Methane and Health-Damaging Air Pollutants From the Oil and Gas Sector: Bridging 10 Years of Scientific Understanding

October 6, 2021

PSE Bringing science to energy policy

Authors

Drew R. Michanowicz, DrPH, MPH, CPH[1,2]
Eric D. Lebel, PhD[1]
Jeremy K. Domen, MS[1]
Lee Ann L. Hill, MPH[1]
Jessie M. Jaeger, MPH, MCP[1]
Jessica E. Schiff, SM[1,3]
Elena M. Krieger, PhD[1]
Zoya Banan, PhD[1,4]
Jackson S.W. Goldman, BS[1]
Curtis L. Nordgaard, MD, MSc[1]
Seth B.C. Shonkoff, PhD, MPH[1,5,6]

1   *PSE Healthy Energy, Oakland, CA*
2   *C-CHANGE, Harvard T.H. Chan School of Public Health, Boston, MA*
3   *Department of Environmental Health, Harvard T.H. Chan School of Public Health, Boston, MA*
4   *South Coast Air Quality Management District, Los Angeles, CA*
5   *Department of Environmental Science, Policy and Management, University of California, Berkeley, Berkeley, CA*
6   *Energy Technologies Area, Lawrence Berkeley National Laboratory, Berkeley, CA*

Funding

About PSE Healthy Energy

Physicians, Scientists, and Engineers (PSE) for Healthy Energy is a multidisciplinary, nonprofit research institute dedicated to supplying evidence-based scientific and technical information on the public health, environmental, and climate dimensions of energy production and use. We put our mission into practice by integrating scientific understanding across multiple disciplines, including engineering, environmental science, and public health. We conduct original research, translate existing research for nontechnical audiences, and disseminate scientific information and analyses to inform policy at the local, state, and federal levels. We focus on the overlap of energy production, public health, and the natural environment and produce vetted scientific analyses.

PSE

PSE Healthy Energy
1440 Broadway, Suite 750
Oakland, CA 94612
510-330-5550
info@psehealthyenergy.org
www.psehealthyenergy.org

Executive Summary

The global atmospheric concentration of methane—a potent greenhouse gas that captures 86 times more heat than carbon dioxide (CO_2) over a 20-year time span—is now greater than at any time in the past 800,000 years.[1] Methane emissions must be reduced by almost half during the next decade to avoid the worst effects of climate change.[2] The oil and gas sector—including upstream oil and gas development, midstream oil and gas transmission, and downstream end use—is simultaneously the largest source of anthropogenic methane emissions and volatile organic compounds (VOCs) in the United States, emitting an estimated 196.7 million metric tons of CO_2 equivalent in methane in 2019 and an estimated 2.49 million tons of VOCs annually.[3]

Together, these VOCs and other health-damaging air pollutants (HDAPs)—defined in this report as any airborne pollutant in particulate or gaseous form that is hazardous to human health—emitted from the oil and gas sector degrade air quality and introduce human health hazards, risks, and impacts at local and regional scales. Reducing anthropogenic methane and HDAP emissions is one of the most cost-effective strategies to rapidly reduce the rate of warming, buy time for deeper global decarbonization,[4] and realize air quality and health benefits.

In 2016, for the first time, the United States Environmental Protection Agency (U.S. EPA) finalized new rules to regulate both methane and VOC emissions from the oil and gas industry.[5] While the rules monetized climate-related benefits for methane reductions, the U.S. EPA was unable to calculate the benefits of VOC and other HDAP reductions due to "difficulties in modeling the impacts with the current data available."[6] These data gaps indicate that benefits of oil and gas methane regulations are currently underestimated across the oil and gas industry, and therefore, have ramifications for present and future energy policy.

To better understand the overlap in sources of methane and HDAPs and approaches to prevent, detect, and mitigate emissions, we conducted a systematic review of scientific peer-reviewed literature published between January 2015 and August 2020. Our review focused on primary methane and HDAP data collection efforts pertinent to emissions throughout the oil and gas supply chain, divided into upstream (development and production), midstream (processing and transmission), and downstream (distribution and end-use) sectors, with the following objectives:

- **Summarize Key Themes and Findings:** From our systematic review, we aimed to equip researchers, communities, and decisionmakers with a clear, concise, and informative guide to the state of the science, current research gaps, and priorities on issues of methane and HDAP emissions across the oil and gas sector.

---

1 IPCC (2021)

2 UNEP & CCAC (2021)

3 U.S. EPA (2021)

4 UNEP & CCAC (2021)

5 81 FR 35824, June 3, 2016

6 U.S. EPA (2020)

PSE

- **Highlight Research and Policy Recommendations:** Based on the findings of the systematic review, we provided recommendations that can be incorporated into actionable climate policy that simultaneously reduces methane emissions and protects public health.

Key Themes, Findings, and Recommendations

We identified 270 unique studies published from January 2015 to August 2020 that met our inclusion criteria. Of this body of literature, 165 articles measured methane emissions alone, 76 articles measured HDAP emissions alone, and 29 articles simultaneously measured both methane and HDAPs (Figure ES-1). We restricted our review to articles that measured emissions from North America (i.e., Canada, the United States, and Mexico). We found more than twice the number of methane studies were published than HDAP studies, and 72% of methane articles and 84% of HDAP articles published during this time focused on some aspect of the upstream oil and gas sector (Figure ES-1). Below we describe key findings, research gaps, and recommendations from our literature review, organized under overarching themes.
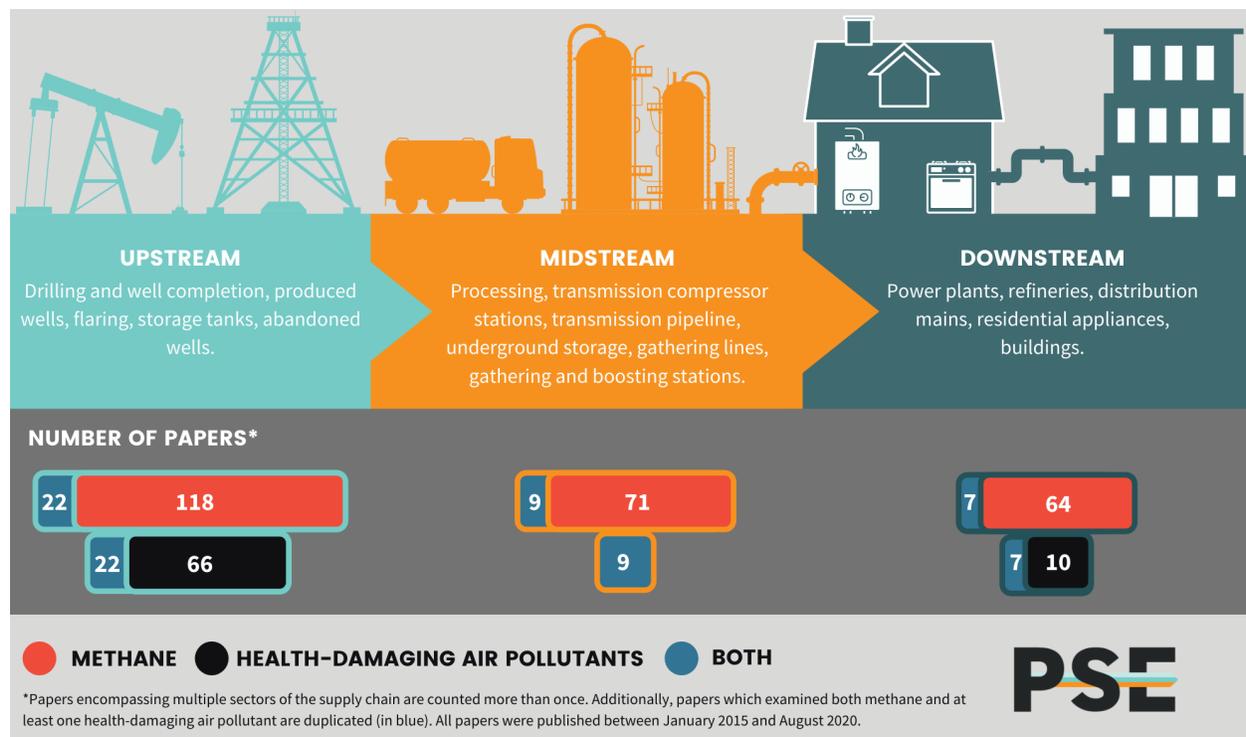
Figure ES-1. Counts of methane and HDAP studies across upstream, midstream, and downstream sectors from the systematic review, 2015–2020. This schematic shows the components from the upstream, midstream, and downstream sectors investigated in this report. The scale bars below each of the sectors indicate the relative weight of evidence available for that sector, as determined by the number of peer-reviewed publications included in our analysis. Papers encompassing multiple sections of the supply chain are counted in each of their respective sectors. Papers which measured methane alone are denoted by red bars, HDAPs alone in black, and both methane and HDAPs in blue.

Theme #1. The composition and magnitude of emissions from oil and gas systems depends on multiple variables.

Finding 1.1. There are two major source-types of air pollutants from oil and gas systems: (1) fugitive emissions from leaks and venting of non-combusted gases that emit relatively high proportions of methane and VOCs and (2) combustion emissions that emit comparatively less methane through incomplete combustion and possibly fewer VOCs, but higher levels of other criteria pollutants as byproducts of combustion such as nitrogen oxides (NO_x), carbon monoxide (CO), and particulate matter (PM), among others. In general, sources of methane emissions are nearly always sources of health-damaging air pollutants (HDAPs). However, not all sources of HDAPs are sources of methane.

Finding 1.2. Major equipment sources associated with emissions of unburned fugitive methane and health-damaging VOCs include liquid storage tanks, produced water tanks, dehydrators, pneumatic controllers, and any fugitive leak point that exists from well pad to end-use. HDAPs emitted during fuel combustion to support the oil and gas supply chain include NO_x, CO, PM, VOCs, black carbon, sulfur dioxide, formaldehyde, and ammonia. Major equipment sources of combustion-related emissions include flares, natural gas-fired compressor stations, and diesel fuel-powered equipment typically deployed for drilling and well completion activities.

Finding 1.3. Emissions from oil and gas systems vary by production basin, operator, emission source, emissions process, activity phase and operational status, and emission control technology and measures applied. Temporal aspects of emissions, including intermittent or persistent emissions, also adds complexity and uncertainty in characterizing emissions and subsequently designing mitigation strategies.

Finding 1.4. All supply chain equipment moves through numerous operational phases—including normal functioning, malfunctioning (i.e., abnormal operation), stand-by, and maintenance—emitting both methane and HDAPs at varying intensities and proportions. Operational vs. stand-by status influences the make-up and intensity of emissions, with greater emission rates observed in functional operating status (vs. standby mode) across a host of facility and equipment types. Similarly, different stages of upstream oil and gas development emit HDAPs and methane at varying mixtures and concentrations. For example, emission rates can range by orders of magnitude with different co-pollutant mixtures observed during hydraulic fracturing, flowback, and production stages of well development.

Finding 1.5. A wide range of methane mitigation strategies and monitoring technologies emerged in the past decade. In general, these include engineered emission controls, operational practices such as leak detection and repair programs (LDAR), and various configurations of air-quality monitoring and remote sensing. The general effectiveness of each of these approaches was studied closely and will remain

PSE

an ongoing area of study. These studies uncovered pros and cons of each of these systems suggesting a future landscape that likely entails tailored mitigation strategies that combine multiple mitigation approaches in a hybridized and redundant form.

Finding 1.6. Despite evidence that methane and HDAPs are emitted from every sector of the oil and gas supply chain, much of the scientific literature tends to evaluate methane and HDAPs as separate issues, resulting in disparate literatures. Only 11% (29) of the studies we reviewed explicitly measured methane and at least one HDAP simultaneously.

Finding 1.7. Certain mitigation measures—such as more frequent upstream leak detection and repair (LDAR)—are not designed to capture non-methane-rich HDAP emissions that contribute to health hazards, risks, and impacts. While methane can be a reasonable indicator for HDAPs when the source is methane-rich (e.g., wellheads, natural gas processing plants, natural gas gathering and transmission infrastructure, household natural gas appliances), methane may be a poor indicator for HDAPs when the source is not typically methane-rich (e.g., flaring, natural gas combustion, heavy oil flashing, produced water management processes, diesel combustion emissions).

Recommendation 1.1. The greatest opportunity to co-reduce both methane and at least one class of HDAPs is by targeting fugitive or vented emissions of natural gas at any point in the oil and gas supply chain. Any efforts to reduce venting and leaks will result in the co-reduction of both methane and certain VOCs, such as benzene, toluene, ethylbenzene, and xylene (BTEX). There is emerging consensus that upstream liquid storage tanks are the single largest fugitive emissions source of both methane and VOCs, and that these emissions are disproportionately underestimated in inventories.

Recommendation 1.2. Additional research is still needed to better understand: (1) the methane to HDAP ratio of emissions from many source-types; (2) the full life cycle of HDAP emissions for many pollutants; and (3) the health impacts associated with HDAP exposures across the oil and gas supply chain. Specifically, more primary research is needed to develop methane to VOC emissions ratios for many source-types. More research is also needed in the downstream sector, specifically for indoor building-level HDAP emissions that can occur very near people, as well as the degree to which widespread distribution leakage affects regional air quality in urban areas. A better understanding of the impacts in these settings could help in quantifying the public health benefits of certain methane emission reduction strategies.

Recommendation 1.3. Four of the most effective emissions controls technologies that simultaneously reduce major sources of methane and HDAPs are: (1) tankless well designs/additional vapor control measures on liquid storage tanks and dehydrators; (2) electrification of compressor engines; (3) preventing the need for flaring through additional separators or vapor-recovery units or improving efficien y of flaring by more accurate monitoring of steam and/or air-assisted flares; and (4) replacement and

PSE

repair of existing pneumatic devices with zero-bleed devices. Permitting requirements for centralized facilities should include emission control measures, such as the use of low-NO_x or electrified compressor engines, as well as emission rate limitations for NO_x, CO, and VOCs for both compressor engines and all associated process equipment. Moreover, all centralized facilities—including gathering and boosting stations, processing plants, refineries, and natural gas-fired power plants—present opportunities for co-reductions of methane and HDAPs through emissions controls such as engine electrification paired with monitoring systems to verify controls. Additionally, operational best practices that reduce the likelihood of fugitive emissions (for example, open thief hatches) can be very effective forms of emissions control.

Recommendation 1.4. A narrow focus on only controlling emissions from methane-rich sources may lead to a lack of control on other important sources of HDAPs, which can degrade air quality and impact public health. Further development of combustion control technologies should be considered since combustion source-types account for the majority of criteria air pollutant emissions across the life cycle. Examples include: (1) engine electrification where possible and the adoption of new front-end fuel combustion technology, which reduces emissions of NO_x and CO; (2) the implementation of continuous emissions monitoring using more sensitive instruments to capture and adequately mitigate pollutant releases and leaks when they happen; and (3) the implementation of responsive, automatic system controls for specific systems (e.g., combustion, flare control, cooling tower, sulfur recovery unit), which could promote more efficient fuel ombustion and reduce process leaks.

Recommendation 1.5. Methane detection and measurement technologies like fixed sensors, mobile laboratories, unmanned aerial vehicles (UAVs), aircraft, and satellites are promising tools that in total meet a range of applications with well-defined tradeoffs related to spatial and temporal sensing resolution and a range of detection/quantification effi acy. Hybridized mitigation strategies or a hierarchy of controls is recommended by multiple research groups. For example, the cost-effectiveness of LDAR programs can be improved by utilizing a multi-platform hybrid screening and confirmation approach, whereby rapid screening technologies—such as vehicle-, aerial-, or satellite-based platforms—can be used to guide ground-based LDAR programs as opposed to routine LDAR sampling regimes (i.e., quarterly). Future mitigation strategies should continue to integrate newer technologies with LDAR programs and should seek to optimize LDAR survey frequency, detection thresholds, and response times.

Recommendation 1.6. On-site stationary monitoring systems were found to be the most capable technology available to simultaneously quantify methane and HDAP emissions. Approaches to on-site air quality monitoring for HDAPs need to first consider the composition of gases emitted from nearby source-types. For instance, components that handle methane-rich gases could benefit from continuous methane monitoring with an auto-trigger VOC-sampling system. These samples can provide

speciated VOCs at concentrations suitable for developing better methane:VOC ratio data that is needed for air quality modeling and health impact analysis. These types of stationary monitoring systems can provide routine data that could support the advent of measurement-based inventories over current activity-based inventories. Such inventories could also support the implementation of performance-based regulations that have been applied in other countries and in other U.S. industrial sectors.

Theme #2. Given that sources of methane emissions are nearly always sources of health-damaging air pollutants, prioritizing methane emission reductions near population centers is essential to protect public health and would increase the benefits of methane emission reductions.

Finding 2.1. There is strong evidence that emissions of both methane and HDAPs vary substantially by location, equipment type, production phase, maintenance practices, and many other factors. Therefore, the health benefits per ton of methane reduced will also likely vary substantially by location, and are further mediated by nearby population density, presence or absence of other ambient pollutants, and meteorology.

Finding 2.2. The location of methane emissions does not change their climate impact; however, the location of HDAP emissions determines local and regional air quality and subsequent public health risks. Natural gas leakage studies indicate that all portions of the vast supply chain are prone to leaks, even in the downstream sector where natural gas is delivered to cities and buildings. In these settings, even small concentrations of co-emitted HDAPs may be important due to their proximity to human populations. Downstream emissions contribute to existing urban air quality burdens and in some cases are directly emitted inside of buildings from appliance leaks that can go undetected.

Finding 2.3. The body of scientific evidence indicates that the likelihood of adverse human health outcomes increases as distance decreases between oil and gas development operations and human populations. The risks associated with HDAPs stemming from upstream oil and gas sites can be reduced by establishing larger distances between upstream oil and gas operations and human populations. However, much less is known about the HDAP exposures and adverse human health outcomes associated with midstream and downstream natural gas distribution systems, even though many of these systems are located near human populations. Only 9% of studies in our review examine midstream HDAP emissions and another 16% measure downstream HDAP emissions, while 84% measure upstream HDAP emissions, highlighting a clear gap in the literature when exposures to human health need to be identified.

Recommendation 2.1. Prioritizing methane reductions near population centers can increase the public health and cost-benefit of emissions reductions. Centralized facilities located near population centers such as compressor stations, gathering and

PSE

boosting facilities, large-scale production pads, and other processing plants should be prioritized for emissions mitigation.

Recommendation 2.2. Regulators and risk managers should consider other risk reduction strategies and safety measures beyond emissions controls and monitoring. These can include implementation of minimum surface setbacks between various types of oil and gas operations—beyond just locations of production wells—and human receptors with additional protections for nearby sensitive populations.

Theme #3. A relatively small proportion of sources are responsible for a disproportionately large fraction of total methane and HDAP emissions. These super-emitters are present in every sector of the oil and gas supply chain and present both the greatest challenge and greatest opportunity for mitigation.

Finding 3.1. A relatively small number of sources located throughout the oil and gas supply chains are responsible for a disproportionately large fraction of methane and HDAP emissions. For methane, these types of sources have been deemed "super-emitters," however, the term and its definition have not been standardized in the scientific literature and are employed differently depending on the context. Attempts have been made to classify sources as super-emitting, such as the "5-50" rule, where the top 5 percent of emitters are responsible for 50 percent of all emissions. However, large intra- and inter-regional variations have been observed for individual components and for regions as a whole.

Finding 3.2. The presence of super-emitters writ large is evidence that substantial emissions reductions are feasible.

Finding 3.3. While the presence of super-emitters is well established in the methane literature, evidence also suggests that super-emitters exist for some fugitive-type VOCs from certain source types and events. These source types include pneumatic controllers, compressor blowdowns, high-emitting flares, and aging wells.

Finding 3.4. The reasons why sources become super-emitters is not entirely clear, although abnormal operational conditions or malfunctions have been identified in some cases. Some evidence suggests that the likelihood of super-emitter events may be more common in idle or stand-by infrastructure, though further research is needed to confirm this observation. Many of the early sampling studies were likely not designed to fully characterize super-emitters and contributed to discrepancies in emissions estimate methods. Bottom-up analyses of individual components typically found much lower emission rates than top-down studies measuring atmospheric methane concentrations, indicating gaps in the bottom-up estimates.

Finding 3.5. While super-emitters are primarily characterized by their emissions relative to other sources, there is strong evidence of substantial temporal intermittency in emission rates, as observed from a few high-frequency sampling

PSE

campaigns. Some evidence also suggests that the large majority (~90%)[7] of super-emitter events occur as one-offs or at relatively low frequency. While it is unclear if these types of low frequency events indicate that the leak was repaired, it has also been documented that many super-emitter events are easily fixable and preventable issues such as with unlit flares or open thief hatches.

Finding 3.6. Mitigation of super-emitters is not incentivized within the context of the current U.S. EPA Greenhouse Gas Inventory. This is largely due to how emissions factors are used to calculate methane emissions and the use of "regulatory reductions" instead of actual measurements to demonstrate reductions. This has major implications for addressing emissions from super-emitter-type events.

Finding 3.7. While super-emitters are a clear problem for climate and contribute to human health hazards, risks, and impacts, relatively smaller individual emission sources can also cause disproportionate impacts if they are close to and upwind of human populations. Relatedly, large-scale facilities with a multitude of emissions points and source-types may simultaneously exhibit both routine and abnormal emissions events and should be high-priority candidates for deployment of on-site continuous methane and HDAP monitoring systems.

Recommendation 3.1. Efforts to prevent, identify, and mitigate super-emitting sites and equipment should continue to be a top priority due to their disproportionate impact, omnipresence, and elusiveness. However, given the urgency to reduce emissions, focusing solely on equipment control systems or aerial survey technologies will likely have limited effectiveness in preventing and mitigating super-emitter events. A more encompassing approach should entail installed equipment controls, routine preventative maintenance, and multi-platform emissions monitoring to inform a more targeted LDAR program. High-resolution satellites, commercial airborne remote sensing systems, and continuous on-site methane and HDAP monitoring systems can all improve detection capabilities; however, prevention and mitigation steps require other types of management and operational practices to be in place in order to take advantage of these emerging technologies.

