission emissions from 2018-2050 are held constant at 0.067 $MMTCO_2e$.[103]

*Figure 28. Miles of natural gas transmission pipeline in Massachusetts, 1990-2019*




## 6.3 Reference Case

The reference case for emissions from natural gas leaks is presented in Figure 29. This reference case assumes that the gas system in Massachusetts will be maintained through 2050, and so serves as an upper limit to emissions from this sector. Even so, the model suggests that emissions will be reduced from 0.73 $MMTCO_2e$ in 2010 to 0.33 $MMTCO_2e$ in 2050.[104] The rate of decline is not constant over time, with the rapid, 10%-per-year reductions seen from 2010-2014 slowing to 1-2%-per-year reductions from 2014-2038, followed by stability through 2050.

Even though the length of the distribution network grows by 10% and the number of gas services grows by 43% between 2010 and 2050 in this model, emissions from distribution pipelines and services are expected to decrease by 96% and 76%, respectively, due to pipe replacements. Emissions from meters are expected to grow by 21% as population and economic growth continue to expand the numbers of gas customers. Following a slight decrease from 2010-2019, emissions from the transmission system are assumed to stay constant at 2019 levels through 2050.

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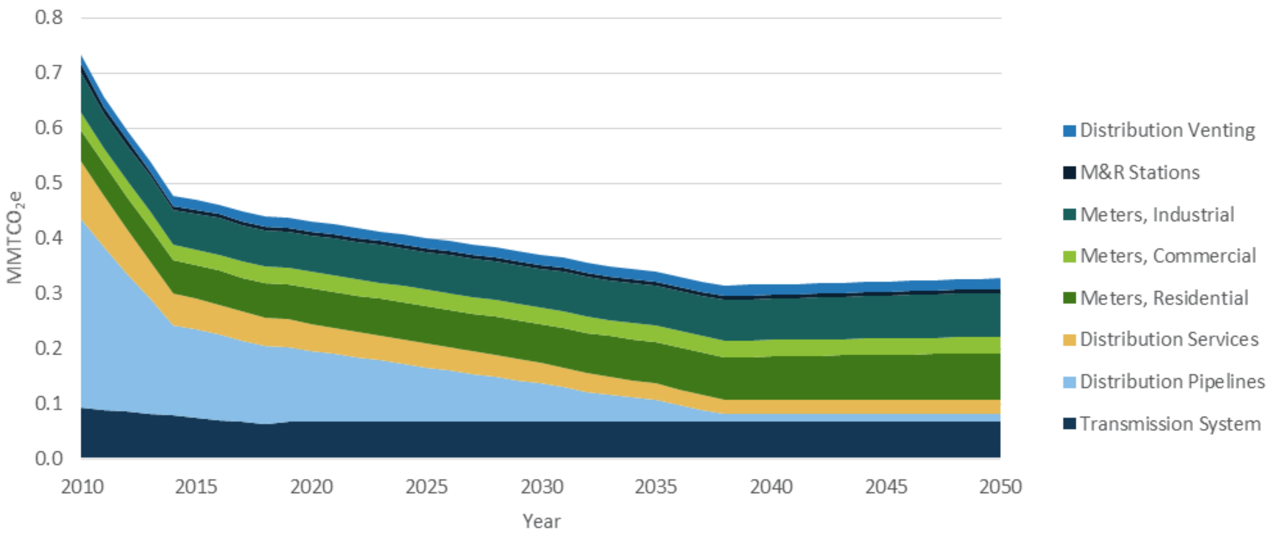
[103] This total is the sum of the 2018 GHGI/FLIGHT estimate shown in Table 9 and the 115 metric tons $CH_4$/year estimated to be released from the Weymouth compressor station, assuming a GWP of 25.

[104] These figures use the standard GWP-100 of $CH_4$ of 25. If a different number is used, these $MMTCO_2e$ figures should be recalculated.

*Figure 29. Reference case for emissions from natural gas leaks in Massachusetts, 2010-2050*




## 6.4 Discussion

Emissions reductions from this subsector are attributable to the replacement of old, leaky distribution pipes and service lines with newer, more-durable materials. Recent efforts to replace old infrastructure is the result of statute and regulatory action by the Department of Public Utilities and MassDEP. The 2014 Massachusetts Gas Leaks Act created a mechanism by which gas distribution companies proposed plans to replace leak-prone pipes with newer pipes.[105] The law required the companies to include timelines for accelerated replacement of aging infrastructure. In 2017, MassDEP promulgated a new regulation, 310 CMR 7.73 "Reducing Methane Emissions from Natural Gas Distribution Mains and Services" to set a limit on the amount of gas that could be leaked from gas operators' distribution networks with a decreasing cap from 2018-2020.[106] The calculations underlying the reference case for this sector assume compliance with state law requiring all leak-prone pipelines and service lines to be replaced.

Gas leaks through meters, although very small overall, are a persistent source of $CH_4$ emissions from this subsector. Residential and industrial meters pose inverse problems: there are millions of residential meters in the field, each emitting a very small amount of gas, while there are just thousands of industrial meters, each of which emits more than 100 times that of a residential meter. It is unlikely that much can be done about residential meters; the design and operation of the device makes some small fugitive emission almost inevitable. There is substantial uncertainty around the emissions factor for industrial meters, and more research and monitoring of this source of emissions is warranted.

The improved method for accounting emissions from gas transmission suggests that this source is much less of a problem than the current MassDEP inventory would suggest. The assumption that emissions from gas

[105] DPU, 2020.
[106] MassDEP, "310 CMR 7.73," 2020.

transmission, LNG, and compressor stations will remain constant through 2050 is uncertain; this system is not static and may continue to evolve in the future.