Recommendation 3.2. While further component-level root cause study would be valuable, volunteer bias in on-site methane surveys is an unavoidable confounding factor that undoubtedly limits generalizability of findings. Aerial survey technologies and airborne methane remote sensing systems can augment further study of super-emitters and should continue research and development in detecting, quantifying, and apportioning emissions throughout the supply chain.

Recommendation 3.3. If super-emitters are to be explicitly targeted for mitigation, the term "super-emitters" needs to be standardized, whether on a proportional loss or absolute loss basis.

---

7   Cusworth et al. (2021)

PSE

Theme #4. Over the past decade, the scientific understanding of methane emissions from oil and gas has increased substantially. There is now unequivocal evidence to support swift and agg essive reductions in methane emissions to avoid shorter-term global warming.

Finding 4.1. Using the 20-year global warming potential of methane, dry gas production-normalized emission rates ranging from 2.4-3.9%[8] for methane were calculated to be the breakeven point where the climate impacts of natural gas equal those of coal. The production-normalized methane emission rate for the entire supply chain is estimated to be 2.3%,[9] with individual regions ranging from 0.4% to 17%.[10] Although many studies are close to or lower than the calculated breakeven point, uncertainty in emission estimates remains high. Methane emissions from the upstream sector represent the largest fraction of oil and gas-related methane emissions. As such, uncertainties associated with upstream measurements dominate overall uncertainties for the oil and gas sector as a whole.

Finding 4.2. A persistent issue throughout the past 10 years of scientific study was the lack of agreement between bottom-up and top-down derived methane emission estimates. Studies suggest that component-level, bottom-up estimates routinely underestimate emissions by a factor of 1.6 to 2, compared to top-down studies and site-level bottom-up studies. A few studies have estimated emissions lower than the U.S. EPA Greenhouse Gas Inventory, but these studies were in the minority and were limited to specific parts of the oil and gas supply chain. The disagreement between bottom-up and top-down studies is a result of both the inaccuracy of inventories and the measurement uncertainties in atmospheric studies. Given the generalized assumptions required in bottom-up approaches and the highly variable (and rapidly changing) oil and natural gas industry in the late 2000s, the level of disagreement observed is not surprising.

Finding 4.3. Uncertainties in methane emissions can be expressed relative to the individual sector or source-type alone or relative to the entire supply chain. According to the studies to date, emissions uncertainties relative to individual sector or source-type estimates are most poorly constrained for downstream behind-the-meter sources, upstream orphaned and abandoned wells and other abandoned infrastructure, upstream gathering facility flares, upstream gathering pipelines, dehydrators, still/reboil vents, acid gas removal units, and pneumatic controllers. Emissions uncertainties relative to total estimated supply chain emissions indicate that more information is needed for gathering and boosting stations, pneumatic controllers, and liquid storage tanks. Moreover, important uncertainties remain for emissions between production basins as certain basins have been much more heavily studied compared to others.

---

8   Hong & Howarth (2016); Ren et al. (2019); Sanchez & Mays (2015). Analyses by Hong and Howarth (2016) and Sanchez and Mays (2015) are based on generating electricity from natural gas versus coal.

9   Alvarez et al. (2018)

10 See Table 3.1

PSE

Finding 4.4. Relative to spatial variability, less is known related to temporal variability of emissions. Most emissions characterization studies have taken place over very short durations (e.g., 2-4 weeks). Shorter duration studies have clear limitations related to generalizability of findings and representativeness of study sample measures, particularly for some source types such as pneumatic controllers, flares, and power plants. Given the large degree of heterogeneity across the North American oil and gas landscape, the presence of long-tailed distributions (i.e., super-emitters), and other temporal variation (e.g., persistent vs. intermittent), this limitation warrants particular attention. The few studies that have assessed temporal aspects of emissions have been particularly informative.

Finding 4.5. Our systematic review indicates that research in the last three years is rapidly accelerating to close major research gaps on methane emissions from oil and gas systems. We identified more methane studies published in 2019 than any of the previous four years. This trajectory is likely to continue. From mid-2020 through August 2021 alone, publication rates were higher than at any other time in the past decade, reflecting the fast-moving pace of the science. While we did not fully review these studies, we incorporated information from select high-priority studies that were identified via external peer review. For example, the first two studies of methane emissions in Mexico were published in the past year (a notable research gap).

Recommendation 4.1. Given current scientific understanding, effective methane and HDAP emissions control technologies and approaches exist today and should be swiftly implemen ed. Additional scientific study should continue where needed but should not delay deployment of emission controls and monitoring systems for source types that have been well characterized. Rather, the focus of additional study should include targeted campaigns that test the effectiveness of control technologies and validate emerging sensing technologies. These data are critical for supporting policies around continuous monitoring that can underpin a move towards performance-based emissions targets.

Recommendation 4.2. Current component-level bottom-up inventory methods would be improved by incorporating new emissions data from recent studies and revising bottom-up emission modelling approaches. Demonstrating the difference between persistent and intermittent emissions sources and their relative emissions contributions can help constrain emissions discrepancies and design more targeted mitigation approaches.

Recommendation 4.3. In the long term, the most effective strategy to reduce emissions from the oil and natural gas sector is to reduce the overall development, transmission, and use of oil and natural gas. However, as society moves away from oil and natural gas for energy use, the transition must be managed in a way that does not result in degraded operation and maintenance practices and potentially increased emissions.

PSE

Conclusion

Feasible approaches to reduce methane emissions have existed for years and could more than halve future anthropogenic methane emissions by 2030 (Clean Air Task Force, 2021). However, widespread deployment of these systems remains a key challenge. The weight of the scientific literature to date indicates there is significant variability in the magnitude and ratios of methane and HDAP emissions across geographic, temporal, and corporate spaces. This degree of variability has challenged the usefulness of both activity-based methane emission inventories, which often rely on outdated emissions factors, and the science aimed at reconciling inventory estimates with real-world observations. While studying emissions from the oil and natural gas sector has its challenges, our analysis found that research in the last five years is rapidly closing major research gaps on both methane and HDAPs and that this trajectory is likely to continue. The likelihood of co-emissions of methane and HDAPs in many, but not all, parts of the oil and gas sector likely means that the cost-benefits of methane mitigation are underestimated, particularly for those sources in proximity to human populations.

In the United States, there is a long history of ambient air quality monitoring and stationary source air monitoring for a multitude of criteria and hazardous air pollutants. These systems have been a critical pillar in identifying HDAP emissions and have played a fundamental role in reducing public health harms from exposures to HDAPs across sectors. Deploying a similar source-focused stationary monitoring network is likely not feasible given the disparate oil and gas landscape. However, centralized facilities located near population centers should exhibit on-site, stationary continuous monitoring systems capable of quantifying both methane and HDAP emissions. On-site continuous monitoring systems can fill data needs for air quality modelling and health impact analysis and would support the development of measurement-based inventories. Such inventories could also support performance-based regulations that would better incentivize prevention of super-emitter type events that currently make up a significant portion of total emissions.

Central to effective air quality monitoring for HDAP emissions from oil and gas systems is improving knowledge of what to monitor for. Our review found that there are two major source-types of air pollutants: (1) fugitive emissions (leaks and venting of non-combusted gases), which emit relatively high proportions of methane and VOCs; and (2) combustion emissions that emit less methane and possibly fewer VOCs, but higher levels of NO_x, CO, and PM. The emission controls that likely provide the greatest opportunity for climate and public health benefits are tankless designs or vapor recovery units on liquid storage tanks, electrification of compressor engines, preventing the need for flaring, and replacement of high-bleed pneumatic devices with zero-bleed devices. Much less is known about the potential health harms associated with downstream fugitive and combustion emissions.

Focusing on emissions controls and monitoring or aerial sensing technologies alone will likely have varied effectiveness in ultimately preventing and mitigating emissions. A more encompassing approach entails installed maximum achievable equipment controls, routine

PSE

preventative maintenance practices, and multi-platform emissions monitoring to inform a more targeted LDAR program. High-resolution satellites, commercial airborne remote sensing systems, and continuous on-site methane and HDAP monitoring systems can all improve detection capabilities and should continue to be deployed; however, prevention and mitigation activities require other types of operational practices to be in place in order to take advantage of these emerging technologies.

Given the powerful, shorter-term global warming potential of methane–and the fact that no study we reviewed identified methane emissions too small to be of climate concern– it is imperative that methane prevention, detection, and mitigation strategies be swiftly and aggressively deployed. The prospect of future research to refine our understanding of methane and HDAP emission estimates should not preempt swift a tion, particularly in areas where oil and gas systems are in proximity to human populations.

Lastly, it should be noted that the most effective strategy to rapidly reduce air pollutant emissions from the oil and natural gas sector is to reduce the overall development, transmission, and use of oil and natural gas. However, as society moves away from oil and natural gas for energy use, the transition must be managed in a way that does not result in degraded operation and maintenance practices and potentially increased emissions.

PSE

References

Alvarez, R. A., Zavala-Araiza, D., Lyon, D. R., Allen, D. T., Barkley, Z. R., Brandt, A. R., Davis, K. J., Herndon, S. C., Jacob, D. J., Karion, A., Kort, E. A., Lamb, B. K., Lauvaux, T., Maasakkers, J. D., Marchese, A. J., Omara, M., Pacala, S. W., Peischl, J., Robinson, A. L., … Hamburg, S. P. (2018). Assessment of methane emissions from the U.S. oil and gas supply chain. *Science*, 361(6398), 186–188. https://doi.org/10.1126/science.aar7204

Cusworth, D. H., Duren, R. M., Thorpe, A. K., Olson-Duvall, W., Heckler, J., Chapman, J. W., Eastwood, M. L., Helmlinger, M. C., Green, R. O., Asner, G. P., & others. (2021). Intermittency of Large Methane Emitters in the Permian Basin. *Environmental Science & Technology Letters*, 8(7), 567–573. https://doi.org/10.1021/acs.estlett.1c00173.

Hong, B., & Howarth, R. W. (2016). Greenhouse gas emissions from domestic hot water: Heat pumps compared to most commonly used systems. *Energy Science & Engineering*, 4(2), 123–133. https://doi.org/10.1002/ese3.112

Ren, X., Hall, D. L., Vinciguerra, T., Benish, S. E., Stratton, P. R., Ahn, D., Hansford, J. R., Cohen, M. D., Sahu, S., He, H., Grimes, C., Fuentes, J. D., Shepson, P. B., Salawitch, R. J., Ehrman, S. H., & Dickerson, R. R. (2019). Methane Emissions from the Marcellus Shale in Southwestern Pennsylvania and Northern West Virginia Based on Airborne Measurements. *Journal of Geophysical Research: Atmospheres*, 124(3), 1862–1878. https://doi.org/10.1029/2018JD029690

Sanchez, N., & Mays, D. C. (2015). Effect of methane leakage on the greenhouse gas footprint of electricity generation. *Climatic Change*, 133(2), 169–178. https://doi.org/10.1007/s10584-015-1471-6

UNEP & CCAC. (2021). Global Methane Assessment (full report). Climate & Clean Air Coalition. https://www.ccacoalition.org/en/resources/global-methane-assessment-full-report

UNEP & CCAC. (2021). Global Methane Assessment (full report). Climate & Clean Air Coalition. https://www.ccacoalition.org/en/resources/global-methane-assessment-full-report

U.S. EPA (United States Environmental Protection Agency). (2020). Regulatory Impact Analysis for the Review and Reconsideration of the Oil and Natural Gas Sector Emission Standards for New, Reconstructed, and Modified Sources, (EPA-452/R-20-004). https://www.epa.gov/sites/production/files/2020-08/documents/oil_and_natural_gas_nsps_review_and_reconsideration_final_ria.pdf

U.S. EPA (United States Environmental Protection Agency). (2021). Inventory of U.S. Greenhouse Gas Emissions and Sinks, 1990-2019: Executive Summary (EPA 430-R-21-005). https://www.epa.gov/sites/production/files/2021-04/documents/us-ghg-inventory-2021-chapter-executive-summary.pdf

PSE

Table of Contents

PSE

PSE

List of Figures

PSE

List of Tables

PSE

1.0   Introduction and Approach

The global atmospheric concentration of methane — a greenhouse gas that captures 86 times more heat than carbon dioxide over the 20-year time horizon — is now higher than at any time in the past 800,000 years. Petroleum and natural gas systems are responsible for an estimated one-third of all anthropogenic methane emissions in the United States, holding direct consequences for the climate. Health-damaging air pollutants (HDAPs) — particulates and gaseous volatile or semi-volatile compounds that are directly hazardous to human health — are also emitted from the oil and gas sector. While methane emissions contribute to climate change on the global scale, HDAP emissions impact air quality and human health on local and regional scales.

A sizable body of scientific literature indicates that to avoid the worst effects of climate change, methane emissions from oil and gas are both underestimated in standard inventories and must be reduced by almost half during the next decade. An adjacent body of scientific literature shows that populations near oil and gas development are exposed to greater health hazards, risks, and impacts compared to those living farther away.

While climate science supports the profound urgency to reduce methane emissions from oil and gas systems, there is no systematic scientific review of the opportunities and challenges of characterizing and mitigating methane emissions in tandem with HDAPs across the upstream, midstream, and downstream oil and gas sectors.

By failing to carefully characterize the sources, monitoring, and control options of these classes of pollutants together, solutions designed to mitigate emissions of one class may not adequately control the other and may fail to maximize the climate and health co-benefits of these approaches to emission identification and control.

To better understand the overlap in sources of methane and HDAPs and approaches to prevent, detect, and mitigate emissions, we conducted a systematic review of scientific peer-reviewed literature published between January 2015 and August 2020. Our review focused on primary methane and HDAP data collection efforts pertinent to emissions throughout the oil and gas supply chain, divided into upstream (development and production), midstream (processing and transmission), and downstream (distribution and end-use) sectors, with the following objectives:

- **Summarize Key Themes and Findings:** From our systematic review, we aimed to equip researchers, communities, and decisionmakers with a clear, concise, and informative guide to the state of the science, current research gaps, and priorities on issues of methane and HDAP emissions across the oil and gas sector.
- **Highlight Research and Policy Recommendations:** Based on the findings of the systematic review, we provided recommendations that can be incorporated into

PSE

actionable climate policy that simultaneously reduces methane emissions and protects public health.

We organized this report into four primary sections:

- In **Section 1**, we introduce our systematic review criteria and methods.
- In **Section 2**, we present the results and publication trends of our systematic review, spanning all oil and natural gas supply chain sectors and covering all studies that met the inclusion criteria.
- In **Sections 3–5**, we present the results of the systematic literature review separately for methane (Section 3), HDAPs (Section 4), and methane and HDAPs jointly (Section 5). These sections are each organized by the three overarching supply chain sectors (i.e., upstream, midstream, and downstream) and where applicable, individual source types are highlighted and discussed in further detail.
- In **Appendix A**, we include a section that examines the understandings, uncertainties, and research gaps that directly preceded 2015–2020. We did this by extracting the major themes within the 2010–2014 literature, with a specific focus on research challenges and limitations, research gaps, and research and policy recommendations.
- In **Appendix B**, we provide additional information on our search methodology and systematic review criteria.

1.1. Approach

We adopted the preferred reporting items for systematic reviews and meta-analyses (PRISMA) framework. PRISMA[1] provides a reproducible set of protocol standards for summarizing and synthesizing primary research and has been utilized previously in the oil and gas air pollution space by PSE and colleagues.[2]

To identify peer-reviewed articles related to emissions from the North American oil and natural gas supply chain, we established inclusion/exclusion criteria to select articles that performed primary, emissions-related research in North America between 2015–2020. All papers reviewed were available from Web of Science and PSE's managed literature database, Repository for Oil and Gas Energy Research (ROGER). Across both methane and HDAPs practice areas, we generated six separate Boolean keyword strings for the six combinations of methane/HDAPs and three supply chain sectors (see Appendix B). Numerous iterations of keyword searches

---

[1] Moher, David, et al. "Preferred reporting items for systematic review and meta-analysis protocols (PRISMA-P) 2015 statement." *Systematic reviews* 4.1 (2015): 1.

[2] Garcia-Gonzales, Diane A., et al. "Hazardous air pollutants associated with upstream oil and natural gas development: a critical synthesis of current peer-reviewed literature." *Annual review of public health* 40 (2019): 283-304.

PSE

were performed at the outset to balance both the probability of capturing articles of interest with the time-intensive task of manual reviewing. We then manually screened titles and abstracts for relevance to inclusion/exclusion criteria. The results of this step-wise process are described in Section 2. In addition to this full literature review process, we also included a preliminary count of studies published between August 2020 to August 2021 to demonstrate the pace at which the field is moving.

To provide both external review and as a supplemental resource, we compared the PRISMA results from Web of Science to PSE's ROGER citation database in October 2020. ROGER is a near-exhaustive, publicly-accessible citation database that focuses on shale and tight gas development and is independently maintained by PSE Healthy Energy. Although ROGER includes a broader concentration of more general oil and gas-related research, we found a 55% overlap of citations between ROGER and PRISMA.

Given the size and scope of scholarly work on oil- and gas-associated air pollutant emissions in North America since 2015, we structured our overall review into six separate systematic reviews (presented in Sections 3–5). For each of these six focus areas, we identified and synthesized important findings and policy recommendations, as well as data gaps that could be filled with more scientific measurement or additional science-policy synthesis efforts and regulatory action. Our synthesis of the peer-reviewed literature in this report aims to inform and support researchers, policymakers, regulators, and other stakeholders working towards policies aimed at realizing integrated climate and health benefits across the oil and gas supply chain. These were categorized along the three major oil and gas supply-chain sectors (upstream, midstream, downstream) and by emission type (methane or HDAP).

1.1.1. Study inclusion and exclusion criteria

We implemented several criteria to filter peer-reviewed articles. We initially included articles that were published between January 2015 to August 2020. Only English-written studies performed in North America (i.e., United States, Canada, or Mexico) were included. We also included government reports, critical reviews, and some studies detailing novel technologies or novel measurement/estimation approaches when warranted or heavily cited. We actively excluded any potential sources of methane that were not from a fossil origin (i.e., wetlands, dairies, landfills, wastewater, forests, soil) and also excluded liquefied natural gas (LNG) to narrow the scope on emissions from the supply, distribution, and stationary combustion phases.

1.1.2. Defining the oil and gas supply chain

Oil and gas drilling has occurred in the United States since the mid-19th century. During this time, a complex system of operational and equipment components has been developed to support the delivery of natural gas to end-users, each of which may be a potential source of methane leakage. The main components include subsurface wells; hydrocarbon gathering and

PSE

production facilities; above ground and underground storage; and transmissions and distribution pipelines. These components generally fall into three sectors: upstream, midstream, and downstream. We considered emissions from oil from components associated with its production through refining but did not evaluate direct emissions of the burning of refined oil. For natural gas, we evaluated all components of the supply chain from production to end use but did not cover aspects of exploration (except for frac sand mining), petrochemical production, or the processes that support production and exportation of liquified natural gas.

The supply chain location of certain facilities, systems, and components is shown in Table 1.1. Notably, the systems that make up the oil and gas supply chain are numerous and there is likely overlap between these supply chain classifications. For example, while pneumatic devices and compressor stations can be located in many portions of the supply chain, we chose to discuss all pneumatic devices and compressors in the midstream sector for ease of reviewing literature. Similarly, storage tanks can be present throughout the supply chain, but are discussed solely in the upstream.

Table 1.1. Systems, equipment, and components that comprise the upstream, midstream, and downstream oil and gas supply chain.

Upstream Sources	Midstream Sources	Downstream Sources
Well pad and production	Inter- and intra-state pipelines	Distribution pipelines
Wells and casing head	All compressor types*	Metering and regulating stations*
Abandoned, idle, and orphaned wells	Underground storage facilities	Customer meters
Pumps	Transmission regulating stations	Gas stations
Storage tanks*	Gathering and boosting facilities	Power generation
Heater treaters	Metering and regulating stations	Refineries
Separators	Natural gas processing plants	End-user appliances
Dehydrators	Pneumatic devices*	Buildings
Oil and condensate separation		
Water removal		
Sulfur and CO_2 removal		
Fractionation of natural gas liquids and other processes such as CO_2 capture		

*For this report, sources may be located in multiple locations in the supply chain, but were discussed exclusively in the category as shown here for ease of review.

PSE

2.0   Results: Review of literature

In this section, we use our literature review to answer the question, "What trends and data gaps have been identified in the peer-reviewed literature since 2015?" Our review encompassed gathering and reviewing articles that advance the state of science as it pertains to methane and health-damaging air pollutants (HDAPs). We summarize counts of papers analyzed in each stage of our literature review here and comment on trends of the attributes of the studies.

2.1   Studies that met inclusion criteria for this review

We focused on studies that reported primary data (data directly collected by the study authors) and were published between January 2015 to August 2020. Other articles that did not report primary data collection but do contribute to the science — for example, through modeling, a literature review, or policy analysis — are mentioned here but are not included in most analyses. These articles which did not collect primary data are referenced in Sections 3–5 of this report during discussion of major themes from this body of scientific literature.

The primary source for our review was the Web of Science, supplemented by PSE's Repository for Oil and Gas Energy Research (ROGER) database.[1] We established the following workflow to search for and review articles, summarized in Figure 2.1:

1. We searched the Web of Science using Boolean search terms (listed in Appendix B) and described in Section 1. We selected 15,280 methane and 10,330 HDAP articles (25,610 total). In this stage, articles were selected from 2012 through the 2020 cutoff dates, which are listed in Table 2.1.
2. We manually read and screened the titles of all articles selected from the Web of Science, eliminating articles that were the most out of scope of our review. We kept 1,232 methane-related titles and 935 HDAP-related titles.
3. We manually read and screened abstracts from the selected titles, aiming to eliminate articles identified as out of scope of our review. We marked abstracts as either "yes" or "maybe" depending on how closely they aligned with our objectives. An additional internal expert review was conducted on these abstracts to select final studies for inclusion in the PRISMA, erring on the side of caution and selecting the article for full review.
4. At this time, additional articles from the ROGER database were added to the final study candidates.