There is an important dependency between gas leaks and the energy sector. As residential and commercial heating and industrial boilers are electrified, gas meters and associated service lines will likely be taken offline, eliminating their contribution to system emissions. Leaks from distribution pipes will be maintained, however, as long as part of the gas system needs to remain pressurized. Conceptually, if electrification were to proceed neighborhood-by-neighborhood (or in a related, geographic-based strategy), it is possible that entire branches of the gas distribution network could be capped, eliminating sources of leakage altogether. At this point, there is not enough resolution in the gas leaks model to be able to quantify the effect of any certain electrification deployment strategy. The reference case can serve as an upper limit.

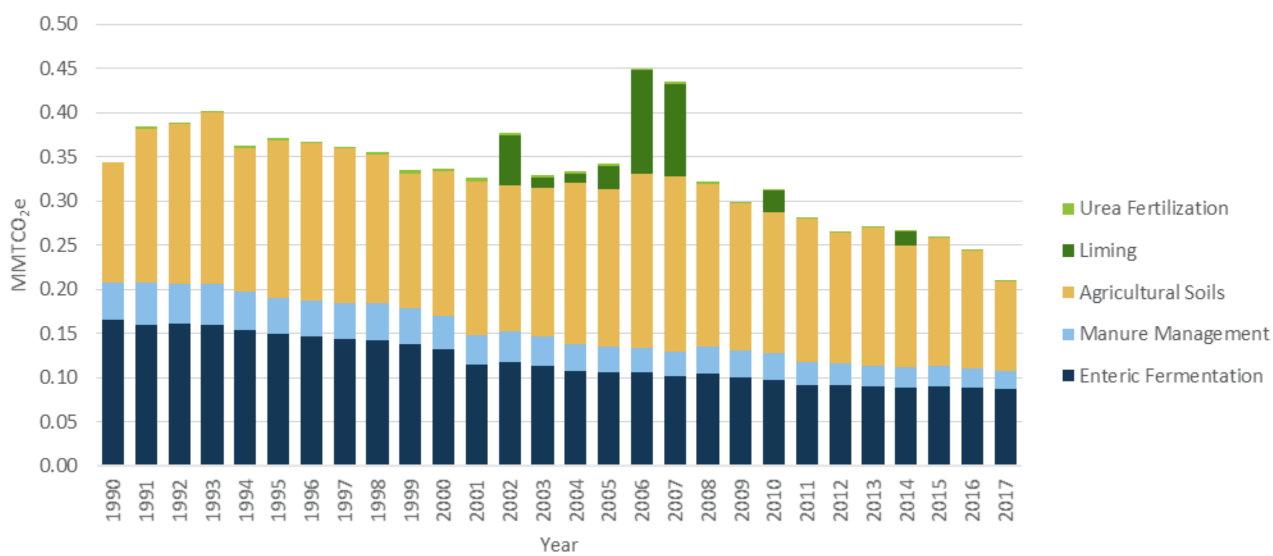
As shown in the *Energy Pathways Report*, all pathways analyzed to achieve net-zero emissions by 2050 included natural gas sales declining starting in 2020. As the Commonwealth moves towards the electrification of buildings, a tension emerges with the continued expansion of the gas distribution network. Specifically, electrification can lock in gas infrastructure that may become stranded assets of utilities and/or ratepayers. Fewer miles of pipeline and substations create fewer opportunities for gas leaks, and consistent with the *Energy Pathways Report* findings, the project team explored the effects on gas leaks of stopping the growth in the number of meters (residential, commercial, and industrial), the number of services, and the mileage of the distribution network in 2025, and holding that value constant from 2025 through 2050. This assumption avoids the installation of 25% additional meters by 2050 (compared with 2025 values), 23% new service lines, and 5% more distribution mileage. While avoiding this potentially unnecessary buildout is likely to have cost and system savings, stopping growth of the system in 2025 had only a modest impact on the gas leak emissions calculated in 2050 (0.30 $MMTCO_2e$ in 2050 as compared to 0.33 $MMTCO_2e$ under the reference case). However, the *Energy Pathways Report* found heavy electrification and minimal natural gas usage in residential customers by 2050, which could result in, or create the opportunity for, the decommissioning of potentially large portions of the state's widespread gas distribution network and allowing for reductions in gas leak emissions beyond those described here.

# 7 Agriculture

## 7.1 Introduction

Non-energy greenhouse gas (GHG) emissions from agriculture are the result of chemical and biological reactions on farms. The U.S. Environmental Protection Agency (EPA) acknowledges seven sources of anthropogenic GHG emissions in the agriculture sector: soil management, enteric fermentation, manure management, rice cultivation, urea fertilization, liming, and burning of agricultural residues. In Massachusetts, three of these activities are included in the state-wide emissions inventory: soil management, enteric fermentation, and manure management. In 2017, these activities emitted 0.21 million metric tons of $CO_2$-equivalent ($MMTCO_2e$), less than 0.3% of the total emissions inventory for that year (see Figure 30). Of that total, roughly half was due to soil management, another 42% from enteric fermentation, and the rest from manure management. Over the entire history of the state-wide GHG inventory, emissions from agriculture have been gradually declining, reflecting a steady decline in the agricultural sector in the Commonwealth.

*Figure 30. Current Massachusetts inventory of agricultural GHG emissions from MassDEP, 1990-2017.*[107]




In this chapter, Massachusetts GHG emissions from agriculture are forecast to 2050. Because of the small contribution this sector makes to the total and inherent limitations in both the emissions sources and the modeling framework, little is done to propose deep decarbonization strategies in agriculture.

## 7.2 Methodology & Data

The U.S. EPA has established a standard method for characterizing GHG emissions from agriculture and implemented it in their State Greenhouse Gas Inventory Tool (SIT). That approach is the same used here to

[107] [MassDEP, "MA GHG Inventory," 2020]. MassDEP, "Massachusetts Annual Greenhouse Gas Inventory: 1990-2017," 2020. Available at https://www.mass.gov/lists/massdep-emissions-inventories (Retrieved 19 August 2020).

develop the forecast. Only three activities are considered: agricultural soil management, enteric fermentation, and manure management.