---

[1] PSE Healthy Energy. Repository for Oil and Gas Energy Research (ROGER).
https://www.psehealthyenergy.org/our-work/shale-gas-research-library/

PSE

In total, 25,610 scholarly titles (15,280 methane and 10,330 HDAP) from Web of Science were manually assessed against our inclusion criteria in multiple stages to produce the final set of studies. In the first stage, we read every title individually and evaluated whether the paper was a possible candidate for review. We kept 1,232 methane-related titles and 935 HDAP-related titles. In the second stage, we reviewed abstracts and marked them as either "yes" or "maybe" to determine whether they should be read in full. In total, 308 methane and 194 HDAP papers were marked as "yes," while 394 methane and 101 HDAP papers were marked as "maybe." At this time, additional articles from the ROGER database were added to the PRISMA. Results by emissions type and supply sector are shown in Table 2.1.

In this report, Sections 3-5 analyze the articles published between January 2015 and August 2020. The numbers that are presented in the figures in Section 2.2, "Description of final inclusion studies" and following sections, are all articles that were published in 2015 or later. At the end of this report, Appendix A analyzes major articles which "set the stage" for the time period of our review, namely articles which we identified which were published in 2014 or earlier.

Table 2.1. PRISMA results for 2012–2020 methane and HDAPs record searches. Note that at this stage, articles may appear in more than one category and may be counted multiple times in the "Total # of titles" row.

Category	Cutoff date for search	Total # of Titles	# "Yes" Titles	# "Yes" Abstracts	# "Maybe" Abstracts
Methane downstream	7/11/2020	3,277	427	72	158
Methane midstream	7/23/2020	1,512	270	46	133
Methane upstream	9/9/2020	10,491	535	190	103
HDAPs downstream	8/13/2020	1,813	389	37	89
HDAPs midstream	8/13/2020	4,087	348	53	5
HDAPs upstream	8/10/2020	4,430	198	104	7
TOTAL		25,610	2,167	502	495

PSE

Figure 2.1. Overall workflow for our review, including total numbers of articles reviewed at each stage in the process. Note: articles which report on both methane and HDAPs are counted individually in each category and therefore are double counted in overall totals.

---
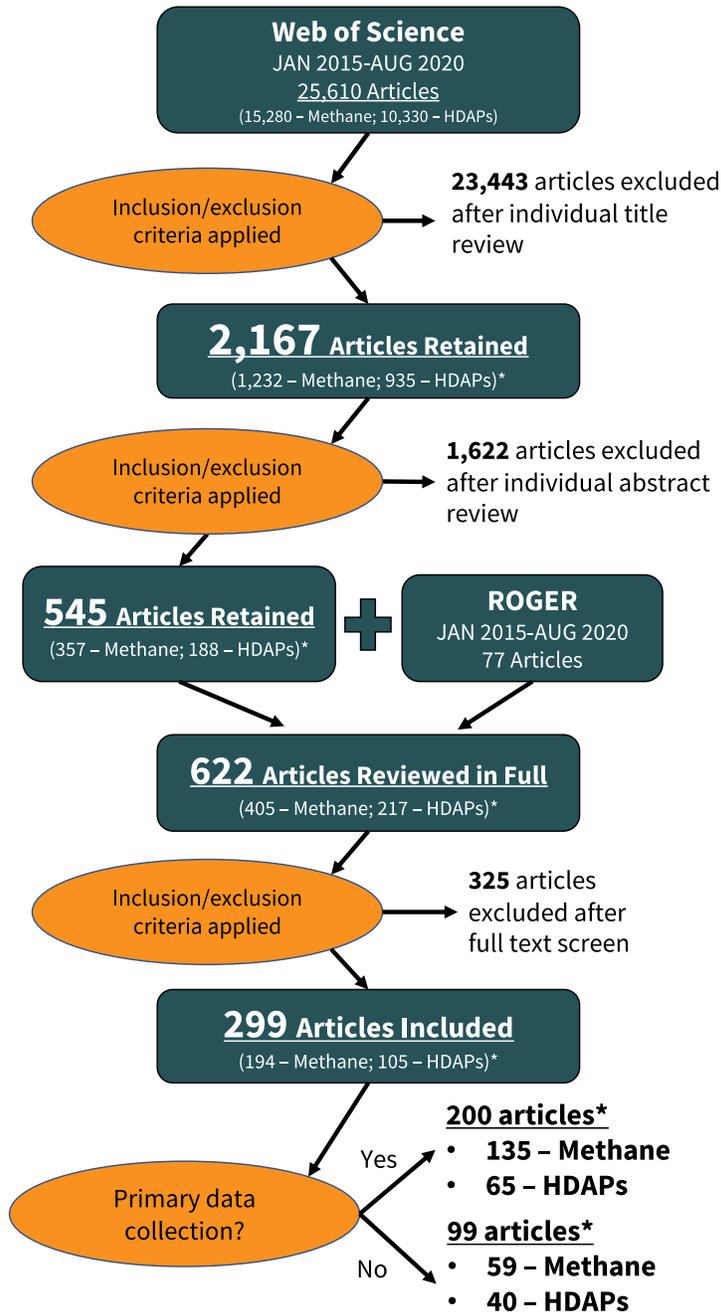
*Articles which report both methane and HDAPs are counted individually in each category and are double-counted in overall totals.

PSE

2.2   Description of final inclusion studies

From the final set of articles identified after reading abstracts from the Web of Science search and adding papers from our ROGER database, we reviewed every paper published since 2015 that fit our study criteria. In total, we reviewed 270 unique studies, including 165 methane studies and 76 HDAP studies. In addition, we evaluated 29 studies which we classified as both methane and HDAPs for their cross-cutting research and are double-counted in overall totals. For the 29 studies classified as both, each was reviewed once by the same person and the information was transcribed from one PRISMA to the other. Some articles were eliminated during review for being out of scope. Articles which were identified and reviewed which did not include primary data collection (i.e. policy analysis, modeling papers, methods papers, other literature reviews, etc.) are considered in other discussion sections of this report.

Among our studies which we reviewed in full, we examined a subset more closely which explicitly collected and reported primary data. In our review, 135 unique methane papers collected primary data, and 65 additional papers measured HDAPs with primary data collection (Figure 2.2). Of the papers that collected primary data, 20 measured both methane and HDAP emissions and were counted in each category. Sections 3-5 discuss the studies in their respective subsectors: upstream, midstream, and downstream. Therefore, we break up our results into the number of articles which collected data from either the upstream, midstream, or downstream sources (specifics of each of the subsectors is described in Section 1. Figure 2.2 shows the numbers of studies that reported emissions from each subsector, colored by the type of article. Because articles could report emissions from more than one subsector, the sum of upstream, midstream and downstream will exceed the count of unique article count.

Nearly every article reviewed which collected primary data was a research article. We found that there were about twice the number of upstream methane articles as midstream or downstream methane, and about eight times the number of upstream HDAP articles as midstream or downstream HDAP.
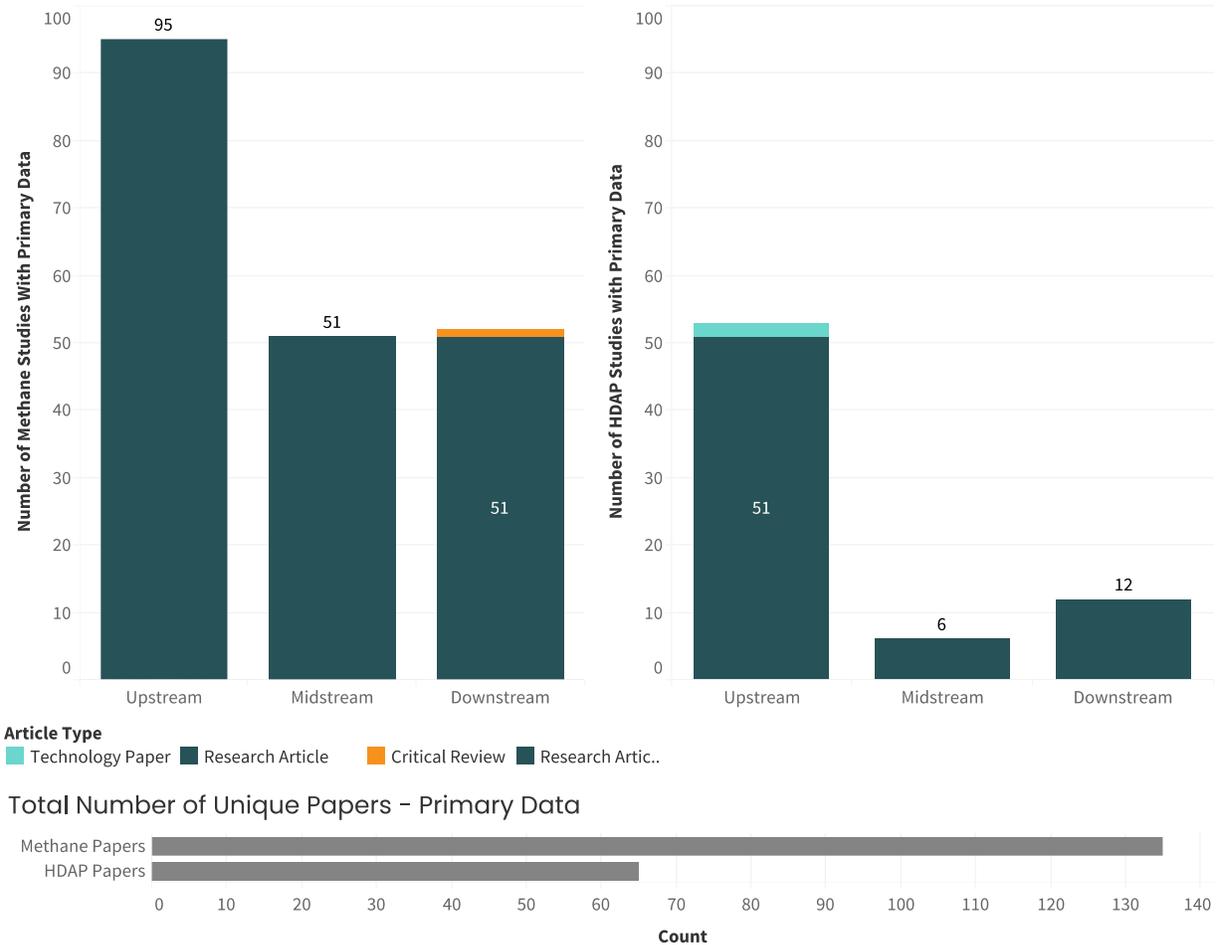
Figure 2.2. Total counts of methane and HDAP papers that included primary data, as well as the number of papers that we identified as measuring emissions in various subsectors. Dark blue indicates research articles, orange indicates critical reviews, and light blue indicates technology papers.

2.3 Characteristics of methane studies

In this section, we describe the 138 studies that collected primary data on methane emissions.

Time series

We reviewed papers published between January 2015 to August 2020. Figure 2.3 shows the time series breakdown of the number of papers we identified that collected primary data. Note that because of our mid-2020 cutoff date, the results from 2020 are incomplete and should not be considered a complete year. The time series analysis revealed a complete gap in 2018 for midstream-specific papers, and only one midstream paper published in 2019 and 2020,

respectively. Overall, 2018 was a low year for publications — the fewest publications were identified in in 2018 — although many more were published the following year.
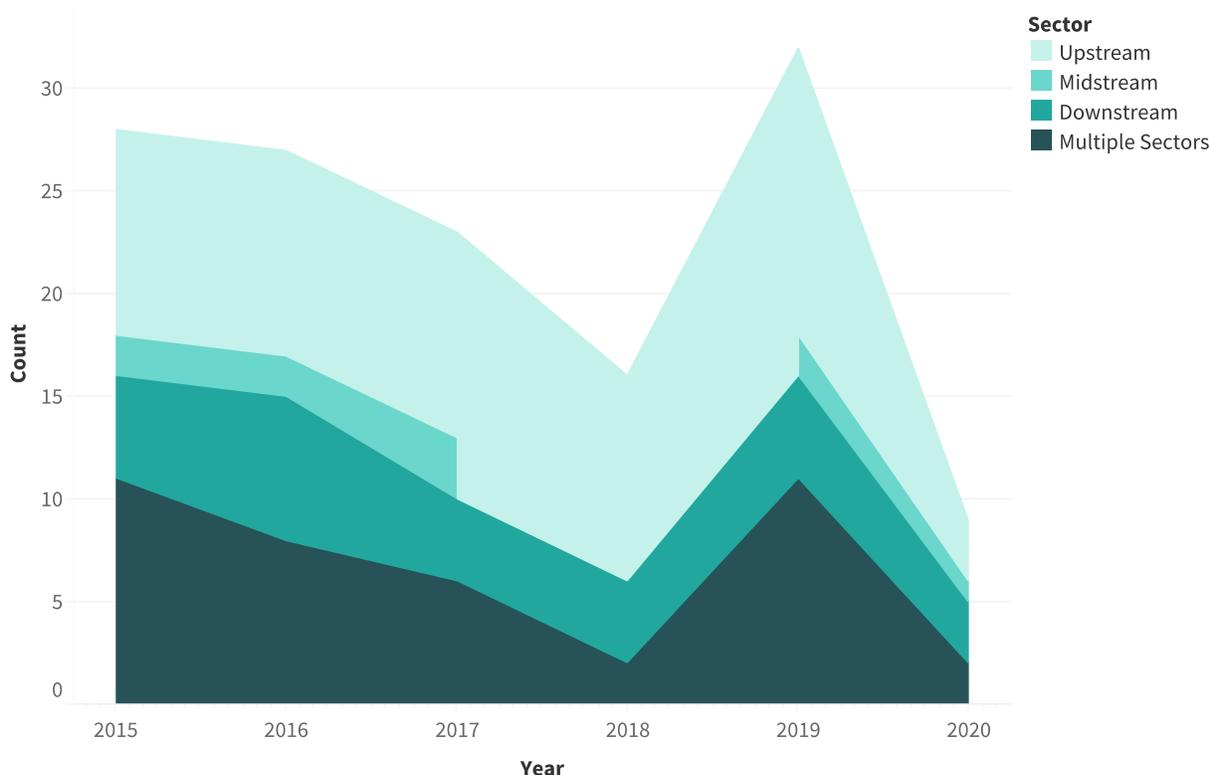
Figure 2.3. Time series of methane emissions papers published between 2015–2020. Note that 2020 only counts articles published from January-August, since August was the cut-off for our study. The bars are colored by the specific sector that was evaluated in the paper.

Reported methods of data collection

We found that a nearly equal number of studies collected samples through ground-based measurements, vehicle-based measurements, or airborne measurements (Figure 2.4. Ground-based measurements refer to data that was collected using equipment directly at the site of the emissions, either through tracer-based quantification, chamber-based, whole air sampling, or another similar method. Vehicle-based measurements were taken from a car, typically with a real-time analyzer that quantified methane concentrations outside the vehicle. In some instances, the vehicle was able to quantify the emissions using a mobile plume integration method. Airborne measurements quantified emissions by flying over a site, either by flying in a spiral over an emission source or by flying in passes downwind of a basin. Airborne measurements were the most popular choice for studies measuring emissions from multiple

sectors, given the versatility of the measurement technique. A smaller number of studies utilized towers.
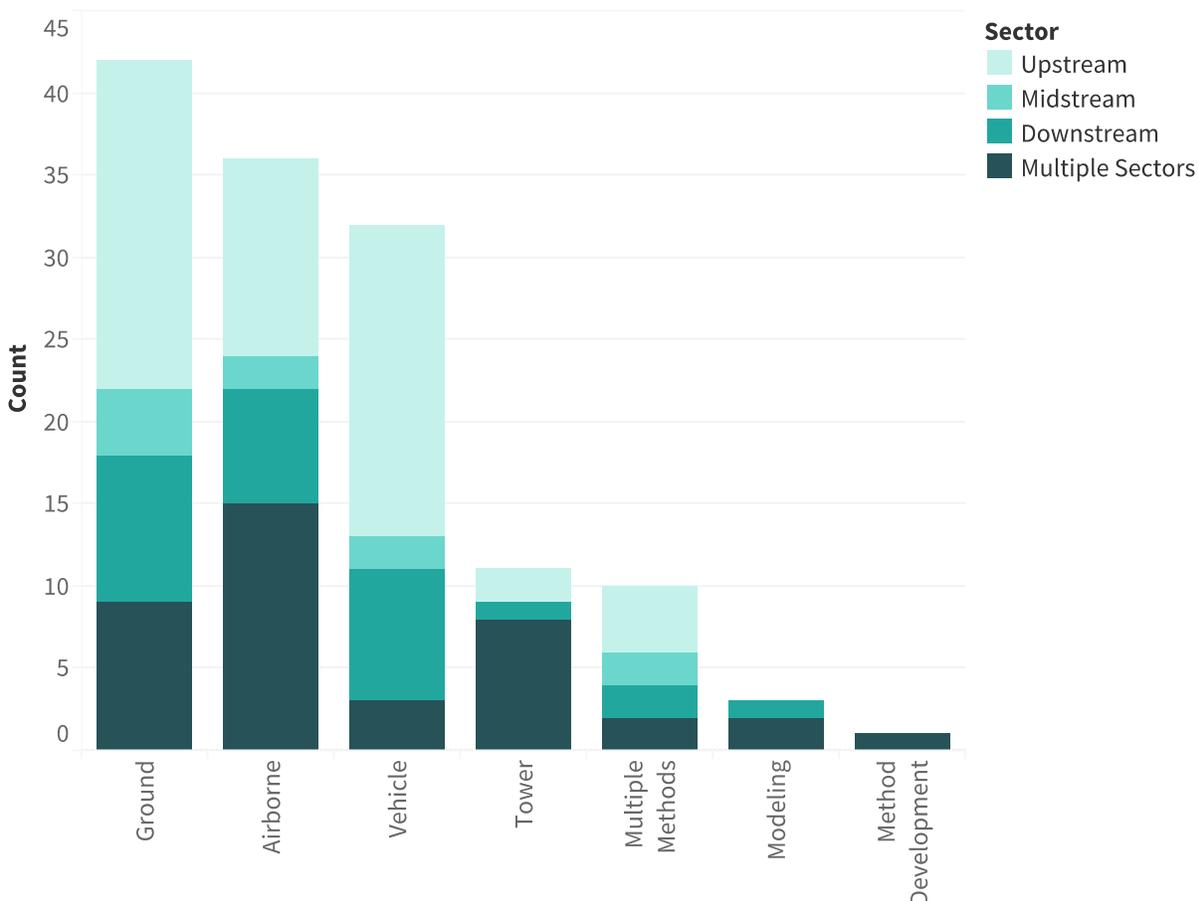
Figure 2.4. Count of number of papers which collected primary data using various measurement techniques. The bars are colored by the specific sector the paper focused on.

Top-down vs. bottom-up scaling methods

Of those reviewed, 70 papers that collected primary data also scaled the emissions into regional emission factors. We generally classified these scaling methods into a top-down or bottom-up (or both, the difference of which is explained in Section 1. We found that for methane papers, nearly the same number of papers – 31 vs. 33 – scaled their results using a bottom-up approach as did those using a top-down approach, respectively. Four additional studies used both methods (Figure 2.5). Generally, we found that top-down approaches were more effective at simultaneously measuring multiple sectors, while bottom-up approaches were more useful for measuring upstream and midstream sectors, likely because of the ability of a bottom-up approach to measure component-level emissions.
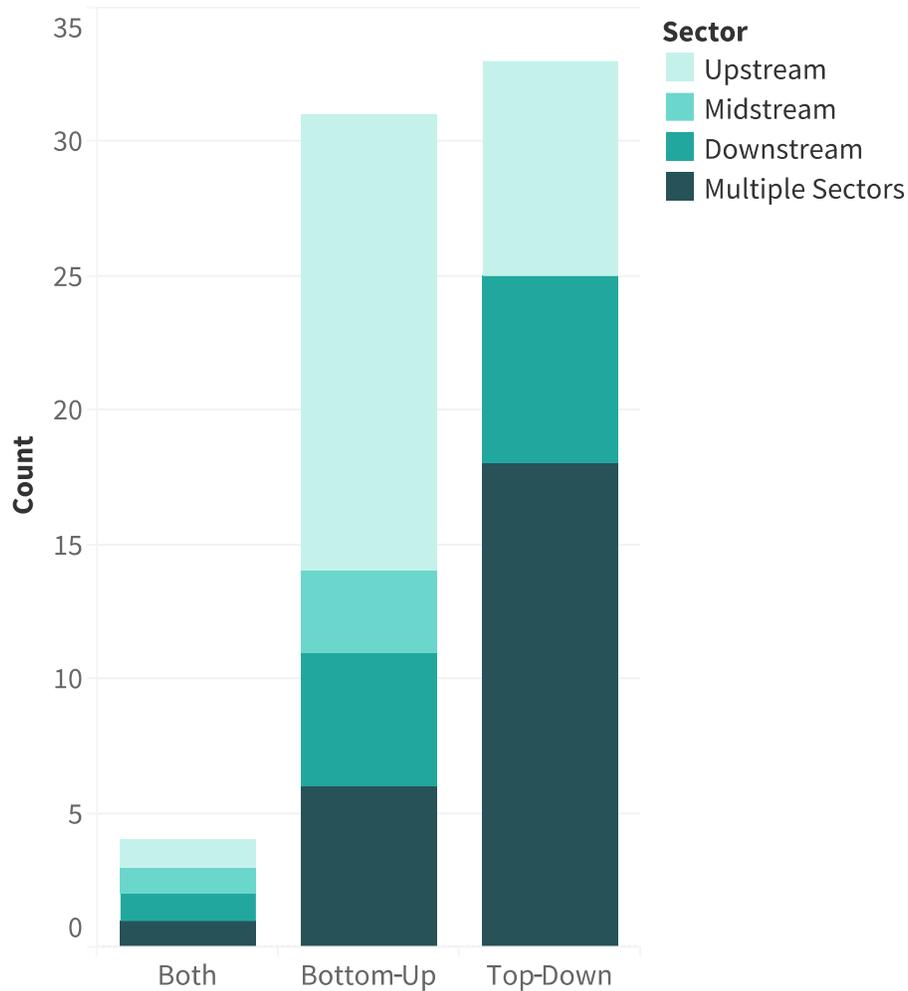
Figure 2.5. Top-down vs. bottom-up approaches in studies that scaled measurements, considering only studies which collected primary data and scaled their measurements. Bars are colored by the specific sector that was measured in the study.

Measure of impact

Another metric included in our PRISMA was a subjective measure of impact that was assigned to each article by a reviewer. Figure 2.6 shows that the most articles with primary data collection were assigned a ranking of "medium" impact while comparatively less were marked "high" or "low." A similar distribution of upstream, midstream, and downstream articles was seen within each impact ranking level.
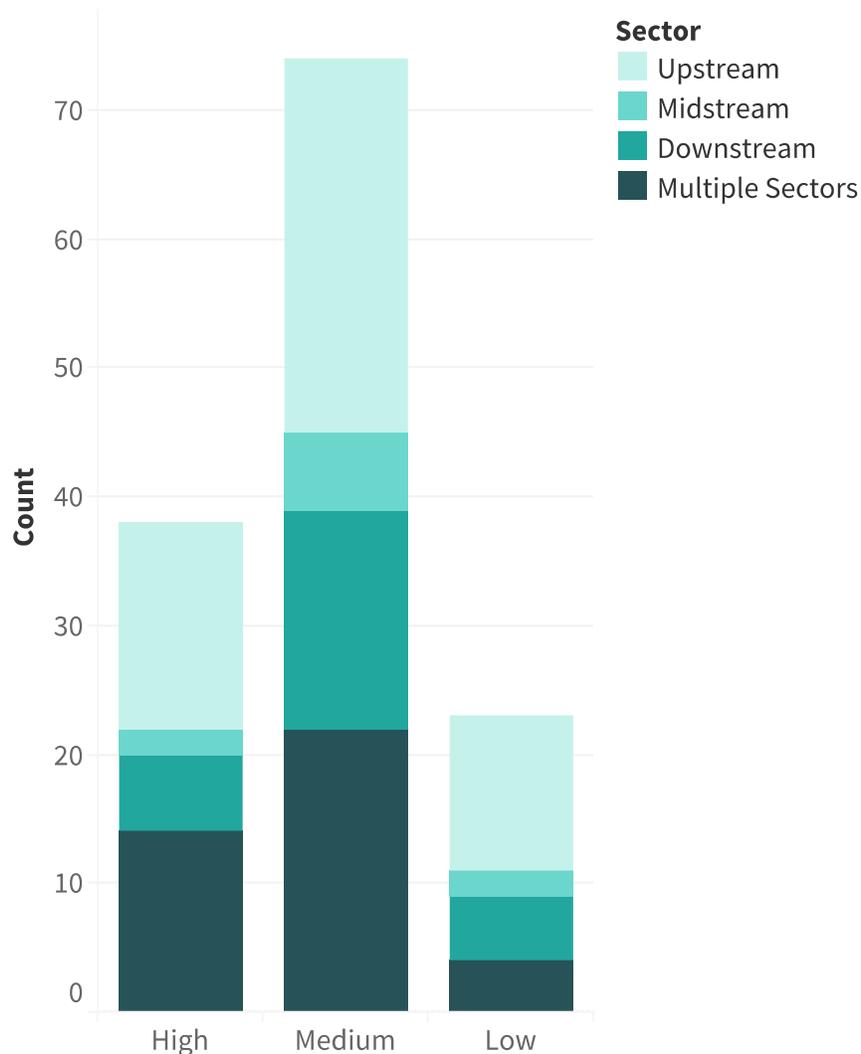
Figure 2.6. Assigned impact of methane articles with primary data collection in our review. Bars are colored by the sector in the supply chain that was measured in the study.

Hydrocarbon type

For studies that collected primary data on methane, we reported the hydrocarbon source type, selected between: oil; natural gas; oil and natural gas; oil and associated natural gas; compressed natural gas; and processed natural gas (Figure 2.7). We found that the vast majority of studies focused on measuring natural gas or oil and natural gas sources, with almost all oil and natural gas sources falling into upstream or multiple sectors categories.

PSE
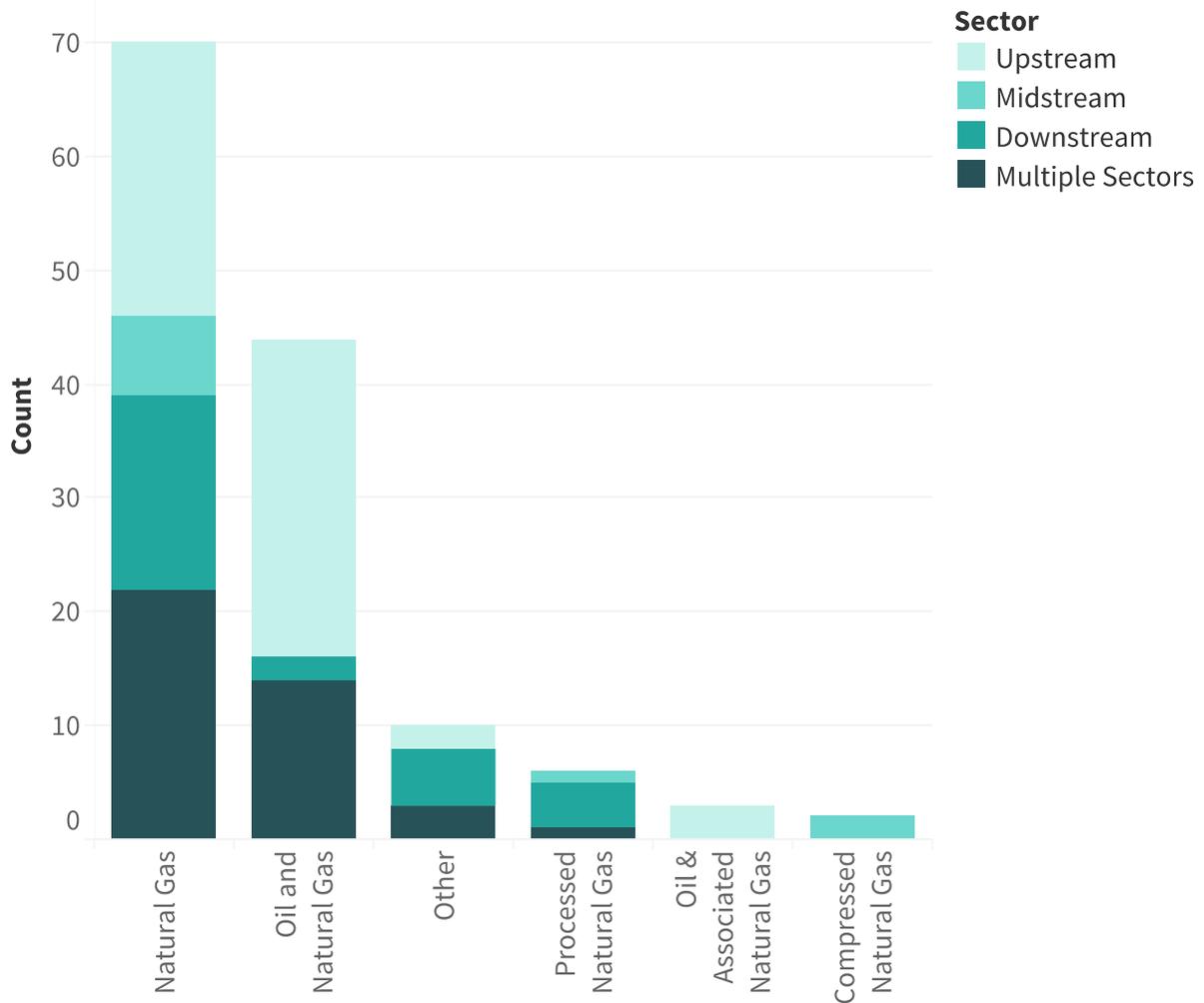
Figure 2.7. Sources of hydrocarbons in reviewed methane papers which collected primary data, colored by the sector of the oil and gas supply chain that was measured by the study.

Location of methane studies

For methane studies, we mapped the locations of studies with primary data collection based on the nine census zones designated by the U.S. Energy Information Administration (EIA) (Figure 2.8). We found that most studies collected data in the Western and South Central United States. Studies from Canada, and the Midwest and Rocky Mountain United States, are mostly upstream studies, highlighting a regional data gap.
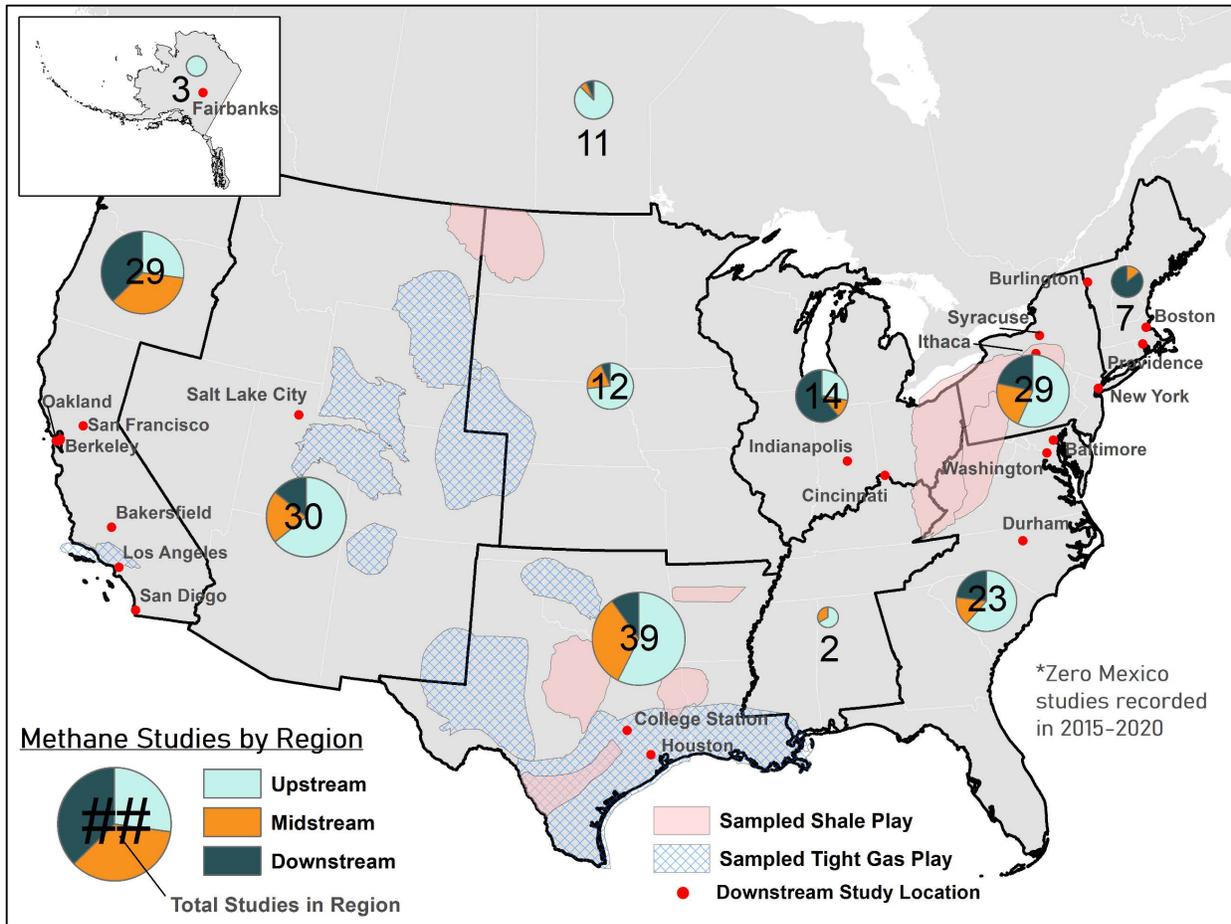
Figure 2.8. Map of methane study locations in North America which collected primary data, colored to represent the number of studies in each sector within each geographic region. Studies encompassing multiple regions are counted as a statistic in each region they sampled.

2.4 Characteristics of HDAP studies

In this section, we summarize the characteristics of the HDAP studies in our PRISMA. In total, we reviewed 105 papers that measured HDAP emissions from the oil and gas sector (including the 29 papers that also measured methane emissions).

We plotted the results over time for the 105 articles reviewed for HDAPs. The highest number of articles were published in 2016, with a relative decline in articles published since that year (Figure 2.9). Additionally, we note that 2019 had the lowest number of articles with primary data collection, suggesting a recent lack of data collection of air pollutants from the oil and gas industry. Note that 2020 is only a partial year as we cut off our review in August that year.
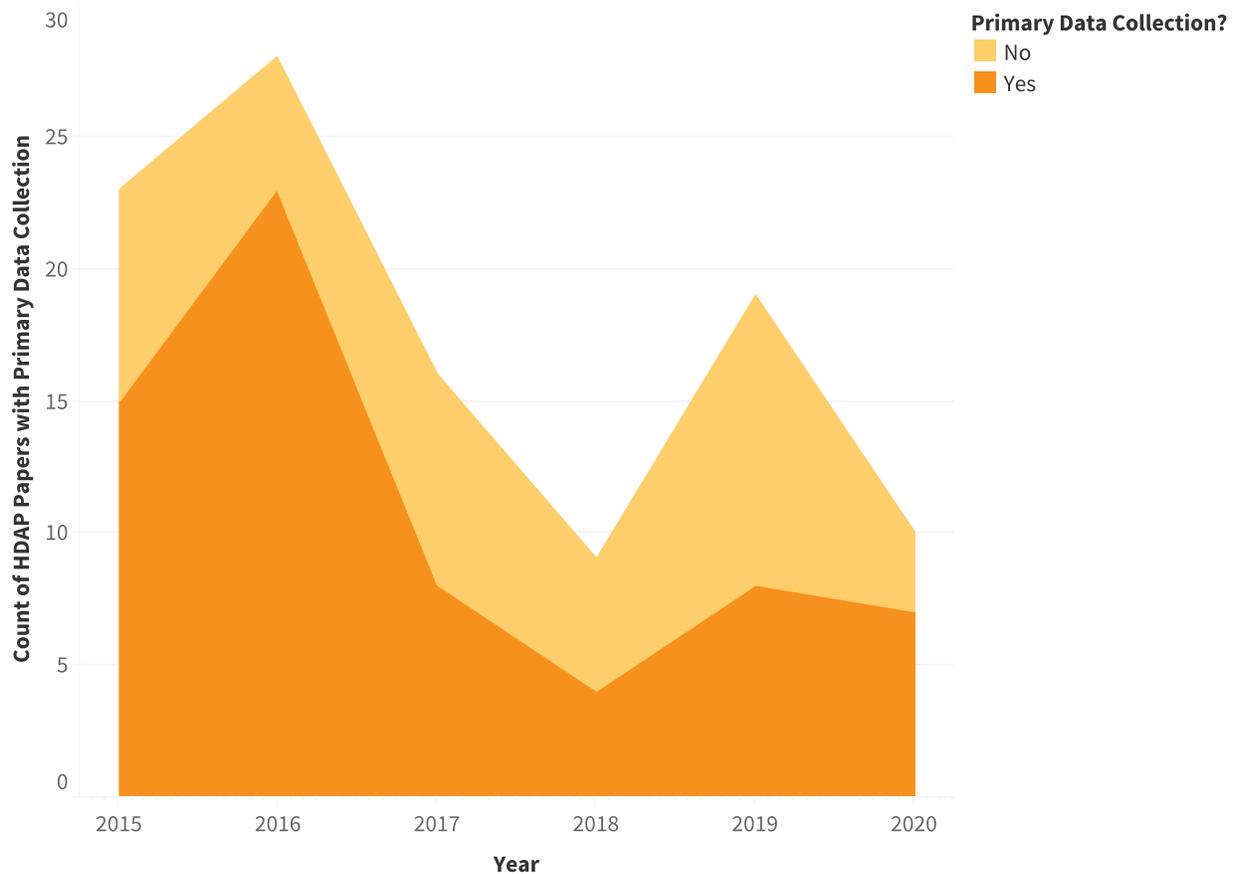
Figure 2.9. Count over time of health-damaging air pollutants from the oil and gas industry, from January 2015 to August 2020, colored by whether or not the article collected primary data.

Measurement technologies

We analyzed the articles that collected primary data (62 HDAP studies to determine the number of studies using various measurement technologies. More than half of the studies used a ground-based methodology, meaning that the researchers collected measurements using stationary technology on the ground. All but one downstream study utilized this methodology. The remaining studies used a combination of airborne, tower-based, vehicle, or modeling approaches (Figure 2.10).
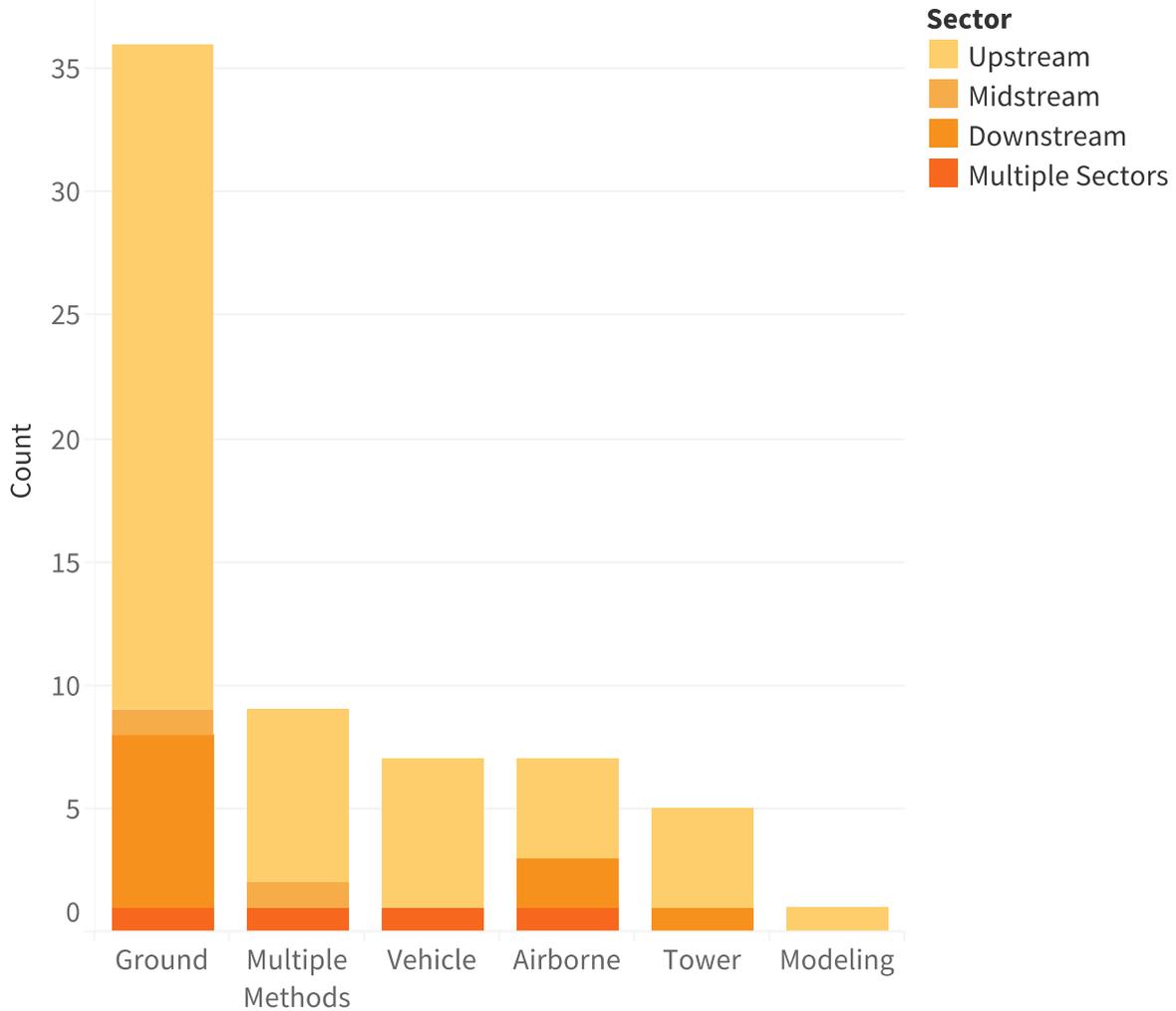
Figure 2.10. Breakdown of HDAP study sectors and methodology for studies that reviewed primary data.