### 7.2.1 Soil management

Emission of $N_2O$ from agricultural soils results from the oxidation and volatilization of nitrogen applied to soils in a variety of ways:

- Application of fertilizer, both synthetic and organic;
- Animal excretion; and
- Nitrogen fixation and residue from crops (in Massachusetts, just alfalfa).

Each source of nitrogen has a complex formula to estimate $N_2O$ emissions from an activity, a methodology too complex for the purposes of this report. Instead, a linear fit was used to forecast agricultural soil emissions to 2050, following the method used in the EPA State Inventory Projection Tool.

### 7.2.2 Enteric fermentation

Enteric fermentation is a digestion process that occurs in many farm animals that converts some animal feed to $CH_4$, which is then released to the atmosphere. The SIT approach uses the following formula:

$$CH_4^{EntFer} = \sum_a N_a \times EF_a$$

where

$$\begin{aligned} CH_4^{EntFer} &= \text{Emission of CH}_4\text{ from enteric fermentation} \\ N_a &= \text{Number of animal } a \\ EF_a &= \text{Emissions factor for animal } a \text{ (kg CH}_4\text{/head).} \end{aligned}$$

Data is provided by the SIT for the state-wide livestock population and forecast of 10 types of dairy and beef cows, five types of poultry, and five other types of farm animals (swine, sheep on feed, sheep not on feed, goats, and horses). The poultry do not contribute to enteric fermentation, and it is assumed that the EF for each type of animal is constant over time.

### 7.2.3 Manure management

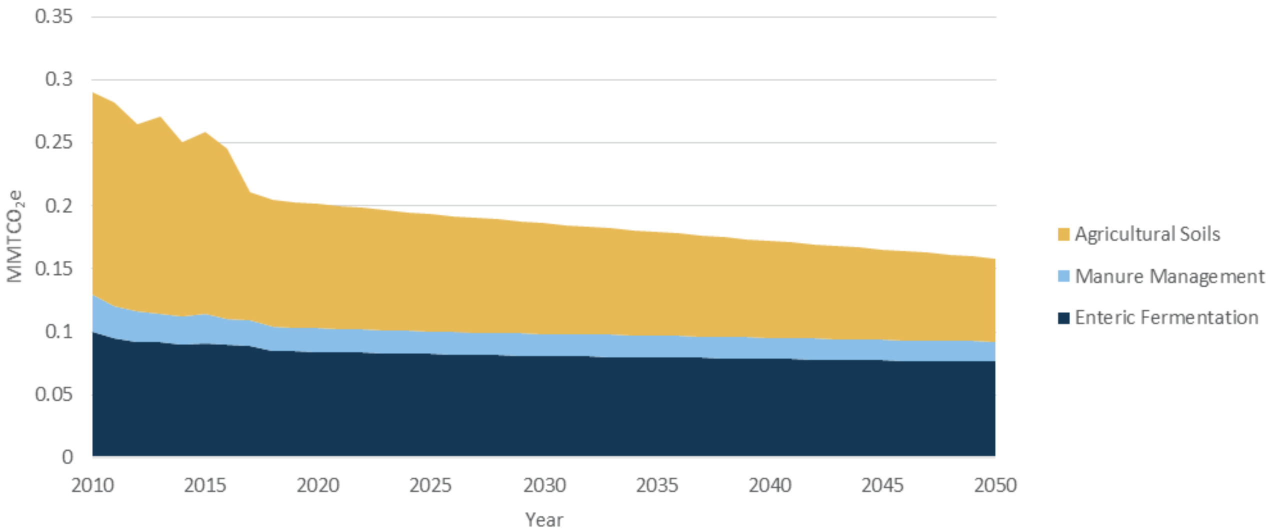
Manure management produces both $CH_4$ from anaerobic decomposition and $N_2O$ from oxidation of nitrogen in manure (distinct from the same process when it occurs in agricultural soils). As with agricultural soil emissions, the bottom-up forecast is very complex and not appropriate for a forecast of this type. Following the approach taken by the SIT Projection Tool, a linear fit was used to forecast manure management emissions to 2050.

## 7.3 Reference Case

The reference case for emissions from agricultural activates in Massachusetts is shown in Figure 31. Based on the continual decrease in agricultural activity in the Commonwealth, emissions are forecast to decline from

0.20 $MMTCO_2e$ in 2020 to 0.16 $MMTCO_2e$. Emissions from agricultural soils are seen to decline faster than the other two sources.

*Figure 31. Reference case for agricultural GHG emissions in Massachusetts, 2010-2050*




## 7.4 Discussion

The use of linear extrapolations of emissions from 2010-2017 do not provide much opportunity for sensitivity analysis. There is a considerable amount of uncertainty in both the models used to estimate emissions in the current inventory as well as the linear forecast. Agricultural activity is not expected to increase in Massachusetts, so the maximum emissions scenario from this sector could be defined as a maintenance of 2017 levels to 2050. The 2050 value would then be bounded as between 0.16-0.21 $MMTCO_2e$.

If agricultural activity declines faster than expected, future emissions may follow suit, but there is no evidence that such a future scenario is any more likely than is a continuation of current trends. In large agricultural states like California, agricultural emissions mitigation policies, practices, and technologies have been explored.[108] Given the small role that agriculture emissions play in the Massachusetts inventory, these policies and practices were not explored in full, however there may be a range of reasons to pursue improved agricultural practices in addition to their contribution to emissions reductions.

[108] [CARB, "Agricultural," 2020]. CARB, "Research on Agricultural Emissions & Mitigation," 2020. Available at https://ww2.arb.ca.gov/research/research-agricultural-emissions-mitigation (Retrieved 18 August 2020).