Analysis of pollutants

From all HDAP studies, we identified various categories of pollutants that were analyzed or measured directly in the paper. Figure 2.11 shows the breakdown of pollutants by category within each upstream, midstream, or downstream sector. We identified whether each article measured one of the following categories of pollutants: NO_x (nitrogen oxides, such as NO and NO_2, BTEX (benzene, toluene, ethylbenzene, and xylene, VOCs (volatile organic compounds, ozone, $PM_{2.5}$ and PM_{10} (particulate matter, the number denotes the maximum size of the particle in microns, SO_x (sulfur oxides, radon, CO (carbon monoxide, black carbon, ammonia, H_2S (hydrogen sulfide, formaldehyde, silica, PAH (polycyclic aromatic hydrocarbon, aerosols and metals. Figure 2.12 shows the number of criteria pollutants (carbon monoxide, lead, ground-level ozone, particulate matter, nitrogen dioxide, and sulfur dioxide and hazardous

pollutants (pollutants known to cause cancer or other serious health impacts) measured from the studies identified in our review over time. The highest number of pollutants were measured in 2016, with relatively fewer papers published that measure these pollutants in 2017–2020. Figure 2.13 shows the total number of articles which measured hazardous and criteria pollutants, as well as the number of each of the individual pollutant categories.
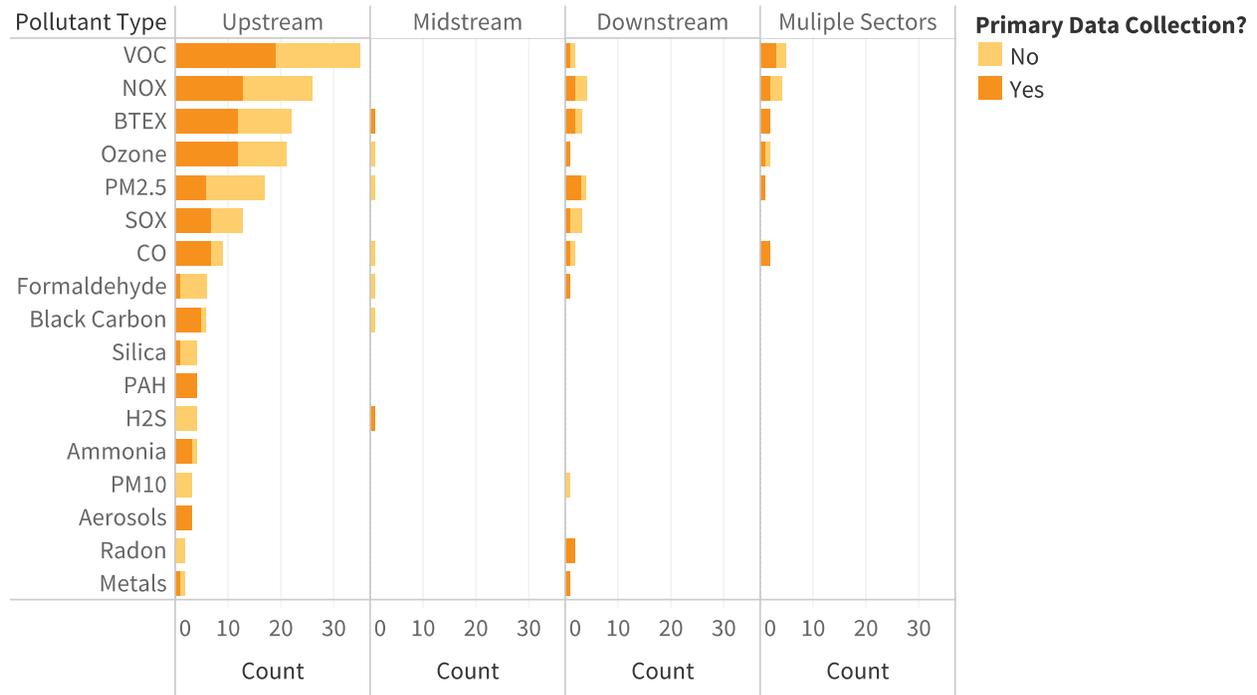
Figure 2.11. Number of pollutants, colored by primary data collection and sorted by upstream, midstream, downstream.
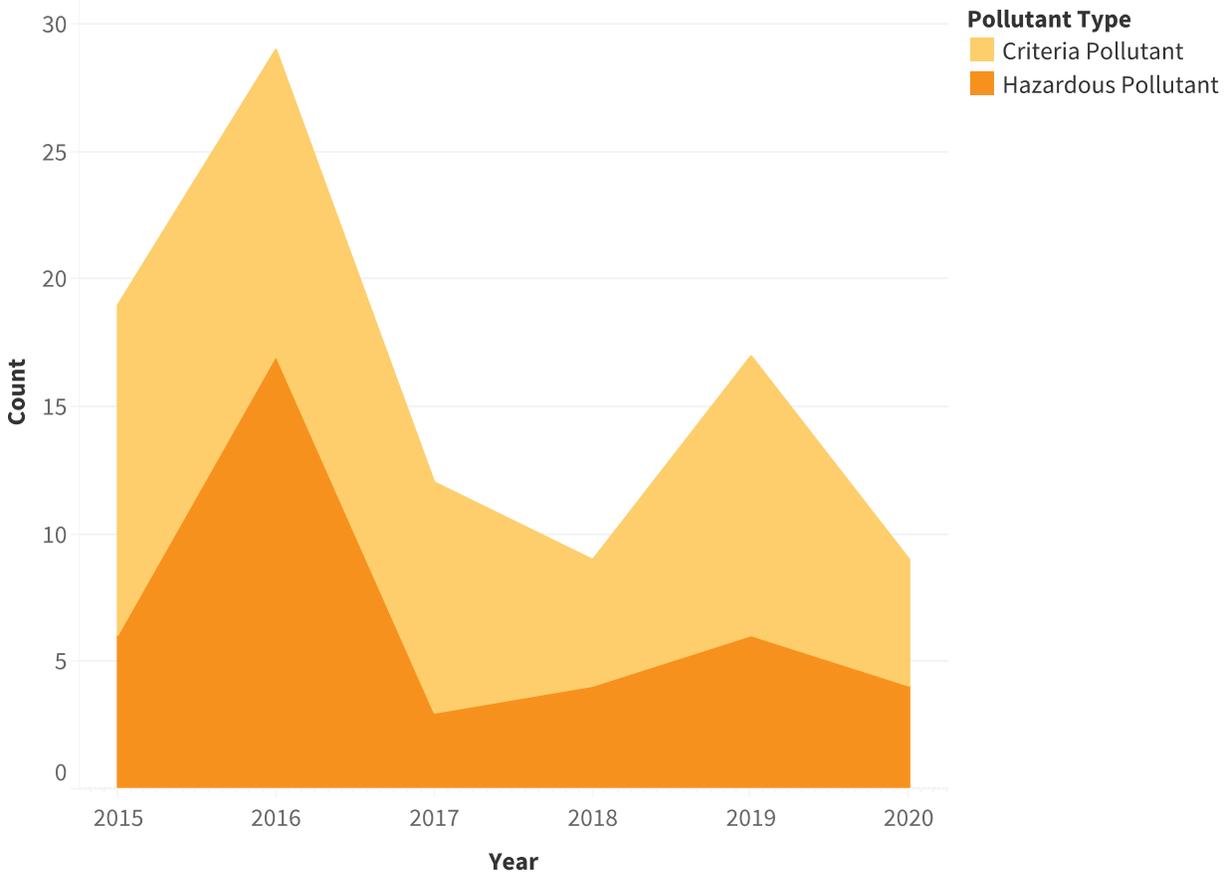
Figure 2.12. Criteria and hazardous pollutants over the period of 2015-2020. Note that 2020 was a partial year due to our systematic review cut-off date of August 2020.
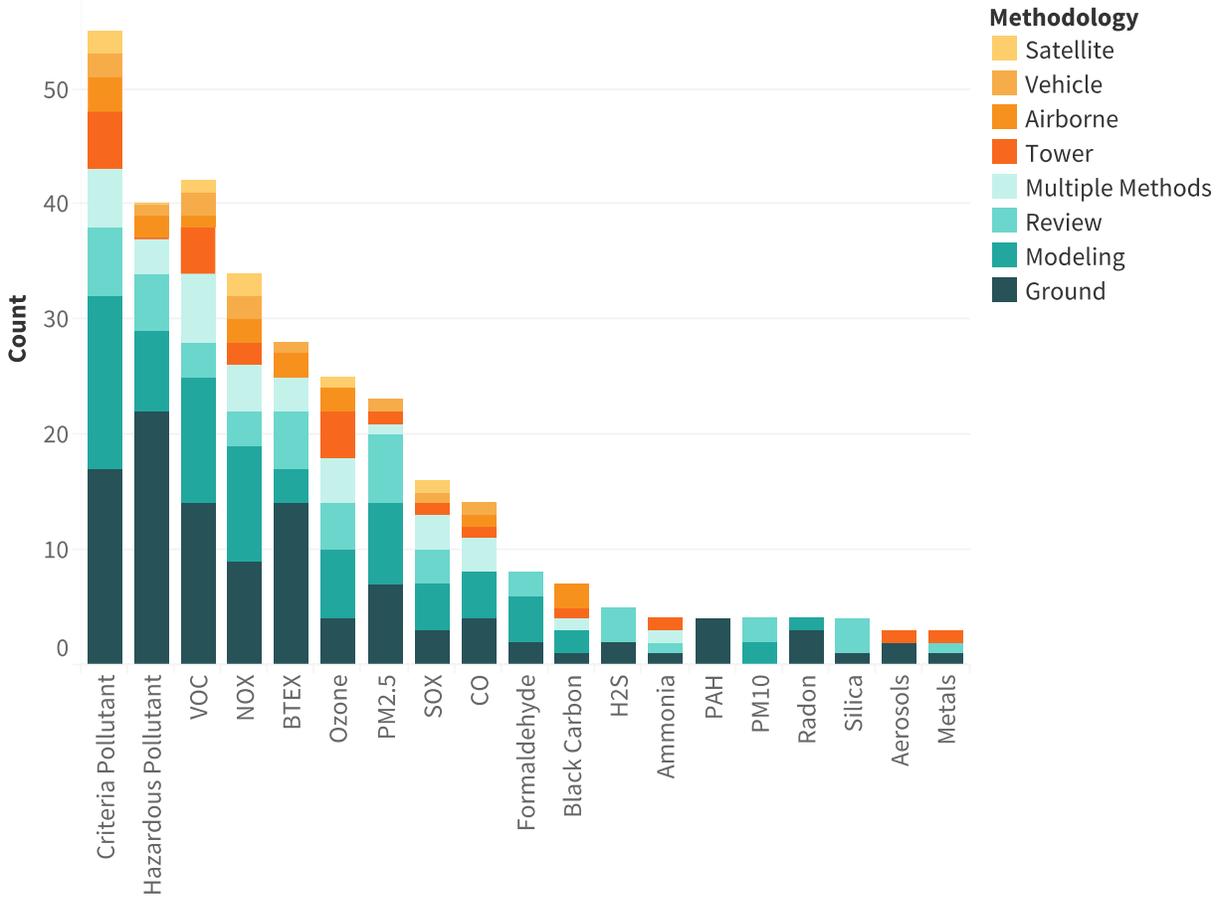
Figure 2.13. Papers that reported health-damaging air pollutants, sorted by pollutants the papers measured and colored by the methodology the paper used.

3.0 Results: Methane studies (2015–2020)

Methane (CH_4) is a potent greenhouse gas and is the primary constituent of natural gas. Thus, anytime natural gas is released or leaked at any point of the oil or natural gas supply chain, the emitted methane impacts climate change. The instantaneous global warming potential—the amount of heat absorbed by methane when it's immediately released into the atmosphere—is 120 times greater than that of carbon dioxide (CO_2). However, methane's ability to absorb heat decreases over time, resulting in a much less potent, but still significant 28-36 times greater warming potential than CO_2 after 100 years (Balcombe et al. 2018).

Global atmospheric methane concentrations have more than doubled in the past 150 years, in conjunction with global industrialization and urbanization (IPCC, 2019). The coincident rapid development of unconventional hydrocarbon extraction and global rise in atmospheric methane led to an increased focus on methane emissions associated with hydrocarbon production, transportation, and consumption. Reducing anthropogenic sources of methane was recently labeled as the "most powerful lever" to slow global warming in the near term, according to the recent United Nations Global Methane Assessment (United Nations Environment Program (UNEP) & Climate and Clean Air Coalition (CCAC), 2021).

Efforts to reduce methane emissions have been limited by a lack of reliable data. However, much progress has been made over the past decade supported by the development of a wide variety of measurement technologies and related scientific study to evaluate their effectiveness (see Appendix A). Moreover, a lack of perfect information should not impede progress to reduce emissions. Recent research has provided a better understanding of the sources and magnitude of methane leakage and the diverse processes and conditions that result in leakage, narrowing the gap between top-down and bottom-up methane emission estimates.

This section provides an assessment of peer-reviewed literature from 2015-2020 to determine the current state of the science regarding methane emissions in North America. We identified 165 methane studies published between 2015-2020 in addition to 29 studies which measured both methane and health-damaging air pollutant (HDAP) emissions during this time. Due to the rapid pace of developments in the science of methane emissions, select high-priority studies identified via external peer review from 2020–2021 that are outside the timeframe of our systematic review are incorporated throughout this section to provide additional context about recent developments. Below we discuss methane studies by the upstream, midstream, and downstream sectors of the oil and gas supply chain as defined in Table 1.1 and identify key findings, research gaps, and recommendations. While all studies in our systematic review were considered in aggregate to produce the findings, research gaps, and recommendations presented here, not every paper will be mentioned explicitly. Recommendations in this section are based on conclusions drawn from the reviewed literature in its entirety.

PSE

3.1.   Review of upstream methane studies

Three of the major themes that emerged in upstream methane literature prior to 2015 and continued throughout the period of our systematic review were:

- Bottom-up inventories versus top-down estimates;
- Super-emitters; and
- Monitoring methodologies and source apportioning

Appendix A provides an in-depth summary of the literature from 2011-2014 surrounding these major themes. Our review of upstream methane studies is structured around these three major themes. Other areas of study including emissions from flares, abandoned wells, and single-point failures are also discussed here.

The overarching issue in early methane emission studies was the uncertainty regarding the mass of methane emitted from the oil and gas sector, rooted in the lack of agreement between bottom-up and top-down derived methane emission estimates. The terms "bottom-up" and "top-down" are general scientific terms that describe two different approaches of data gathering and processing. Bottom-up methane estimates take a disaggregated approach and rely on emissions measurements made directly from components or at the site level with the goal of obtaining a statistically representative sample of sources, then extrapolating these emission factors to all sources. Top-down approaches measure concentrations of methane in the atmosphere and then apply various models to determine emissions per unit time, space, or source(s). Bottom-up component-based methane inventories, such as the U.S. Environmental Protection Agency (EPA) Greenhouse Gas Inventory (GHGI), generally underestimate methane emissions compared to top-down studies.

A potential explanation for the lack of agreement between early bottom-up and top-down studies was the presence of super-emitters. The term "super-emitter" was used to describe the few disproportionately large emitting sources observed in early upstream measurement studies. More technically, the observed emissions distributions demonstrated "heavy tails" or "fat tails" that exhibited more high-emitting sources than would be expected in a normal distribution. If total emissions are dominated by a small number of high-emitting sources, emissions factors derived from measures of central tendency (i.e., arithmetic mean of samples) will systematically under-predict total emissions.

Given the highly integrated, spatially dispersed, and heterogeneous nature of the North American oil and natural gas industry, direct measurement of the quantity of emissions from every source is untenable. Monitoring methodologies and source apportioning techniques are being developed and refined to reduce uncertainties in both top-down and bottom-up emission estimates in the most cost-effective manner possible.

PSE

3.1.1. Upstream methane: detailed findings

The majority of studies evaluating methane emissions (but not from health-damaging air pollutants, discussed in Sections 4 and 5) from the oil and gas industry focus on the upstream sector (72%). We identified 118 studies published between 2015-2020 that focused on upstream methane emissions from oil and gas development in North America. Of the 118 studies, 54 studies also included data on midstream methane sources (e.g., compressor stations), and 30 studies examined emissions from the entire natural gas supply chain. For studies that covered both upstream and midstream sources, efforts were made to discuss results relevant to each part of the supply chain when possible. Studies that focus on the whole natural gas supply chain, or oil and gas producing basins, typically did not delineate emissions according to upstream, midstream, or downstream and are discussed as a whole.

Emission estimates

Upstream studies in our review generally focused on characterizing emissions from major oil and gas producing regions, as opposed to specific parts of the oil and gas supply chain. Emission estimates for the upstream sector often utilize aerial-based top-down techniques which capture all sources of methane within the study region. As such, upstream studies of major oil and gas producing regions may also capture any midstream—and to a lesser extent, downstream—emissions that are present. Major studies of methane emissions in North America are described below in further detail and summarized in Table 3.1. The direct comparison of emission estimates for the same basin is challenging due to variations in methods and coverage areas, increasing overall uncertainty.

United States

A major study by Alvarez et al. (2018) estimated U.S. site-level, bottom-up methane emissions for the oil and gas sector in 2015 to be 13 teragrams/year (Tg/yr), with the production, gathering, and processing sectors contributing 7.6 Tg/yr, 2.6 Tg/yr, and 0.72 Tg/yr, respectively. In comparison, the U.S. EPA Greenhouse Gas Inventory (GHGI) for the oil and gas sector in 2015 was 8.1 Tg/yr for the entire supply chain, with the production sector contributing 3.5 Tg/yr. Based on unified facility-scale bottom-up and aerial-based top-down estimates, Alvarez et al. (2018) estimated a production-normalized methane emission rate—emissions as a percentage of total production—for the oil and gas sector of 2.3% (+0.4%/-0.3%). The production sector accounts for over half of the total U.S. oil and gas sector emissions (Alvarez et al. 2018), in line with regional estimates from the Permian Basin (Texas/New Mexico) (Cusworth et al. 2021a). Another major study, which synthesized the results of the Barnett Shale Coordinated Campaign, found agreement between top-down and bottom-up studies and estimated a production-normalized emission rate of 1.5% for the Barnett Shale region (Texas) (Zavala-Araiza et al. 2015a).

On a national scale, Lan et al. (2019) analyzed atmospheric methane and found moderate increases in overall U.S. methane emission rates of 3.4 ±1.4% per year from 2006 to 2015, which

aligns with the 3% increase in methane emissions from 2018 to 2019 calculated in the 2019 U.S. EPA GHGI (US EPA, 2021). Previous studies of methane emissions across North America found substantially larger increases in emissions (Franco et al. 2016; Helmig et al. 2016; Turner et al. 2016). However, these apparent increases may have been due to overestimations caused by the use of static ethane to methane ratios (Lan et al. 2019). Similarly, Turner et al. (2016) found an overall increase of more than 30% in methane emissions from the United States using 2002 to 2014 satellite data. However, this apparent increase may have been due to variability in atmospheric transport, satellite sampling bias, the choice of background regions, and inability to account for variations in seasonal cycles (Bruhwiler et al. 2017). Smith et al. (2017) found similar results in the San Juan Basin (New Mexico/Colorado), with a five-day aerial study in 2015 finding emissions in line with satellite-based estimates from 2003–2009. Additional frequent and dense sampling from a combination of methods and platforms is needed to accurately simulate variability and determine long-term emission trends on a continental basis (Bruhwiler et al. 2017).

An aerial survey in California estimated statewide point source emissions to be 0.618 Tg/yr, or 34-46% of the 2016 California Air Resources Board (CARB) inventory, with the oil and gas sector contributing 26% of observed point source methane emissions (Duren et al. 2019). Upstream oil and gas production was responsible for 79% of methane point source emissions from the oil and gas industry, with 85%, 14%, and 1% of emissions originating from the southern San Joaquin Valley, Los Angeles or Ventura Counties, and the Sacramento Valley, respectively (Duren et al. 2019).

In a study outside the timeframe of our review, Lyon et al. (2021) examined the effects of oil prices during the COVID-19 pandemic on methane emissions in the Permian Basin (Texas/New Mexico). They found production-normalized emissions decreased from 3.3% to 1.9% with declining oil prices, but recovered to near initial values (>3.0%) as the price of oil recovered. High emission rates during periods with high oil prices are likely due to increases in associated gas production—gas produced as a byproduct of crude oil—outpacing midstream capacity, leading to increased flaring and venting. They suggest the temporary reduction in emissions was due to the decline in well development and associated effects on midstream infrastructure and flaring (Lyon et al. 2021).

PSE

Table 3.1. Summary of emission rates, production-normalized emissions rates, and comparisons to bottom-up inventories from studies of oil and gas production regions. Confidence intervals provided as a range when available. Adapted[1] from Ren et al. (2019).

Normalized emission rate %	Emission rate	Compared to inventory estimates	Region(s)	Approach	Study
17%	32±7 Gg/yr	1.5x (CARB)	Los Angeles Basin - CA	Aircraft measurement and emissions inventory	Peischl et al. 2013
~12%	N/A	N/A	Eagle Ford Shale - TX	Remote sensing	Howarth 2015; Schneising et al. 2014
10.1±7.3%	N/A	N/A	Bakken Shale - ND	Remote Sensing	Peischl et al. 2016; Schneising et al. 2014
2.8–17.3%	5.6–39.2 kg/s	6.4–12.8 kg/s	Southwestern PA	Aircraft mass balance	Caulton et al. 2014b
6.2–11.7%	55±15×10³ kg/hr	N/A	Uintah Basin - UT	Aircraft mass balance	Karion et al. 2013
6–20%	$(3.3±1.5)×10^7$ g/hr	NA	Western Arkoma Basin - AR	Aircraft mass balance	Peischl et al. 2015
4.2–8.4%	0.25±0.05 Tg/yr	0.51–0.85x (2014 EPA GHGI)	Bakken Shale - ND	Aircraft mass balance	Peischl et al. 2016
3.7±0.7%	2.7±0.5 Tg/yr	>2x (2017 EPA GHGI)	Permian Basin -TX, NM	Satellite-based methane flux inversion	Zhang et al. 2020
3.6–7.9%	N/A	N/A	U.S.	Based on emission estimates from EPA and other government reports	Howarth et al. 2011
5.4±2.0%	28±7 t/hr	N/A	Bakken Shale - ND	Aircraft mass balance	Peischl et al. 2018
2.3–7.7%	67–229 Gg/yr	>1.6x (2006 WRAP)	Northeastern CO	Ground level ambient tall tower and mobile sampling	Pétron et al. 2012

[1] Adapted as permitted by the creative commons license. Changes were made to the formatting of the original work in order to produce this figure.