# 8 Industrial Processes

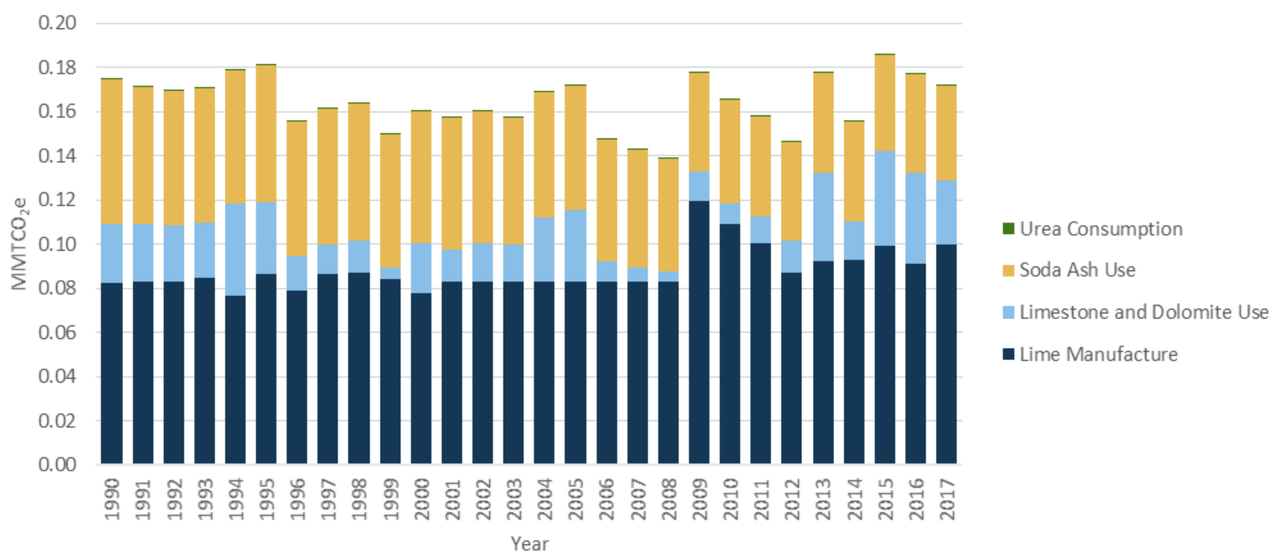
## 8.1 Introduction

Non-energy industrial emissions are the result of mineral and chemical processes that produce greenhouse gases (GHGs) as byproducts, usually through the oxidation of carbon and nitrogen into carbon dioxide ($CO_2$) and nitrous oxide ($N_2O$), respectively. The U.S. GHG Inventory includes a long list of industrial processes and products that fall into this category:[109]

- $CO_2$ is released from the production of iron & steel, metallurgical coke, cement, petrochemicals, ammonia, lime, and others; and the industrial consumption of carbonates, $CO_2$, and others;
- $N_2O$ is released during the production of adipic acid, nitric acid, and other chemicals.

The Massachusetts GHG Inventory includes a small subset of these industrial processes, reflecting the relatively minor presence of these industries in the Commonwealth. In 2017, industrial processes (excluding those involving F-gases, which are accounted for elsewhere in this study) were responsible for 0.17 million metric tons $CO_2$-equivalent (MMT$CO_2$e), less than 0.3% of total emissions (see Figure 32). The largest contributor was lime manufacturing, followed by soda ash consumption, limestone and dolomite use, and a very small quantity of non-agricultural urea consumption.

*Figure 32. Current Massachusetts inventory of industrial process GHG emissions from MassDEP, 1990-2017.*




In this chapter, these industrial process emissions are forecast to 2050. Like agricultural emissions, industrial process emissions comprise a small enough component of even the non-energy sector—oftentimes the result of just one or two facilities statewide per process—that performing a meaningful forecast is a challenge.

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[109] Emission of F-gases is included in the Industrial Processes and Product Use category in the U.S. GHG Inventory. All activities that release these compounds are included in the F-gases chapter and category here.

Accordingly, the analysis does not offer much insight into approaches for reducing emissions from industrial processes in Massachusetts.

## 8.2 Methodology & Data

Five industrial processes are shown to generate non-energy GHG emissions in Massachusetts: lime manufacturing, limestone use, dolomite use, soda ash consumption, and urea consumption. Urea consumption is so insignificant in the inventory that it is excluded from the forecasting model. The U.S. Environmental Protection Agency's State Greenhouse Gas Inventory Tool (EPA SIT) applies emissions factors to production and/or consumption quantities for each of the other four industrial sources. Data on production and consumption of these materials is available from the U.S. Geological Survey (USGS) Minerals Yearbooks.[110]

Since 2010, data on emissions from lime manufacturing is available from the U.S. EPA's Facility Level Information on Greenhouse Gases Tool (FLIGHT). In 2018, lime manufacturers in Massachusetts reported just over 100,000 $MTCO_2e$.[111]

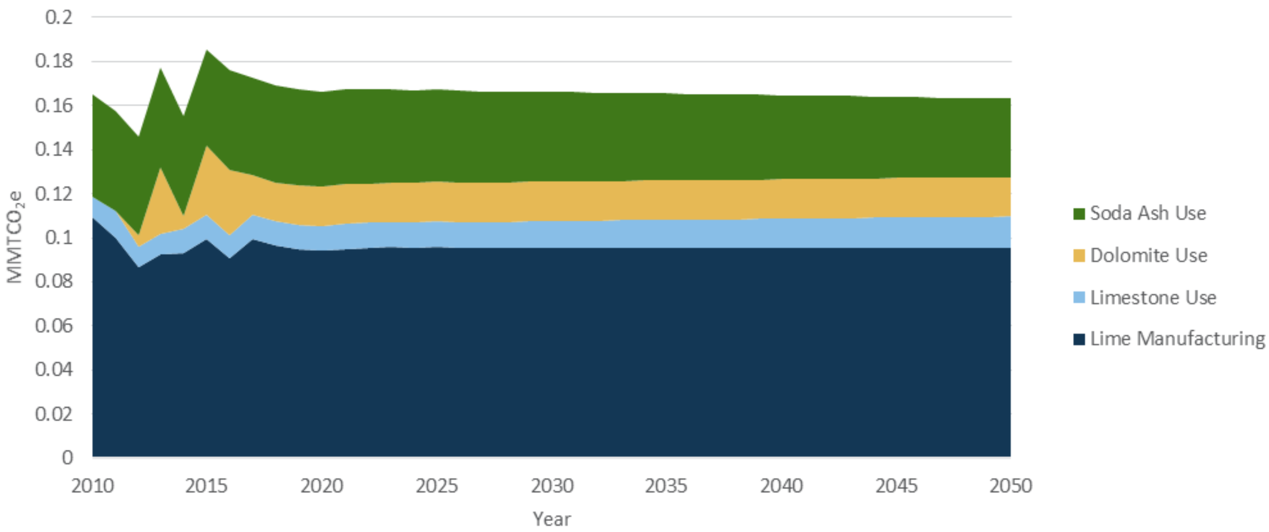
Due to the simplicity of the models and lack of data, the forecast methodology is limited to linear extrapolation based on the inventory from 2010-2017. Two of the processes (lime manufacturing and dolomite use) showed some degree of volatility in that time period. This is likely due to the very small number of relevant facilities. Given that we do not know the future production of these facilities, the average activity over the most recent years was used to forecast a constant level of emissions through 2050. Limestone use showed a slight increase and soda ash use showed a slight decrease over the period 2010-2017. These sectors were forecasted using linear extrapolation.

## 8.3 Reference Case

The reference case for industrial process emissions in Massachusetts is shown in Figure 33. In general, the case forecasts a stable emissions trajectory from these activities: 0.16 $MMTCO_2e$. Lime manufacturing contributes 58%, soda ash use 22%, dolomite use 11%, and limestone use 9%.

[110] [USGS, 2020]. USGS, "Minerals Yearbook," National Minerals Information Center, 2020. Available at https://www.usgs.gov/centers/nmic/minerals-yearbook-metals-and-minerals (Retrieved 18 August 2020).
[111] This FLIGHT category also includes emissions from glass manufacturing, but in Massachusetts, a single small glass manufacturing facility that reported to FLIGHT in 2018 has since shut down.

*Figure 33. Reference case for industrial process GHG emissions in Massachusetts, 2010-2050*




## 8.4 Discussion

The future of emissions from this sector are extremely volatile, but unlikely to grow much. Because each activity has a small number of firms involved, it is not unreasonable to expect entire sectors—and their corresponding $CO_2$ emissions—to disappear from the inventory at some point in the future. There is ongoing research to drive down emissions from various basic industrial processes,[112] but it is not feasible to apply findings from experimental studies to such small industrial activities in a quantitative modeling context.

[112] For example, [European Lime Association, 2014]. European Lime Association, "A Competitive and Efficient Lime Industry," Prepared by Ecofys, 2014. Available at https://www.eula.eu/wp-content/uploads/2019/02/A-Competitive-and-Efficient-Lime-Industry-Technical-report-by-Ecofys_0.pdf (Retrieved 18 August 2020).

# References


[CARB, 2016]. CARB, “California’s High Global Warming Potential Gases Emissions Inventory: Emissions Inventory Methodology Technical Support Document. 2015 Edition,” 2016. Available at https://ww3.arb.ca.gov/cc/inventory/slcp/doc/hfc_inventory_tsd_20160411.pdf (Retrieved 18 August 2020)

[CARB, 2017]. CARB, “Short-Lived Climate Pollutant Reduction Strategy,” 2017. Available at https://ww2.arb.ca.gov/sites/default/files/2018-12/final_slcp_reduction_strategy_w_appx_march2017%20Final%202017.pdf (Retrieved 18 August 2020).

[CARB, 2018]. California Air Resources Board, “Current and Projected Emission Inventory and Methodology of HFC-Gases, Black Carbon, and Methane in California and Other U.S. Climate Alliance States,” Draft Final, 2018.

[CARB, “Agricultural,” 2020]. CARB, “Research on Agricultural Emissions & Mitigation,” 2020. Available at https://ww2.arb.ca.gov/research/research-agricultural-emissions-mitigation (Retrieved 18 August 2020).

[CARB, “HFC Prohibitions,” 2020]. CARB, “Proposed Regulation Order: Prohibitions on Use of Certain Hydrofluorocarbons in Stationary Refrigeration, Stationary Air-conditioning, and Other End-Uses,” 2020. Available at https://ww2.arb.ca.gov/sites/default/files/2020-07/DRAFT%20CA%20SNAP%20Amendments-Reg%20Text.pdf (Retrieved 18 August 2020).

[CARB, “RMP,” 2020]. CARB, “Refrigerant Management Program: About,” 2020. Available at https://ww2.arb.ca.gov/our-work/programs/refrigerant-management-program/about (Retrieved 18 August 2020).

[CARB, “$SF_6$,” 2020]. CARB, “Staff report: Initial statement of reasons,” Regulation for Reducing Sulfur Hexafluoride Emissions from Gas Insulated Switchgear, 2020. Available at https://ww2.arb.ca.gov/rulemaking/2020/sf6 (Retrieved 28 September 2020).

[DPU, 2020]. DPU, “GSEPs Pursuant to 2014 Gas Leaks Act,” 2020. Available at https://www.mass.gov/lists/gseps-pursuant-to-2014-gas-leaks-act. (Retrieved 18 August 2020).

[EC, 2020]. European Commission, “Climate-friendly alternatives to HFCs,” 2020. https://ec.europa.eu/clima/policies/f-gas/alternatives_en (Retrieved 27 September 2020).