PSE

Normalized emission rate %	Emission rate	Compared to inventory estimates	Region(s)	Approach	Study
3.2±1.1%	42±11 t/hr	N/A	Eagle Ford Shale (East) - TX	Aircraft mass balance	Peischl et al. 2018
2.3% (2.0–2.7%)	13 (+2.1/-1.7) Tg/yr	1.6x (2015 EPA GHGI)	U.S.	Compiled site-level measurements	Alvarez et al. 2018
2.1±0.9%	18±8 t/hr	N/A	Denver Basin - CO	Aircraft mass balance	Peischl et al. 2018
2.0±0.6%	41±11 t/hr	N/A	Eagle Ford Shale (West) -TX	Aircraft mass balance	Peischl et al. 2018
1.6±0.3%	$(60\pm11)\times10^3$ kg/hr	3x (2014 GHGRP)	Barnett Shale -TX	Aircraft mass balance	Karion et al. 2015
1.5±1.0%	66±22 t/hr	N/A	Barnett Shale -TX	Aircraft mass balance	Peischl et al. 2018
1.5±0.3%	59 [48–73] t/hr	1.9x (2014 EPA GHGI)	Barnett Shale -TX	Aircraft mass balance and ground-based site-level measurements	Zavala-Araiza et al. 2015a
1.4% (0.98–2.0%)	1150 Gg/yr	N/A	PA, WV	Ground site-level source sampling	Omara et al. 2016
1.3±0.1%	6.6 (6.1–7.1) Tg/yr	1.8x (2015 EPA GHGI)	U.S.	Compiled component-level measurements	Rutherford et al. 2021[1]
0.8%[2]	6305 Gg/yr	N/A	U.S.	EPA GHGI 2019 and EIA gross natural gas withdrawals in 2019	US EIA, 2021; US EPA, 2021
1.1% (0–3.5%)	5.94 kg/s	N/A	Southwestern PA, Northern WV	Aircraft mass balance	Ren et al. 2019
1.0–2.8%	$(3.9\pm1.8)\times10^7$ g/hr	N/A	Fayetteville Shale - AR	Aircraft mass balance	Peischl et al. 2015
1.0±0.5%	42±18 t/hr	1.7x (Maasakkers et al. 2016)	Haynesville Shale - LA, TX	Aircraft mass balance	Peischl et al. 2018
0.5±0.3%	N/A	2–8x (PA DEP)	PA	Aircraft weather research and forecasting modeling (unconventional gas sites)	Barkley et al. 2019b
1.0–2.1%	$(8.0\pm2.7)\times10^7$ g/hr	N/A	Haynesville Shale - LA, TX	Aircraft mass balance	Peischl et al. 2015

PSE

Normalized emission rate %	Emission rate	Compared to inventory estimates	Region(s)	Approach	Study
0.53% (0.45–0.64%)	0.35–0.49 Tg/yr	4x (2015 PA DEP)	Marcellus Shale - PA	Vehicle-based downwind sampling with Gaussian plume modeling	Caulton et al. 2019
0.47±0.05%	2,421 (+245/-237) Gg/yr	0.59x (2012 EPA GHGI)	U.S.	Ground source sampling at gathering and processing facilities	Marchese et al. 2015
N/A	1,290 [1,246–1,342] Gg/yr	0.66x (2017 EPA GHGI)	U.S.	Ground source sampling at gathering and processing facilities	Zimmerle et al. 2020
0.42%	957±200 Gg/yr	0.63–0.96x (2011 EPA GHGI)	U.S.	Source sampling and national emissions inventory estimates (well completion flowback, pneumatic pump, controllers)	Allen et al. 2013
0.36±0.09% (mean) 0.4±0.32% (weighted mean)	20 t/hr	N/A	Northeastern PA	Aircraft mass balance and model optimization	Barkley et al. 2017
0.18–0.41%	$(1.5±0.6)×10^7$ g/hr	N/A	Northeastern PA	Aircraft mass balance	Peischl et al. 2015
N/A	520–610 Gg/yr	5.5–9x (EPA NEI)	Permian Basin – NM	Vehicle-based downwind sampling with inverse Gaussian dispersion modeling	Robertson et al. 2020[1]
N/A	N/A	1.6–2.2x (ECCC NIR)	British Columbia - Canada	Airborne LiDAR and on-site OGI surveys	Tyner and Johnson 2021[1]
N/A	3.0 Tg/yr	~2x (ECCC NIR)	Western Canada	Aircraft Lagrangian particle dispersion modeling	Chan et al. 2020[1]
N/A	$5.1×10^6$ m^3/day	1.5x (ECCC NIR)	Alberta -Canada	Vehicle-based downwind sampling with Gaussian dispersion modeling	MacKay et al. 2021[1]
1.9–3.3%[3]	N/A	N/A	Permian Basin – TX, NM	Aircraft, tower, and satellite	Lyon et al. 2021[1]
N/A	29-49 t/hr	10-20x (2018 Mexico national inventory)	Onshore Sureste Basin - Mexico	Aircraft and satellite	Zavala-Araiza et al. 2021[1]

PSE

Normalized emission rate %	Emission rate	Compared to inventory estimates	Region(s)	Approach	Study
N/A	5.0±0.2 Tg/yr	1.4x[4]	Eastern Mexico	Satellite	Shen et al. 2021[1]
N/A	N/A	1.25–1.5x (ECCC NIR)	Alberta -Canada	Aircraft mass balance and industry data	Johnson et al. 2017
N/A	76 t/hr	0.89–1.82x (2012 EPA GHGI)	Haynesville-Bossier basin – TX, LA	Aircraft inverse modeling	Cui et al. 2019a
N/A	N/A	1.8±0.7x (2012 EPA GHGI)	South Central and Midwestern U.S.	Aircraft weather research and forecasting modeling	Barkley et al. 2019a
N/A	118–197 BCF	2x (TxRRC)	Permian Basin - TX; Eagle Ford Shale - TX	Satellite (flaring and venting)	Willyard and Schade 2019
N/A	N/A	2–5x (GHGRP)	Prudhoe Bay - AK	Aircraft based mass balance	Floerchinger et al. 2019
N/A	46,200 kg/hr	1.5x (2013 EPA GHGI); 2.7x (2013 GHGRP)	Barnett Shale - TX	Compiled bottom-up measurements with state and federal databases. Monte Carlo simulations	Lyon et al. 2015

GHGI – EPA Greenhouse Gas Inventory; GHGRP – EPA Greenhouse Gas Reporting Program; EIA - Energy Information Administration; PA DEP - Pennsylvania Department of Environmental Protection; TxRRC - Texas Railroad Commision; WRAP - Western Regional Air Partnership; CARB - California Air Resources Board; ECCC NIR - Environment and Climate Change Canada National Inventory Report; N/A – Not available.

1. Study outside the timeframe of our systematic review.

2. Calculated using U.S. EIA natural gas gross withdrawals of 1.158 trillion m^3 in 2019, and EPA GHGI for natural gas systems of 9.27 billion m^3 in 2019.

3. Range represents estimates made prior to and during decreased oil prices caused by the COVID-19 pandemic.

4. Based on gridded inventory from Scarpelli et al. 2020, which used Mexico's National Inventory of Greenhouse Gases and Compounds

PSE

Canada

In Canada, the production and processing of oil and gas in Alberta and Saskatchewan account for approximately 70% of methane emission from the oil and gas sector (Chan et al. 2020). Sheng et al. (2017) disaggregated 2013 emissions inventories to determine methane emission contributions from individual source sectors. Emissions from oil production/processing and natural gas production/processing accounted for 22% and 61% of total oil and gas sector emissions, respectively. In Alberta, the region with the highest emissions from the oil and gas sector in Canada (Sheng et al. 2017), the total upstream oil and gas methane emissions are estimated to be 25% to 50% greater than bottom-up inventories (Johnson et al. 2017).

In more recent studies outside the timeframe of our review, methane emission estimates from the upstream sector in Alberta and British Columbia were 1.5 times and 1.6–2.2 times greater than national inventory estimates, respectively (MacKay et al. 2021; Tyner and Johnson, 2021). Similarly, overall oil and gas sector emission estimates for Canada were almost twice that of national inventory estimates from 2010-2017 (Chan et al. 2020).

Mexico

Few studies in our review period examined methane emissions in Mexico. Sheng et al. (2017) disaggregated 2010 emissions inventories to find that emissions from onshore oil production/processing and natural gas production/processing accounted for 22% and 18% of total oil and gas sector emissions, respectively, with offshore oil production accounting for 52%. The overall trend in methane emissions from Mexico show a decrease from 2010 to 2016, likely due to a decrease in cattle (Sheng et al. 2017, 2018).

More recent studies outside the timeframe of our systematic review have examined emissions from Mexico in greater detail. Scarpelli et al. (2020) developed a gridded inventory of emissions and found approximately 15% (0.7 Tg/yr) of anthropogenic methane emissions are from the oil and gas sector, 51% of which originates from offshore oil and gas activities. Flaring and venting during oil production were identified as the highest contributors to sector emissions, accounting for 32% and 24%, respectively. In Mexico's Sureste Basin, where the majority of oil and gas production occurs, satellite and aerial data indicates that emissions from onshore production and processing are greater than national inventories by more than an order of magnitude, with inefficient flaring cited as a primary source of emissions (Zavala-Araiza et al. 2021). Another satellite study in Eastern Mexico found a much smaller difference with national inventories, with oil and gas sector emissions (1.3 Tg/yr) greater than bottom-up inventories by a factor of two (Shen et al. 2021).

Major sources of emissions

Major sources of upstream emissions include pneumatic controllers, equipment leaks due to malfunctions (e.g., stuck valves, open thief hatches, etc.), well venting (i.e., liquid unloading), and maintenance activities (Allen et al. 2015a, 2015b; Lyon et al. 2015; Zavala-Araiza et al.

2015b). Tank-related emissions (e.g., tank vents, thief hatches, etc.) are consistently cited as a major contributor to upstream methane emissions in various regions of the United States and Canada, supporting the need for additional controls or tankless designs (Brandt et al. 2016a; Cardoso-Saldaña et al. 2019; Englander et al. 2018; Lyman et al. 2019; Lyon et al. 2016; Ravikumar et al. 2020; Tyner and Johnson, 2021). Emissions from venting and flaring (e.g., unlit flares or incomplete combustion) are also cited as a key contributor to upstream methane emissions in the Bakken Shale (North Dakota), the Permian Basin (Texas/New Mexico), British Columbia, and Mexico (Brandt et al. 2016b; Gvakharia et al. 2017; Johnson et al. 2017; Lyon et al. 2021; Scarpelli et al. 2020; Tyner and Johnson, 2021; Zavala-Araiza et al. 2021), but overall are not as prevalent in the literature as tank emissions.

No relationship was observed between operator size and emissions intensity in the Bakken Shale (Pennsylvania) and Weyburn-Midale (Saskatchewan) regions; however, differences in emission intensities were observed between conventional and unconventional sites in co-located developments (Baillie et al. 2019). The percent of wells with reported fugitive emissions was significantly higher for unconventional wells compared to conventional wells in the Marcellus Shale (Pennsylvania) (18.5% vs. 3.4%) (Ingraffea et al. 2020).

Methane breakeven point

Research in the early 2010s addressed the lifecycle opportunity costs of switching from coal to natural gas, noting that while direct CO_2 emissions are lower from gas combustion than coal combustion to generate an equivalent amount of electricity, a certain breakeven threshold level of methane leakage exists whereby net climate benefits of switching from coal to gas could be nullified. Estimates prior to 2015 calculate production-normalized rates—emissions as a percentage of total production—where natural gas breaks even with coal ranging from 1-5% (see Appendix A).

Sanchez and Mays (2015) calculated a production-normalized emission rate where methane breaks even with coal for electricity generation of 3.9% and 9.1% using the 20-year and 100-year global warming potential of methane, respectively. Hong and Howarth (2016) updated a previous breakeven analysis by Alvarez et al. (2012) and calculated a breakeven point of 2.7% for electricity generation. Ren et al. (2019) calculated a more conservative 2.4% breakeven point using the 20-year global warming potential of methane.

As shown in Table 3.1, production-normalized methane emission rates have a high degree of uncertainty, with rates in various regions ranging from 17% to 0.4%. Emissions rates vary between regions, within the same region, and as a result of measurement methodology. Although emissions rates in many studies are close to or below the 2.4% breakeven point calculated by Ren et al. (2019), some individual regions or components have emissions above this breakeven point and overall uncertainty in emissions estimates remains high. Nonetheless, the environmental case for natural gas does not depend on beating the emissions

PSE

performance of the most carbon-intensive fuel, but in ensuring that emissions intensity is as low as practicable.

Top-down vs. bottom-up: underestimation of methane emissions by component-level inventories

Bottom-up methane emissions estimates are generally categorized as either component-level, such as the U.S. EPA Greenhouse Gas Inventory (GHGI), which relies on emission factors and activity data, or site-level, such as downwind vehicle-based studies which capture emissions from individual sites or facilities. Top-down studies typically use aerial- and satellite-based measurements, ideally capturing all methane emissions from a region. In the context of differences between top-down and bottom-up studies, site-level and component-level bottom-up estimates are considered separately. It is important to note that top-down and bottom-up methods are complimentary. Top-down methods provide an overall picture of methane emissions for validation. Bottom-up methods excel at identifying and quantifying emissions from individual sources to inform leak detection and repair (LDAR) programs, impacts from future development, and ultimately guide policy and program implementation.

Studies suggest that component-level bottom-up estimates routinely underestimate emissions by a factor of 1.6 to 2 in the United States (Table 3.1), compared to top-down studies and site-level bottom-up studies (Alvarez et al. 2018; Barkley et al. 2019a; Zavala-Araiza et al. 2021; Zhang et al. 2020). Undifferentiated top-down anthropogenic methane estimates for California found similar results, with emissions 1.14 to 1.47 times greater than state inventories (Cui et al. 2019b). A few studies estimate emissions lower than the U.S. EPA GHGI, but these are in the minority of findings and were limited to specific parts of the oil and gas supply chain (Allen et al. 2013; Marchese et al. 2015; Zimmerle et al. 2020). Studies in Canada and Mexico also indicate that national component-level bottom-up inventories routinely underestimate emissions (Chan et al. 2020; MacKay et al. 2021; Scarpelli et al. 2020; Shen et al. 2021; Tyner and Johnson, 2021; Zavala-Araiza et al. 2021).

Methane emissions from the upstream sector represent the largest fraction of oil and gas-related methane emissions (Alvarez et al. 2018). As such, uncertainties associated with upstream measurements dominate overall uncertainties for the oil and gas sector as a whole (Alvarez et al. 2018; Barkley et al. 2019a). Emissions estimates from component-level bottom-up inventories and top-down studies often disagree, with top-down estimates generally being the greater of the two (Alvarez et al. 2018; Brandt et al. 2014; Klemun and Trancik, 2019). Studies suggest that bottom-up component-level inventories may be lower than top-down estimates due to multiple factors including (Alvarez et al. 2018; Barkley et al. 2017; Vaughn et al. 2018):

- Missing, underrepresented, or inadequately characterized sources (e.g., super-emitters);
- Temporal variability (e.g., maintenance activities, unloading, etc.);

PSE

- Poor source attribution;
- Inaccurate or generalized emissions factors; and
- Uncertainties and constraints associated with chosen methodologies.

Underrepresented and inadequately characterized sources

Inadequate characterization of super-emitters and the prevalence of heavy-tailed emissions distributions—where more high-emitting sources are observed than would be expected in a normal distribution—have been suggested by many studies to be a potential reason for top-down versus bottom-up discrepancies. If total emissions are dominated by a small number of high-emitting sources, emissions factors derived from measures of central tendency (i.e., arithmetic mean of samples) will systematically underpredict total emissions. Alvarez et al. (2018) used downwind site-level emissions data from studies in six production basins to estimate emission probability density functions. They found that when observed probability distributions were integrated into bottom-up estimates, they did not see a statistical difference with top-down estimates, with a mean difference of 11% (95% confidence interval of -17 to 41%) across the nine basins Figure 3.1). As found in other studies, both the top-down and calculated bottom-up estimates were significantly greater than existing U.S. EPA GHGI for 2015. These findings suggest that one of the primary reasons for differences between bottom-up inventories and top-down estimates is that current component-level emissions used for the U.S. EPA GHGI development undersample emissions from super-emitters and abnormal operating conditions, or do not adequately incorporate observed highly skewed emissions distributions, effectively underestimating total emissions. A global probabilistic emissions model developed by Balcombe et al. (2018) found a log-log-logistic distribution for multiple natural gas supply chain scenarios, suggesting that even lognormal distributions may underestimate emissions in some regions due to the highly-skewed distributions observed in some studies (Balcombe et al. 2018; Brandt et al. 2016a; Omara et al. 2018). Higher likelihoods of super-emitters require greater sample sizes to adequately characterize the underlying distribution. Undersampling of super-emitters and high-emitting components and sites may be due to small sample sizes, temporal limitations of studies, technical or accessibility difficulties, liability, and safety risks associated with measuring large emissions using ground-based techniques (Alvarez et al. 2018; Barkley et al. 2017; Lyon et al. 2016; Zavala-Araiza et al. 2017).
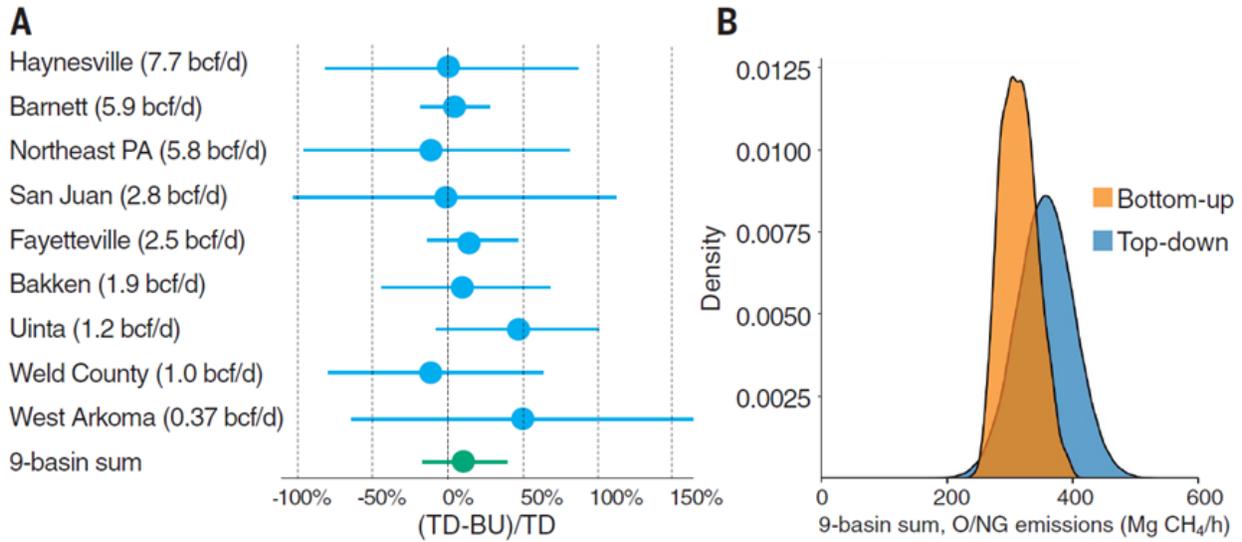
Figure 3.1. Comparison of top-down (TD) and bottom-up (BU) site-level estimates for nine major oil and gas producing basins when emission probability distributions were integrated into bottom-up site-level estimates. (A) Top-down normalized difference between TD and BU site-level emissions by region. Error bars represent 95% CI. (B) Probability densities for nine basin sum of TD and BU site-level estimates. Source: Reproduced from Alvarez et al. (2018) with permission[2].

Another suggested reason for the top-down versus bottom-up inventory discrepancy is the use of generalized emission factors (Cui et al. 2019b). A comparison of upstream emission distributions from nine major oil and gas basins found that skewed emissions distributions varied greatly between basins (Alvarez et al. 2018). A study in the Eagle Ford Shale (Texas) also found that the location and quantities of emitters, and the magnitude of emissions, can vary significantly within a single basin (Lavoie et al. 2017a). Previous studies that developed gridded inventories based on United States, Canada, and Mexico national inventories spatially allocated national emissions totals according to regional well and site-level information (Maasakkers et al. 2016; Sheng et al. 2017). In the case of Mexico's National Inventory of Greenhouse Gases and Compounds, constructed by the Instituto Nacional de Ecología y Cambio Climático (INECC), emission factors for the oil and gas sector are primarily based on those from other countries and are not Mexico specific (Shen et al. 2021). It is generally agreed that emissions vary both temporally and spatially, and the use of aggregated annual national emission factors and activity levels may not adequately capture inter- and intra-basin

---

[2] From "Assessment of Methane Emissions from the U.S. Oil and Gas Supply Chain", Alvarez et. Al, 2018. Reprinted with permission from AAAS.

variability, resulting in overall discrepancies between bottom-up inventories and top-down estimates (Vaughn et al. 2018).

The presence of uninventoried or undercounted sources may also contribute to differences in emissions inventories (Cui et al. 2019b). Known emissions sources from the oil and gas sector that are not included in the current U.S. EPA GHGI include well blowout events, mud degassing, storage wells, and post-meter emissions (US EPA, 2021). Other emissions sources (such as abandoned wells) that were added to the U.S. EPA GHGI in 2018 contributed to underestimates from bottom-up inventories prior to 2018 and are likely underestimated in the current U.S. EPA GHGI due to poor well counts. Similarly, emissions from active and abandoned wells in British Columbia are also likely underreported according to operator data reported to the British Columbia Oil and Gas Commission (Wisen et al. 2020). Emissions reporting to the U.S. EPA Greenhouse Gas Reporting Program (GHGRP) is only required for facilities that exceed reporting thresholds (Allen et al. 2016). These uninventoried sources are likely captured in top-down studies, if present, and may contribute to differences with bottom-up inventories.

Top-down approaches may also fail to accurately attribute sources of methane to particular sectors in regions with complex enhancement patterns and multiple co-located emission sources (Barkley et al. 2017), resulting in errors and large uncertainties. Ratios of ethane to methane (C_2H_6:CH_4), commonly used for source apportioning, vary both temporally and spatially. Changes to these ratios over time may lead to overestimates in methane emissions from biogenic sources if the changes are not accounted for (Townsend-Small et al. 2016a). Biogenic methane sources typically have low ethane:methane ratios, but care must be taken to identify and differentiate other sources of low ethane:methane ratio emissions, such as coalmine gas and midstream natural gas (Allen et al. 2017; Ren et al. 2019).