[European Lime Association, 2014]. European Lime Association, “A Competitive and Efficient Lime Industry,” Prepared by Ecofys, 2014. Available at https://www.eula.eu/wp-content/uploads/2019/02/A-Competitive-and-Efficient-Lime-Industry-Technical-report-by-Ecofys_0.pdf (Retrieved 18 August 2020).

[Gallagher, et al., 2014]. Gallagher, G., T. Zhan, Y.-K. Hsu, P. Gupta, et al., “High-global warming potential F-gas emissions in California: Comparison of ambient-based versus inventory-based emissions estimates, and implications of refined estimates,” *Environmental Science & Technology*, 48, 1084-1093, 2014. http://dx.doi.org/10.1021/es403447v.

[ICF, 2019]. ICF, “User’s guide for estimating carbon dioxide, nitrous oxide, HFC, PFC, NF3, and SF6 emissions from industrial processes using the State Inventory Tool,” Prepared for the U.S. EPA, November 2019.

[IPCC, 2019]. IPCC, “Wastewater Treatment and Discharge,” Ch. 6 in *2019 Refinement to the 2006 IPCC Guidelines for National Greenhouse Gas Inventories, Volume 5: Waste*, Switzerland: IPCC, 2019. Available at https://www.ipcc-nggip.iges.or.jp/public/2019rf/vol5.html (Retrieved 18 August 2020).

[Kaufman, et al., 2019]. Kaufman, N., D. Sandalow, C. Rossi di Schio & J. Higdon, “Decarbonizing Space Heating with Air Source Heat Pumps,” Columbia University Center on Global Energy Policy, 2019. Available at https://www.energypolicy.columbia.edu/research/report/decarbonizing-space-heating-air-source-heat-pumps (Retrieved 10 November 2020).

[Kigali Amendment, 2019]. “Kigali Amendment to the Montreal Protocol on Substances that Deplete the Ozone Layer." Available at https://ozone.unep.org/sites/default/files/2019-04/Original_depositary_notification_english_version_with_corrections.pdf (Retrieved 16 November 2020).

[Lamb, et al. 2015]. Lamb, B.K., et al., “Direct Measurements Show Decreasing Methane Emissions from Natural Gas Local Distribution Systems in the United States,” *Environmental Science & Technology*, 49: 5161-5169, 2015. https://doi.org/10.1021/es505116p

[Leverenz, Tchobanoglous & Darby, 2010]. Leverenz, H.L., G. Tchobanoglous & J.L. Darby. 2010. “Evaluation of Greenhouse Gas Emissions from Septic Systems.” DEC1R09. Alexandria, VA & London, UK: Water Environment Research Foundation (WERF) & IWA Publishing.

[MassDEP, “Capacity Study,” 2019]. MassDEP, “Massachusetts Materials Management Capacity Study,” prepared by MSW Consultants, 2019. Available at https://www.mass.gov/doc/massachusetts-materials-management-capacity-study-february-2019/download (Retrieved 19 August 2020)

[MassDEP, “Draft 2030 SWMP,” 2019]. MassDEP, “Massachusetts 2030 Solid Waste Master Plan,” Draft for public comment, 2019. Available at https://www.mass.gov/guides/solid-waste-master-plan (Retrieved 19 August 2020).

[MassDEP, “310 CMR 7.73,” 2020]. MassDEP, “Reducing Methane (CH4) Emissions from Natural Gas Distribution Mains & Services (310 CMR 7.73),” 2020. Available at https://www.mass.gov/service-details/reducing-methane-ch4-emissions-from-natural-gas-distribution-mains-services-310-cmr (Retrieved 18 August 2020).

[MassDEP, “310 CMR 7.76,” 2020]. MassDEP, “310 CMR 7.76: Proposed Prohibitions on Use of Certain Hydrofluorocarbons,” October 2020. Available at https://www.mass.gov/doc/310-cmr-776-proposed-prohibitions-on-certain-hydrofluorocarbons/download (Retrieved 10 December 2020).

[MassDEP, “310 CMR 15.000,” 2020]. MassDEP, “310 CMR 15.000: Septic Systems ("Title 5"),” 2020. Available at https://www.mass.gov/regulations/310-CMR-15000-septic-systems-title-5 (Retrieved 18 August 2020).

[MassDEP, “MA GHG Inventory,” 2020]. MassDEP, “Massachusetts Annual Greenhouse Gas Inventory: 1990-2017,” 2020. Available at https://www.mass.gov/lists/massdep-emissions-inventories (Retrieved 19 August 2020).

[McKain, et al., 2015]. McKain, K., et al., “Methane emissions from natural gas infrastructure and use in the urban region of Boston, Massachusetts,” *PNAS*, 112(7), 1941-1946, 2015. https://doi.org/10.1073/pnas.1416261112.

[Medalie, 1995]. Medalie, L., “Wastewater collection and return flow in New England, 1990,” Water Resources Investigation Report 95-4144, USGS, 1995. Available at https://pubs.usgs.gov/wri/1995/4144/report.pdf (Retrieved 10 December 2020).

[Metcalf & Eddy, 2003]. Metcalf & Eddy, Inc, Wastewater Engineering: Treatment, Disposal, Reuse, McGraw-Hill: New York, 2003.

[NEBRA & MassCEC, 2019]. NEBRA & MassCEC, “The Mass Sludge Survey 2018, v 1.1: Wastewater Solids Generation and Management in Massachusetts,” 2019.

[U.S. EPA, 2010]. U.S. EPA, “Greenhouse Gas Emissions Estimation Methodologies for Biogenic Emissions from Selected Source Categories: Solid Waste Disposal, Wastewater Treatment, Ethanol Fermentation,” prepared by RTI International, 2010.

[U.S. EPA, 2016]. U.S. EPA, "Proposed Determination on the Appropriateness of the Model Year 2022-2025 Light-Duty Vehicle Greenhouse Gas Emissions Standards under the Midterm Evaluation: Technical Support Document," EPA-420-R-16-021, 2016. Available at https://nepis.epa.gov/Exe/ZyPDF.cgi?Dockey=P100Q3L4.pdf (Retrieved 18 August 2020).

[U.S. EPA, 2018]. U.S. EPA, "EPA'S Vintaging Model of ODS Substitutes: A Summary of the 2017 Peer Review," EPA-400-F-18-001, 2018. Available at https://www.epa.gov/sites/production/files/2018-09/documents/epas-vintaging-model-of-ods-substitutes-peer-review-factsheet.pdf (Retrieved 10 December 2020).

[U.S. EPA, 2020]. U.S. EPA, "Inventory of U.S. Greenhouse Gas Emissions and Sinks: 1990-2018," EPA 430-R-20-002, 2020. Available at https://www.epa.gov/ghgemissions/inventory-us-greenhouse-gas-emissions-and-sinks-1990-2018 (Retrieved 19 August 2020).

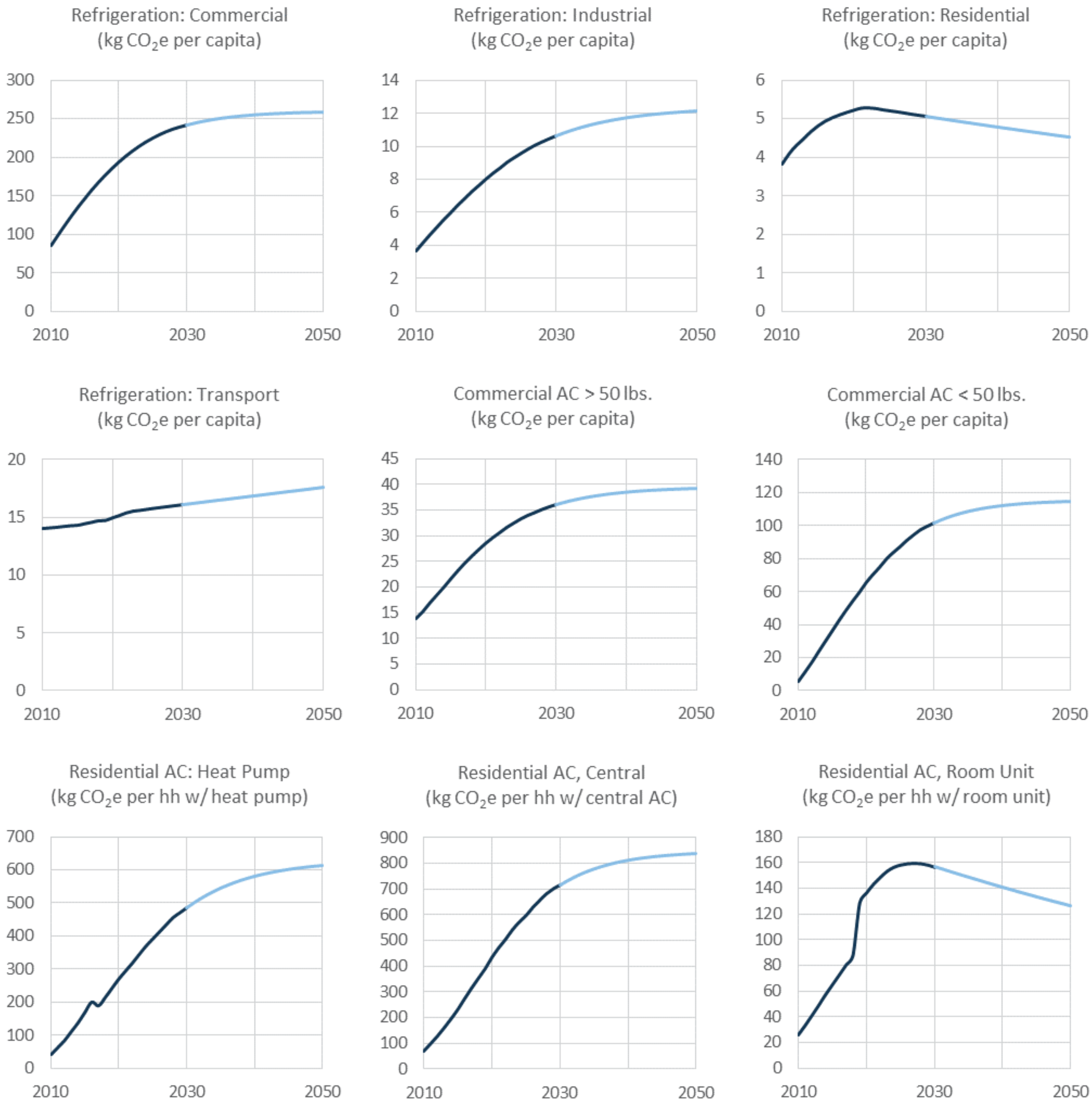
[UNEP & IEA, 2020, p. 20]. UNEP & IEA, "Cooling Emissions and Policy Synthesis Report," Nairobi & Paris: UNEP & IEA, 2020. Available at https://www.iea.org/reports/cooling-emissions-and-policy-synthesis-report (Retrieved 19 August 2020).

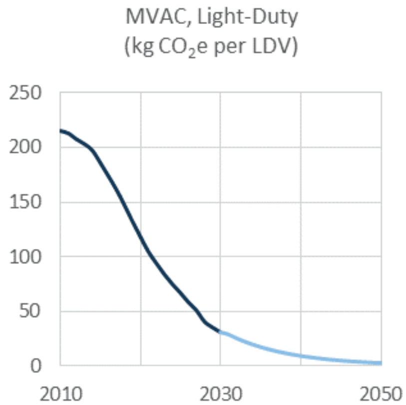
[USGS, 2020]. USGS, "Minerals Yearbook," National Minerals Information Center, 2020. Available at https://www.usgs.gov/centers/nmic/minerals-yearbook-metals-and-minerals (Retrieved 18 August 2020).