Temporal variability

Temporal variability has also been suggested as a cause of differences between top-down studies and bottom-up inventories. In some regions of the U.S., up to 90% of high-emitting infrastructure displayed intermittent emissions activity (Cusworth et al. 2021a). Measurements that only represent a single snapshot in time may fail to capture hourly, daily, or seasonal variability associated with intermittent activity and may not be representative of typical emissions (Jakober et al. 2015; Lan et al. 2015; Lavoie et al. 2015, 2017b; Nathan et al. 2015; Vaughn et al. 2018). Additionally, the predominance of studies occurring during daytime may result in higher emissions measurements because the data are unable to capture diurnal differences from onsite activities and maintenance (Vaughn et al. 2018). Vaughn et al. (2018) compiled the results of a 2015 coordinated multi-study campaign of basin, facility, and component-level measurements in the Fayetteville region (Arkansas) to reconcile top-down and bottom-up estimates (Bell et al. 2017; Conley et al. 2017; Robertson et al. 2017; Schwietzke et al. 2017; Vaughn et al. 2017; Yacovitch et al. 2017; Zimmerle et al. 2017). They found that diurnal emissions ranged from approximately 10–25 metric tons/hour (t/hr) and observed temporal variations were primarily due to manual liquid unloadings—the intentional venting

PSE

of a well to atmosphere to increase gas flow. Other methane sources such as production, gathering, and transmission did not show significant diurnal trends. Manual liquid unloadings are high-emitting events and account for 30–50% of emissions in the Fayetteville region (Schwietzke et al. 2017), where unloading emissions are five times the expected values of a nine-basin study (Alvarez et al. 2018). Although top-down and bottom-up estimates in the Fayetteville region agreed based on a 95% confidence interval, bottom-up estimates still had slightly lower mean emissions (Vaughn et al. 2018). Differences in bottom-up and top-down estimates could be explained by activity data accuracy, temporal uncertainty in manual liquid unloadings and between top-down aircraft-based transects, manual liquid unloading emissions factor accuracy, or other underrepresented high-emitting sources (Vaughn et al. 2018). Manual operations, and resulting emissions, are more likely to occur during the day when operators are onsite. However, Barkley et al. (2019a) found that high top-down emissions rates observed in an airborne study from the South Central United States, which included the Fayetteville Basin, could not be explained by diurnal differences because their measurements were a mixture of daytime and nighttime emissions across multiple regions. An analysis by Alvarez et al. (2018) found that higher daytime emissions are unlikely to be the cause of differences between top-down and bottom-up inventories in basins outside the Fayetteville. They also argue that daytime bias may result in lower emissions during the day, as upstream sites operate continuously and it is more likely that abnormal operating conditions associated with high-emitting sources are detected and corrected during the daytime when operators are present (Alvarez et al. 2018). Overall, the literature suggests that differences in bottom-up and top-down estimates due to diurnal activities are unique to the Fayetteville, due to the high percentage of emissions associated with manual liquid unloadings compared to other basins. Nonetheless, care should be taken when comparing bottom-up and top-down estimates from different timescales as the result can be misleading (Vaughn et al. 2018).

Emission estimate uncertainties

Uncertainties and constraints of chosen methodologies affect both bottom-up inventories and top-down estimates. The current U.S. EPA GHGI methane emissions estimates for natural gas systems, petroleum systems, and abandoned oil and gas wells have uncertainty bounds based on 95% confidence intervals of 14% to -15%, 29% to -24%, and 217% to -83%, respectively (US EPA, 2021). These bounds stem from uncertainties surrounding the accuracy and representativeness of measured data used for emissions factor calculations and component-level activity data. Top-down methods also have a high degree of uncertainty, as represented in emissions estimate confidence intervals (Table 3.1). Although these uncertainties in methodologies contribute to the discrepancy between top-down and bottom-up estimates, they are usually insufficient to fully explain the difference between top-down and bottom-up inventories.

Reconciling top-down and bottom-up estimates

Many of the recommendations stemming from the late 2014 period were calls for hybridized measurement campaigns—combining direct measurements at the facility scale and top-down atmospheric measurements capturing the same facility among other sources. The Barnett Shale Coordinated Campaign encompassed numerous concurrent measurements in the Barnett Shale, with the goal of addressing many of the potential reasons for disagreement between bottom-up and top-down estimates (Harriss et al. 2015). In Figure 3.2, the sampling campaigns are shown as a function of distance from the target source and segmented by airborne vs. ground-based methods. Overall, the utility of this type of spatial sampling saturation was evident. The researchers were able to pinpoint the causes of both top-down and bottom-up methane emissions estimates—namely the omission of gathering compressor stations from the GHGI. The authors noted that:

"By combining measurements made at multiple spatial scales, the Barnett Shale field campaign contributes to a more robust understanding of methane emissions from an active oil and gas production area. Region-wide emission estimates can be efficiently obtained using airborne top-down methods, while source-specific measurements can provide insights about the contribution of specific source types."

Data from the Barnett Shale Coordinated Campaign was used by Zavala-Araiza et al. (2015) to reconcile top-down and bottom-up estimates within 10% for fossil methane and 0.1% for total methane. They successfully reduced uncertainties in top-down estimates by a combination of multiple mass balance measurements and ethane:methane measurements for source attribution. Bottom-up uncertainties were reduced by integrating observations from multiple ground-based datasets, increasing the number of counted facilities and more accurately capturing high-emitting facilities. Similarly, Vaughn et al. (2018) used data from a 2015 coordinated multi-study campaign in the Fayetteville region and reconciled top-down and bottom-up estimates within 95% confidence intervals. Estimates were reconciled with the use of concurrent multiscale measurements, high resolution spatiotemporal activity data, and unbiased site access. Although these recent studies have had success at reconciling top-down and bottom-up estimates within 95% confidence intervals, average bottom-up estimates remained slightly lower than top-down estimates (Alvarez et al. 2018; Vaughn et al. 2018; Zavala-Araiza et al. 2015a).

PSE
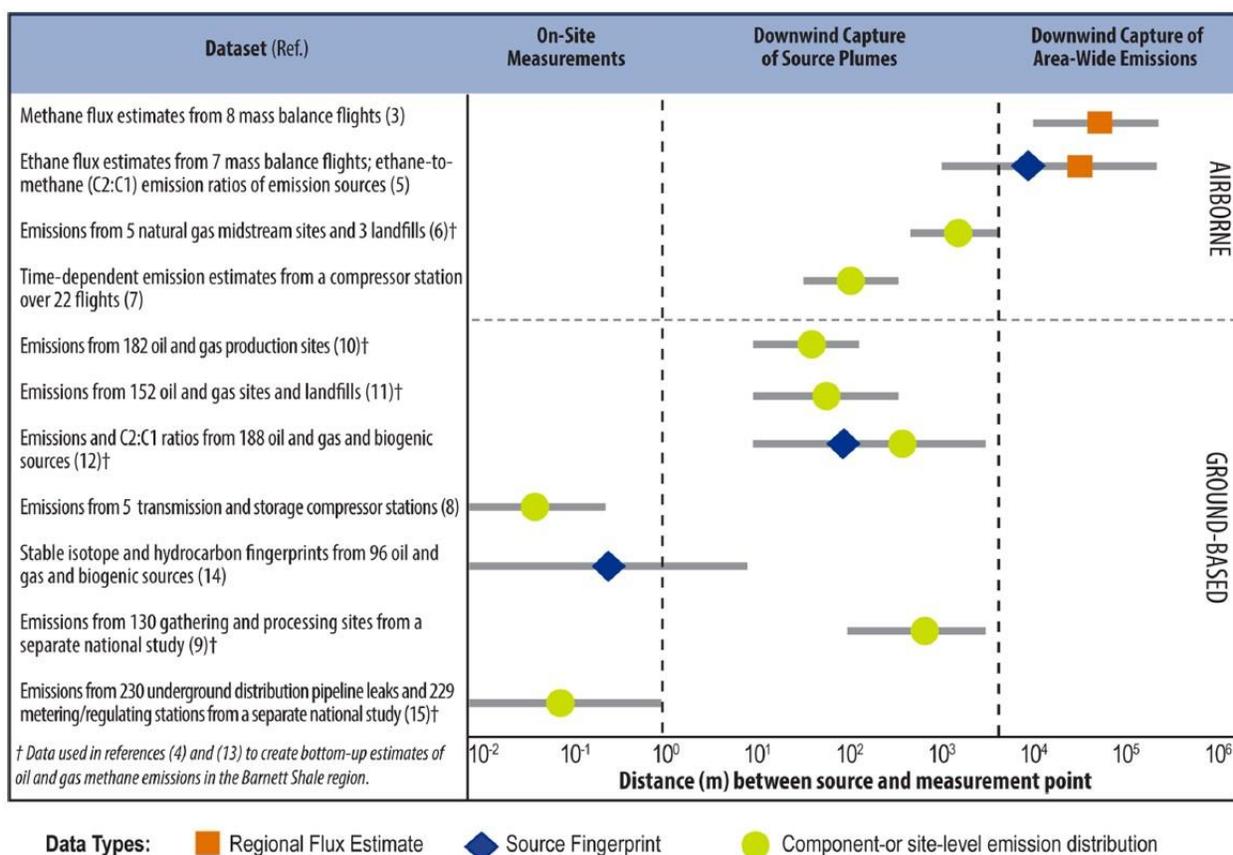
Figure 3.2 Spatial scales of measurements taken over the course of the Barnett Shale Coordinated Campaign. Source: Reprinted with permission[3] from Harriss et al. (2015).

A recent study that falls outside of the timeframe of our review developed a new, inventory-based bottom-up approach using component-level measurements to estimate national methane emissions (Rutherford et al. 2021). Their emission estimates were roughly double that of the 2015 U.S. EPA GHGI. Their production-normalized emission estimate of 1.3% agreed with the site-level estimates by Alvarez et al. (2018) and empirical site-level distributions measured by Omara et al. (2018). The Rutherford et al. (2021) study suggests that differences between component-level estimates like the U.S. EPA GHGI and top-down or site-level studies are not due to inherent issues with bottom-up approaches, but differences in data sources. They suggest that the differences can be resolved by updating component-level bottom-up methodologies and emission factors. This new bottom-up approach also allows for the

---

[3] Reprinted with permission from Robert Harriss, Ramón A. Alvarez, David Lyon, Daniel Zavala-Araiza, Drew Nelson, and Steven P. Hamburg. Environmental Science & Technology 2015. DOI: 10.1021/acs.est.5b02305. Copyright 2015 American Chemical Society.

diagnosis of differences in inventories at the equipment level and identifies equipment leaks and storage tank venting as key contributors to the top-down bottom-up gap.

Studies that reconcile top-down with component-level bottom-up estimates suggest that the gap between bottom-up and top-down estimates can be closed by implementing a multi-tiered observational and monitoring strategy to better capture skewed emissions distributions (Alvarez et al. 2018; Duren et al. 2019; Vaughn et al. 2018) or by implementing new component-level bottom-up approaches (Rutherford et al. 2021). Data from multi-tiered measurements can be used to improve component-level emission factors through increased frequency of measurements that identify malfunctioning or abnormal conditions. Uncertainties can be further decreased by improved access to facilities, reporting for operator and facility data, and accounting of equipment and components (Johnson et al. 2017). Complete and consistent reporting data from operators will also be important for closing the gap between bottom-up and top-down estimates (Ingraffea et al. 2020).

Super-emitters

Super-emitters are present in all parts of the natural gas supply chain (Balcombe et al. 2018; Brandt et al. 2016a; Gvakharia et al. 2017) and are responsible for a disproportionate amount of methane emissions, resulting in skewed or heavy-tailed emissions distributions (Caulton et al. 2019). The term "super-emitter" is not standardized and not all studies use the term "super-emitter," instead describing these sources as high-emitting sources, or in one case as "ultra-emitters" (Lauvaux et al. 2021). Studies that provided percentage breakdowns of high-emitting sources and corresponding disproportionate percent of emissions are listed in Table 3.2.

Based on measurement data from 17 studies and one regulatory dataset, Brandt et al. (2016a) used a working definition of super-emitters as the top 5% of emitting sites and proposed a rule of thumb called the "5-50" rule, where 5% of emission sources or leaks emitted approximately 50% of measured emission. Recent literature regarding major upstream equipment and facilities appears to follow a similar skewed emissions distribution, although the degree varies between individual components and regions. Direct comparison of recent studies to the "5-50" rule is difficult, as many studies do not use the same working definition of super-emitters as the top 5% of emitting sites. Emissions percentages in studies listed in Table 3.2 are reported for anywhere from the top 2.5% to the top 30% of emitting sites.

Super-emitters (or high emitters) are found throughout the upstream sector and are often attributed to abnormally-operating equipment, leaks, and routine operations such as manual liquid unloadings (Alvarez et al. 2018; Englander et al. 2018; Schwietzke et al. 2017). Studies of wellpads show large variations between regions, with 20% of wellpads responsible for 54% of emissions in the Upper Green River (Wyoming) region compared to the Marcellus Shale and Barnett Shale regions, where 20% and 22% of wellpads are responsible for 78% and 80% of emissions, respectively (Omara et al. 2016; Rella et al. 2015; Robertson et al. 2017). A study by Ingraffea et al. (2020) in Pennsylvania suggests that high-emitting unconventional gas and

PSE

conventional oil wells have greater overall emissions compared to unconventional oil wells. In the Bakken Shale and Weyburn-Midale regions, the top 20% of unconventional developments are responsible for 49% of emissions, compared to 28% of emissions for conventional developments (Baillie et al. 2019). In a multi-basin study, Alvarez et al. (2018) found that pneumatic controllers and equipment leaks are responsible for the largest emissions from the production sector, with malfunctioning pneumatic controllers responsible for 66% of total controller emissions (38% of production emissions). Englander et al. (2018) found emissions from tank vents and hatches accounted for 90% of emission sources in the Bakken Shale. Due to the frequency of observations, it was suggested that abnormal operating conditions are the cause, as opposed to routine operations. In a study of 114 gathering facilities throughout the United States, Mitchell et al. (2015) found that substantial venting from liquid storage tanks, observed at 20% of facilities, resulted in a four-fold increase in emissions compared to facilities without venting. High-emitting abandoned wells in Pennsylvania may be due to well depth, status, type, and coal area designation, with 21% of abandoned wells in coal areas responsible for 72% of emissions and 32% of abandoned gas wells responsible for 74% of emissions (Kang et al. 2016). This difference in high emitters is likely due to the presence of unplugged gas wells and regulations requiring the venting of wells in coal areas to prevent explosions.

Super-emitter events have also been observed in flares. Allen et al. (2016) examined GHGRP flare data and found 3.7% of the approximately 20,000 natural gas flares reported to the GHGRP were responsible for 80% of emissions. Associated gas flares had a similar distribution with 7.8% of 503 reported flares accounting for 87% of emissions. Gvakharia et al. (2017) also observed heavily skewed distributions in flaring emissions with 25% of flares responsible for 70% of flaring emissions in the Bakken region. Although outside the timeframe of our review, Lyon et al. (2021) found that the 5% of flares that were unlit and venting in the Permian Basin were responsible for 65% of estimated flare emissions.

A study by Balcombe et al. (2018) found that globally, the top 5% of emitters contribute 40–60% of total methane emissions, similar to the "5-50" rule. Lavoie et al. (2017a) found that even among high-emitting facilities in the Eagle Ford Basin, defined as the top 10% of emitting facilities, the magnitude of emissions varied by up to a factor of six. Caulton et al. (2019) found that 77% of emissions from wellpads in the Pennsylvania parts of the Marcellus Shale were from 10% of sites; proportional loss rates were more skewed with 93% of emissions from 10% of sites. They identified different sites depending on if super-emitters were defined using proportional loss rates or absolute emissions rates. Similarly, Omara et al. (2018) found that in the Marcellus region, high-emitting sites such as unconventional natural gas well pads were typically also high-producing sites, however, lower-producing sites such as conventional natural gas sites emitted a larger fraction of their production. Mitchell et al. (2015) found that production-normalized emissions rates were negatively correlated with throughput for gathering facilities throughout the United States. Proportional loss rates are useful to simplify the comparison of emission rates between areas with similar characteristics and are important to identify excess emissions associated with abnormal conditions (Zavala-Araiza et al. 2018). Zavala-Araiza et al. (2015b) suggested a definition of functional super-emitters as components

PSE

that are in the 85th percentile of proportional loss rates for wells. However, this metric is less useful when comparing regions with significant differences in production rates, as high proportional loss rates may occur at sites with low production volume and resultantly low overall emissions (Zavala-Araiza et al. 2018).

Emission distributions

Zavala-Araiza et al. (2018) compiled the results of multiple ground-based downwind studies, applied a statistical estimator approach, and plotted Lorenz curves for emissions distributions of seven major oil and gas producing basins in North America (Figure 3.3). The curve for the Denver-Julesburg Basin (Colorado) and the Uintah Basin (Utah) overlap in Figure 3.3. Overall, skewed distributions were observed in all studied oil and gas producing regions, with large variations between basins. The Fayetteville region had the most skewed emissions distribution, while the Upper Green River Basin had the least skewed distribution. One possible explanation for regions with highly skewed emissions is overall production volumes, whereby high-emitting sites in high-production volume regions have an even greater impact on distribution skewness than in comparatively low production volume regions (Zavala-Araiza et al. 2018). In a similar study by Brandt et al. (2016a), heavy-tailed distributions were observed in 18 natural gas leakage datasets.

PSE
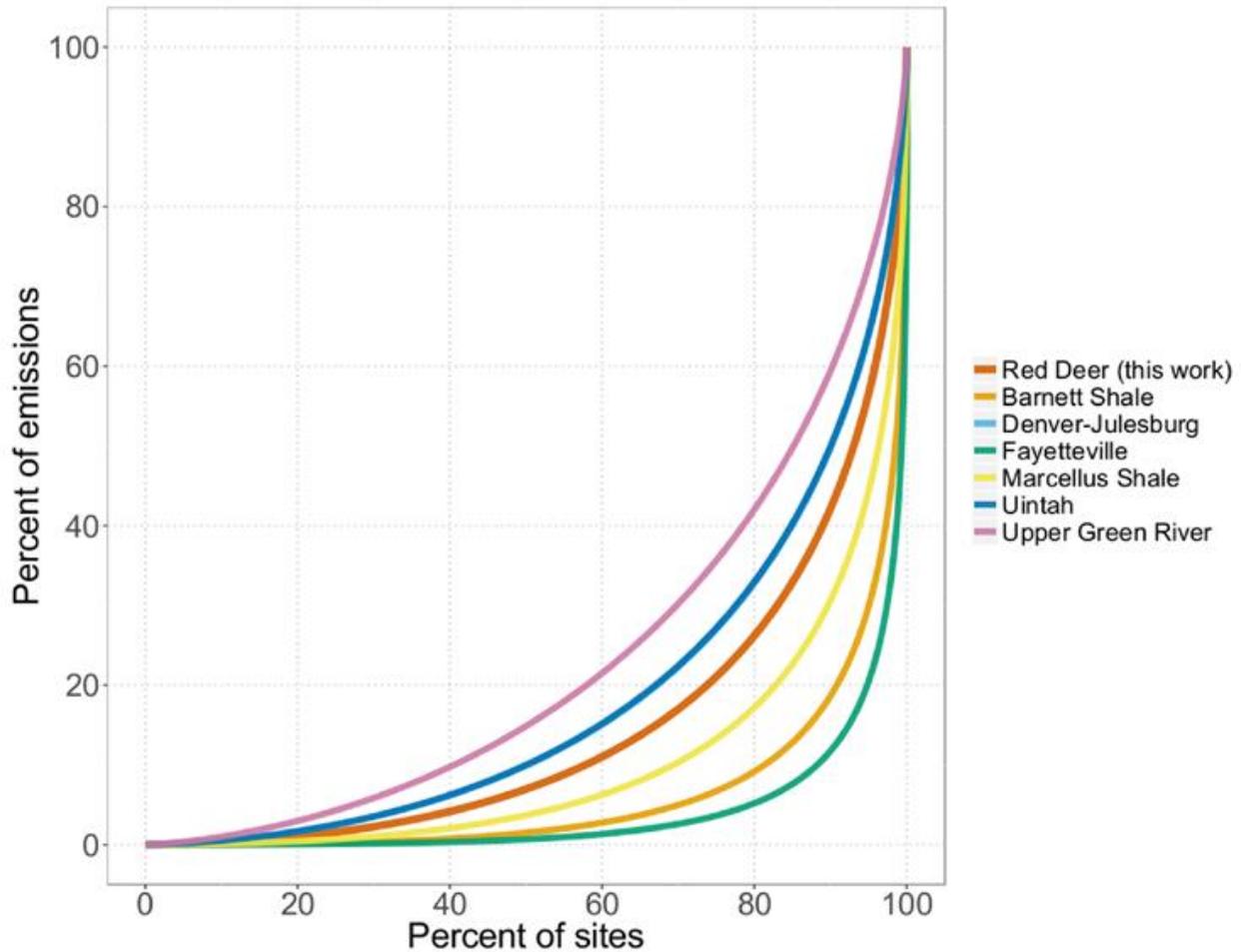
Figure 3.3. Lorenz curve of the distribution of emissions from emitting sites in various oil and gas producing regions in the United States and Canada where ground-based measurements have been reported. Source: Reproduced[4] from Zavala-Araiza et al. (2018).