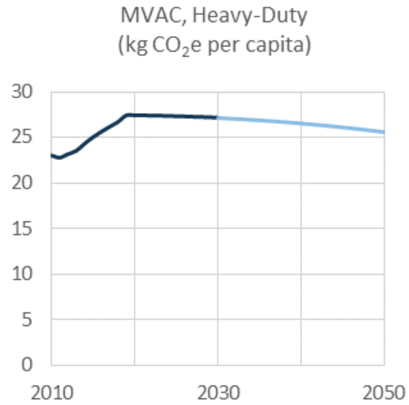
[WMO, 2018]. World Meteorological Organization, "Scientific Assessment of Ozone Depletion: 2018. Global Ozone Research and Monitoring Project—Report No. 58," Geneva, Switzerland: WMO, 2018. Available at https://csl.noaa.gov/assessments/ozone/2018/ (Retrieved 10 December 2020).

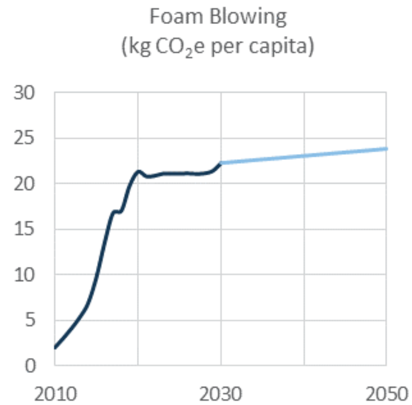
# Appendix A. F-gas Emissions Intensity Factors

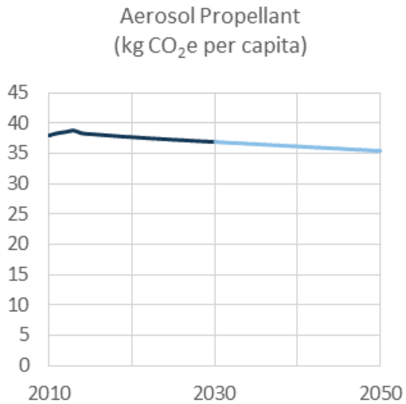
Emissions Intensity Factors for the 14 End-Use Sectors in the CARB/USCA model, 2010-2030 and 2030-2050.

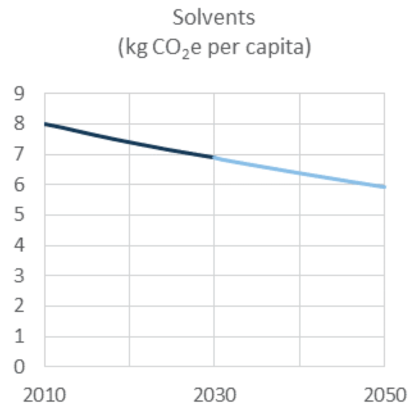



MVAC, Light-Duty
(kg $CO_2$e per LDV)
250
200
150
100
50
0
2010
2030
2050


MVAC, Heavy-Duty
(kg $CO_2$e per capita)
30
25
20
15
10
5
0
2010
2030
2050


Foam Blowing
(kg $CO_2$e per capita)
30
25
20
15
10
5
0
2010
2030
2050


Aerosol Propellant
(kg $CO_2$e per capita)
45
40
35
30
25
20
15
10
5
0
2010
2030
2050


Solvents
(kg $CO_2$e per capita)
9
8
7
6
5
4
3
2
1
0
2010
2030
2050

# Appendix B. Landfills in MA, Closed & Active

Landfill data, including inventory start and end years, are drawn from the EPA Greenhouse Gas Reporting Program and EPA FLIGHT.

| Facility Name | Starting Year | Closure Year | Landfill capacity (MMT) | Surface area (ha.) |
|---|---|---|---|---|
| ***Bondi Island (Agawam)*** | 1968 | *active* | 1.68 | 28.9 |
| ***Bourne*** | 1967 | *active* | 2.61 | 19.6 |
| ***Carver-Marion-Wareham*** | 1973 | *active* | 2.20 | 26.3 |
| ***Chicopee*** | 1967 | 2019 | 14.65 | 30.6 |
| ***Crapo Hill (New Bedford)*** | 1995 | *active* | 1.81 | 15.2 |
| ***East Bridgewater*** | 1973 | 1996 | 3.84 | 34.4 |
| ***Fall River*** | 1940 | 2014 | 6.21 | 59.5 |
| ***Fitchburg Westminster*** | 1972 | *active* | 11.16 | 36.2 |
| ***Gardner Street (Boston)*** | 1963 | 1980 | 2.40 | 34.4 |
| ***Granby*** | 1970 | 2014 | 7.15 | 22.3 |
| ***Halifax*** | 1977 | 1992 | 2.79 | 17.8 |
| ***Hardwick*** | 1970 | 2007 | 0.73 | 8.6 |
| ***Haverhill*** | 1982 | *active* | 7.74 | 26.7 |
| ***Martone (Barre)*** | 1976 | 2015 | 4.50 | 24.0 |
| ***Middleborough*** | 1962 | *active* | 1.98 | 13.2 |
| ***Northampton*** | 1969 | 2013 | 2.63 | 15.2 |
| ***Plainville*** | 1975 | 1998 | 6.74 | 36.8 |
| ***Randolph*** | 1960 | 1995 | 2.44 | 37.6 |
| ***South Hadley*** | 1951 | 2014 | 1.89 | 14.6 |
| ***Southbridge*** | 1981 | 2018 | 4.36 | 20.8 |
| ***Taunton*** | 1938 | *active* | 4.32 | 21.9 |
| ***West Street (Gardner)*** | 1970 | 2005 | 2.73 | 11.7 |