Multiple studies suggest that super-emitters and skewed emissions distributions are among the primary reasons for discrepancies between top-down and bottom-up emissions inventories (Alvarez et al. 2018; Cui et al. 2019b). Difficulties in measuring super-emitters stem from their stochastic nature, in both spatial and temporal distribution, and relationship of emissions to the age of a site and overall production volumes (Zavala-Araiza et al. 2018). In a study outside the timeframe of our review, high-frequency measurements in the Permian Basin found that approximately 90% of high-emitting infrastructure displayed intermittent emissions activity (Cusworth et al. 2021a). Ground- and vehicle-based studies with small samples sizes or temporally-limited measurements may not capture these heavy-tailed

---

[4] This material is in the public domain.

PSE

emission distributions, resulting in underestimates in emissions factors and resulting emission estimates (Fischer et al. 2017; Omara et al. 2016). Frequent monitoring, modelling studies, and probability emission distributions may be key to closing the gap between bottom-up and top-down emission estimates caused by skewed emissions distributions (Zavala-Araiza et al. 2018).

Studies suggest that super-emitters represent "low hanging fruit"—with potential for large reductions in methane emissions—and should be prioritized for mitigation and reduction. (Zavala-Araiza et al. 2015b) calculated that the reduction of emission from super-emitters, which they defined as the top 15% of emitting sites by proportional loss rate, to mean levels of the lower 85% of emitting sites, would result in an overall reduction in total supply chain emissions by 65–87%.

PSE

Table 3.2. Percentage of sites or observations responsible for a given percentage of emissions from studies that identified upstream super-emitters or skewed emission distributions.

Percent of sources contributing to...	Percent of emissions	Component or sector	Region	Method	Study
15	70	Wellpads	Permian Basin – NM	Vehicle	Robertson et al. 2020*
5	30				
20	60	Point source emissions	Permian Basin – TX, NM	Aerial	Cusworth et al. 2021a*
5	65	Flares	Permian Basin – TX, NM	Aerial	Lyon et al. 2021*
10	77	Unconventional wellpads	Marcellus Shale – PA	Vehicle	Caulton et al. 2019
5	66				
5	50	Wellpads and ancillary surface equipment	Marcellus Shale – PA; Eagle Ford Shale– TX; Pinedale Field – WY; Uinta Basin – UT; Upper Green River Basin – WY; Barnett Shale – TX; Denver-Julesburg Basin – CO; Fayetteville Shale – AR	Ground	Omara et al. 2018
20	78	All wells and wellpads	Marcellus Shale – PA	Vehicle	Omara et al. 2016
17	50	Conventional natural gas wells			
23	85	Unconventional natural gas wells			
6.6	50	Wellpads	Barnett Shale – TX	Vehicle	Rella et al. 2015
7.7	56	Upstream - unspecified	Bakken Shale – ND	Aerial	Englander et al. 2018
10	58–65	Oil and gas production sites	Alberta, Canada	Vehicle	Zavala-Araiza et al. 2018
20	74–79				
10	73	Connectors, flanges, valves, etc.	Permian Basin – TX,NM; Anadarko Basin – OK, KS, TX; Gulf Coast Basin – TX, LA, MS; San Juan Basin – NM, CO;	Ground	Pacsi et al. 2019

PSE

Percent of sources contributing to…	Percent of emissions	Component or sector	Region	Method	Study
25	79	Wellpads	Fayetteville Shale – AR	Aerial and ground	Schwietzke et al. 2019
21	72	Abandoned wells in coal areas	PA	Ground	Kang et al. 2016
32	74	Abandoned gas wells			
20	28	Conventional developments	Bakken Shale – ND; Weyburn-Midale Field – Saskatchewan, Canada	Vehicle	Baillie et al. 2019
20	49	Unconventional developments			
20	72–83	Well pads	Upper Green River Basin – WY; Wattenberg Field, Denver-Julesburg Basin – CO; Greater Natural Buttes Field, Uintah Basin – UT; Fayetteville Shale – AR	Vehicle	Robertson et al. 2017
20	54	Well pads	Upper Green River – WY		
30	80	Gathering and processing	Appalachian Basin – PA, NY, WV; San Juan Basin – NM, CO; Uintah Basin – UT; Green River Basin – CO, WY; Powder River Basin – WY; Anadarko Basin – OK, KS, TX; Permian Basin – TX, NM; Fort Worth Syncline – TX; Mid-Gulf Coast Basin – TX, LA, MS; East Texas Basin – TX, LA; Arkoma Basin – AR, OK	Vehicle	Mitchell et al. 2015

PSE

Percent of sources contributing to…	Percent of emissions	Component or sector	Region	Method	Study
2.5	67	Unconventional gas wells	PA	State and operator data	Ingraffea et al. 2020
2.5	62	Conventional oil wells			
25	59	Unconventional combined oil and gas wells			
25	69	Coalbed methane wells			
16	88	Production facilities	Fayetteville Shale – AR	Simulated	Bell et al. 2017
3.7	80	Flares (natural gas production)	US	GHGRP data	Allen et al. 2016
7.8	87	Flares (associated gas)			
25	69	Flares	Bakken Shale – ND	Airborne	Gvakharia et al. 2017
20	70	pneumatic controllers, chemical injection pumps, equipment leaks, compressors, water tank flashing, condensate/oil tank flashing, liquid unloadings, dehydrators	Barnett Shale – TX	Monte Carlo simulation	Zavala-Araiza et al. 2017
19	95	Pneumatic controllers	US	Ground	Allen et al. 2015a
20	83	Wells without plunger lifts	US	Ground	Allen et al. 2015b
20	65–72	Wells with plunger lifts			
6.6	50	Well pads	Barnett Shale – TX	Vehicle	Rella et al. 2015
22	80				

PSE

Percent of sources contributing to…	Percent of emissions	Component or sector	Region	Method	Study
Studies that included other parts of the supply chain					
10	49–66	Wellpads, pipeline leaks, storage tanks, gas processing	San Juan Basin – NM,CO	Aerial	Frankenberg et al. 2016
2	50	Production sites, compressor stations, processing plants	Barnett Shale – TX	Modeling	Zavala-Araiza et al. 2015a
10	90				
10	60	All methane emitting industries	CA	Aerial	Duren et al. 2019
5	40–60	Total oil and gas supply chain	Global	Modeling	Balcombe et al. 2018
2	19	Upstream and midstream	Barnett Shale – TX	Ground	Lyon et al. 2015
7.5	60	Natural gas processing plants, compressor stations, production well pads	Barnett Shale – TX	Ground	Yacovitch et al. 2015
10	50	Midstream - Compressor stations in the Transmission and Storage segment	16 U.S. states	Vehicle (tracer-tracer)	Subramanian et al. 2015
15	50			On-site	

*Study was outside the cutoff dates for our systematic review

PSE

Monitoring methodologies and source apportioning

A general summary of different methods for measuring emissions and source apportioning is provided in Table 3.3.These measurement methods are not unique to upstream studies; many are also used in midstream and downstream studies. However, the vast majority of studies with primary data collection focused on the upstream sector or include upstream facilities and sites. As such, methodologies are primarily discussed in this section.

From 2015-2020, significant advancements have been made in monitoring methodologies and technologies to reduce uncertainties in upstream emission estimates. A hybrid approach of combining LDAR programs with rapid screening methods is a promising solution to reduce costs associated with LDAR programs. Aerial-based hyperspectral imaging systems and recently launched satellite-based instruments have the potential to quantify methane emissions from point sources at greater scales and with higher resolution. However, there needs to be continued research and evaluation of emerging methane detection and monitoring technologies to demonstrate emissions reductions equivalent to existing regulatory approaches (Ravikumar et al. 2019). Continued research will also reduce uncertainties between approaches and the collection of additional field data will help close the gap between top-down estimates and bottom-up component-based inventories (Bell et al. 2017). As with all improvements to methodologies for data gathering and monitoring, there is a trade-off between added value and cost (Turner et al. 2016).

Table 3.3. Summary of emissions monitoring and measurement approaches. Adapted[5] from Hopkins et al. (2016) and Caulton et al. (2018).

Approach	Key Characteristics	Advantages	Disadvantages	Examples
Ground-based onsite component-level sampling	Techniques may use chambers, enclosures, or probes to capture and measure emissions from individual components. Requires direct access to sites.	Direct flux measurements. Accurate component level emissions.	Possible issues due to inaccessible sources, safety requirements, or impractical sampling requirements. Chamber-based applications limited by shape. Probe samples may not be representative.	High flow meters, chamber- or tent-based methods. Handheld flame ionization detectors (FID)
Vehicle-based mobile dispersion	Vehicle-based measurements downwind of facilities along public or access roads. Measurements taken at heights ranging from 1–4 m off the ground, at typical frequencies of 1–2 Hz.	Source attribution with additional tracer measurements. Allows for point source measurements over large scales. Cheaper deployment than aerial based methods.	Limited temporal coverage. Requires access to roads nearby sites. Requires modeling to determine emissions flux	Aerodyne mobile laboratory, modified pickup-trucks and ATVs, Princeton atmospheric chemistry experiment (PACE)
Tower-based monitoring	Stationary towers with measurement equipment. Single tower or network of multiple towers.	Continuous, long-term in situ measurements can detect trends in time. Source attribution possible with additional tracer measurements	Requires measurements of planetary boundary layer height, meteorology, and inverse models to calculate flux. Height and location determines footprint; access to appropriate sites is challenging. Limited spatial coverage	California Statewide GHG Monitoring Network

---

[5] Adapted as permitted by the creative commons license. Changes were made to the formatting of the original work in order to generate this figure.

PSE

Approach	Key Characteristics	Advantages	Disadvantages	Examples
Aerial-based	Measurement of methane and other gasses from the air. Multiple upwind and downwind transects are typically performed. Often used in conjunction with mass balance or atmospheric modeling approaches.	Large spatial coverage. Allows for vertical profiling. Potential to detect fugitive emissions. Source attribution possible with additional tracer measurements.	Infrequency and expense results in limited temporal coverage. Requires atmospheric data and transport modeling. Use of airspace requires air traffic control permission/approval	Planes, helicopters, drones
Satellite-based remote sensing	Utilizes solar backscatter or thermal imagery techniques to estimate atmospheric CO_2 and CH_4 columns.	Dense and continuous coverage over regional or global scales	Limited by cloud cover, target revisit frequency, pixel resolution, and ability to resolve on smaller scales. Limited source apportioning.	GOSAT, GOSAT-2, TROPOMI, PRISMA, SCIAMACHY, GHGSat
Tracer-tracer	Release of tracer compound(s) at the location of CH_4 or HDAP emissions. Tracer concentrations measured downwind.	Eliminates the need for complex atmospheric modeling. Ability to quantify flux from a point source in areas with complex methane signals	Requires site access and proper release height. Require access for downwind measurements. Used for site characterization; does not identify specific sources. Does not work for co-located sources.	Tracers: N_2O, C_2H_2, SF_6
Isotopic measurement	Utilized for source attribution. Thermogenic methane from oil and gas is relatively enriched in $^{13}CH_4$ compared to biogenic methane.	Allows for apportionment of biogenic vs. fossil sources	Requires isotopic signatures for individual source categories to improve constraints. Cannot distinguish between midstream and downstream sources.	$\delta^{13}C\text{-}CH_4$, $\delta^2H\text{-}CH_4$, $\delta D\text{-}CH_4$,

PSE

Approach	Key Characteristics	Advantages	Disadvantages	Examples
Ethane:methane, propane:methane, and ammonia:methane enhancement ratios	Utilized for upstream source attribution for specific producing formations and/or oil and gas production infrastructure. Ratio of C_2+ alkanes in natural gas varies according to thermal maturity and region. Biogenic methane has relatively low ethane.	Allows for apportionment of biogenic vs. fossil sources. Ammonia (NH_3) is used for agricultural and concentrated animal feeding operation apportioning.	Requires signatures for individual sources to improve constraints. Source signatures may change over time. Other sources of co-emitted species may not be well constrained.	EM27/SUN spectrometer, Solar Occultation Flux instrument, in-situ measurements
Optical gas imaging (OGI)	Use of infrared and other optical techniques for identifying sources of leaks. Used from either the ground or from the air. Often used in combination with other ground-based techniques for quantitative measurements.	Plume visualization and ability to pinpoint emissions sources. Precise evolution of emission sources over repeat measurements.	Sensitivity to different hydrocarbons varies (Lyon et al. 2016). Detection limits vary with wind speed. Poor quantitative measurements and scaling. Detection rates can vary according to operator experience.	Infrared imaging, FLIR
Hyperspectral imaging systems	Aerial-based infrared imaging spectrometers	Plume visualization and quantification. Sufficient resolution to detect point sources. Allows for real-time identification in some cases.	Limited commercial availability.	AVIRIS-NG, GAO, Kairos LeakSurveyor

PSE
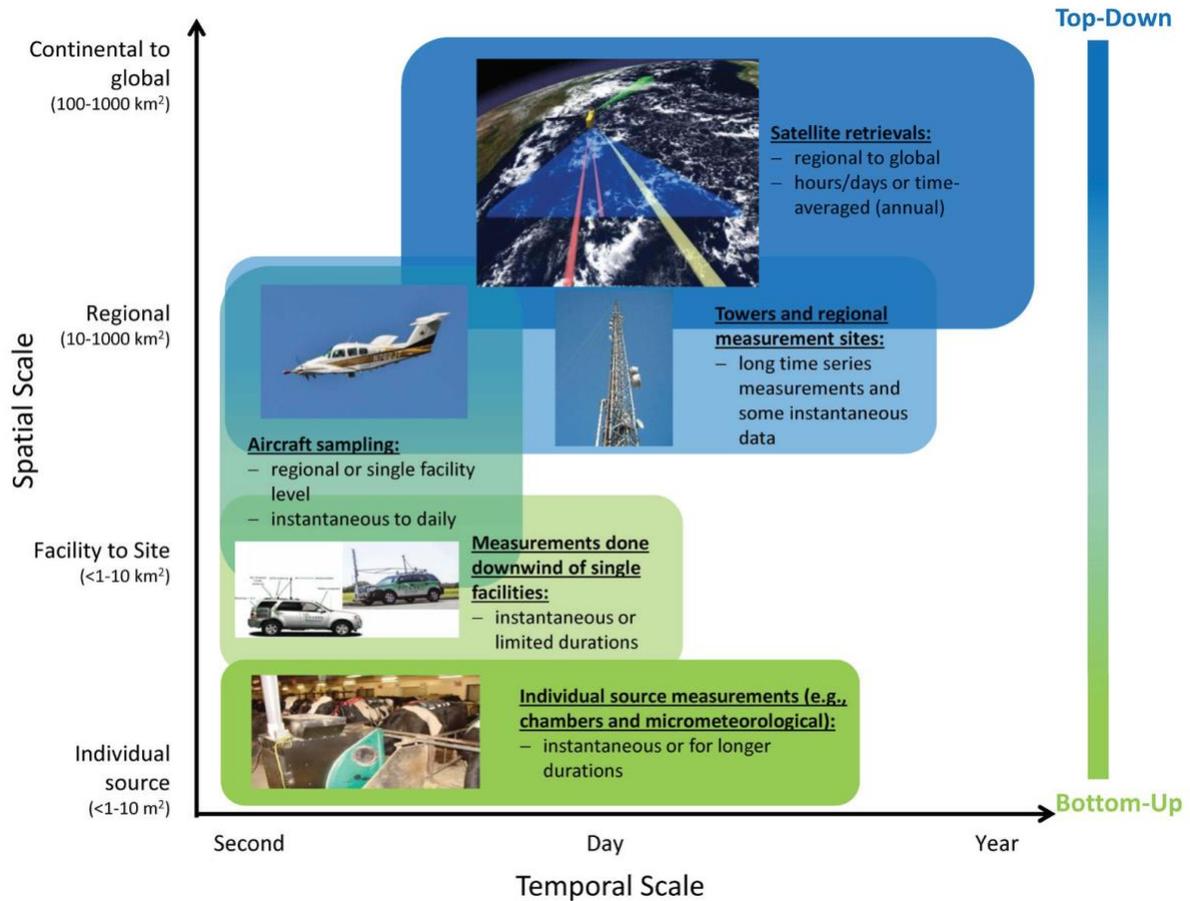
Figure 3.4. Spatial and temporal scale of various methane measurement platforms. Source: Reproduced with permission[6] from National Academies of Sciences, Engineering, and Medicine (2018).

Optical gas imaging and LDAR

A number of jurisdictions in the United States and Canada currently require the use of optical gas imaging (OGI) in LDAR programs (Johnson et al. 2021). In a study that was outside the timeframe of our review, Ravikumar et al. (2020) investigated the effectiveness of OGI-based LDAR programs in Alberta, Canada. LDAR programs were shown to be effective, with over 90% of leaks found in the initial survey not observed during the follow-up survey. However, only a

---

[6] Republished with permission from National Academies of Sciences, Engineering, and Medicine, from "Improving Characterization of Anthropogenic Methane Emissions in the United States". National Academies of Sciences, Engineering, and Medicine 2018. The National Academies Press: https://doi.org/10.17226/24987. Permission conveyed through Copyright Clearance Center, Inc.

PSE

22% reduction in fugitive emissions was observed due to new leaks occurring between surveys, supporting the need for frequent, low-cost LDAR surveys.

A recent study by Kemp and Ravikumar (2021) that falls outside the timeframe of our review analyzed LDAR programs and monitoring technologies using the Fugitive Emissions Abatement Simulation Toolkit (FEAST). They found that, dependent on the emission-size distribution for a specific region, tradeoffs can be made between survey frequency and detection thresholds for LDAR programs while maintaining equivalent emissions mitigation. Median detection thresholds represent upper and lower bounds for emission mitigation, as lower detection thresholds will not proportionally increase mitigation due to skewed emissions distributions caused by super-emitters. Additionally, higher detection thresholds will not detect smaller emissions, even with increased survey frequency. Balancing survey frequency and detection thresholds as well as being able to differentiate between sites with vented emissions—where there is little mitigation benefit—and sites with fugitive emissions will increase the cost effectiveness of LDAR programs (Kemp and Ravikumar, 2021).

One proposed solution to frequent LDAR surveys while minimizing costs is to use a multi-platform hybrid screening and confirmation approach, whereby rapid screening technologies—such as vehicle-, aerial-, or satellite-based platforms—can be used to guide ground-based OGI LDAR programs (Fox et al. 2019; Schwietzke et al. 2019). Methane monitoring technologies vary on both spatial and temporal scales (Figure 3.4) and can complement each other when used in combination. Because a relatively small number of leaks are responsible for a disproportionate amount of emissions, a rapid screening approach that targets high-emitters could be a viable alternative to current periodic inspection requirements (Schwietzke et al. 2019). Evaluation and comparison of various methane detecting technologies suggest that although technologies like fixed sensors, mobile laboratories, unmanned aerial vehicles (UAVs), aircraft, and satellites are promising for use in hybrid screening approaches, additional research is needed to improve detection, quantification, and to establish equivalent emissions reduction under various environmental and operational conditions (Fox et al. 2019; Ravikumar et al. 2019).

Pacsi et al. (2019) compared the performance of OGI and handheld flame ionization detector (FID) technologies in a ground-based measurement campaign of emissions from components of production, gathering, and boosting sites in the Permian (Texas/New Mexico), Anadarko (Oklahoma/Texas), Gulf Coast (Texas/Louisiana), and San Juan (New Mexico/Colorado) Basins. They found that OGI- and FID-based methods, two of the primary methods for detecting leaking components, identified different populations, with OGI detecting only 33% of leaks identified by FID-based methods. However, total emissions from the components identified by each method were similar overall. Under the study conditions, FID-based methods used a 500 parts per million threshold for leak definition and identified a much larger percentage of small leaks that had minor contributions to overall measured emissions, while OGI uniquely identified a few high emitters that FID methods did not. The discrepancy between the two methods may be due to OGI being used from elevated locations and in areas that FID surveys

did not include, or due to insufficient dispersion near the FID probe location (Pacsi et al. 2019). Because FID and OGI methods detected two different populations of emitters, but overall similar emissions, a comparison of methods based simply on the number of leaks detected, or defining a leak threshold for equivalency between the two methods, would be inadequate (Pacsi et al. 2019).

Vehicle-based mobile labs

Rella et al. (2015) developed and successfully deployed a vehicle-based mobile flux plane technique to measure total methane emissions downwind of sites as part of the Barnett Shale Coordinated Campaign. Caulton et al. (2018) measured methane downwind of wellpads in the Marcellus Shale using mobile Gaussian plume methods. This method requires longer sampling times, more transects (suggested at least 10 transects per site), and increased weather and atmospheric monitoring to constrain uncertainties from atmospheric variability associated with plume modeling, but is still less time intensive than multi-transect large eddy simulation methods (Caulton et al. 2018). Similar mobile inverse Gaussian plume techniques were utilized by MacKay et al. (2019) in southeastern Saskatchewan, Canada, and Zavala-Araiza et al. (2018) in Alberta, Canada.

Vehicle-based studies are often limited by operator cooperation and/or access to production sites and private roads (Atherton et al. 2017; Goetz et al. 2015; Lan et al. 2015; O'Connell et al. 2019; Omara et al. 2016; Rella et al. 2015; Robertson et al. 2020; Vaughn et al. 2017; Yacovitch et al. 2015). These studies are often restricted to conducting measurements from public roads downwind of facilities, limiting the ability to measure from a randomized and representative set of sites. Vehicle-based studies are further limited by the height at which data collection occurs and may be unable to fully capture emissions from tall infrastructure (O'Connell et al. 2019).

Bell et al. (2017) compared three measurement methods at natural gas production pads in the Fayetteville region: OGI followed by Hi-flow meters; dual tracer flux; and EPA Other Test Method 33A (OTM33A). OGI and Hi-flow measurements were combined with simulated sources based on estimated emissions sources to form a study onsite estimate and represents a lower bound of emissions for most cases. OTM33A is overall less accurate than tracer release methods, generally underestimating emissions overall compared to the study onsite estimate, but is relatively quick, non-invasive, and also requires downwind access to sites (Bell et al. 2017). Tracer methods generally agree with study onsite estimates, but overestimates emissions overall, and allow for the measurement of high-rate emission sources (Bell et al. 2017).

Aerial-based imaging

Emerging aerial-based hyperspectral imaging systems are useful for quickly identifying large point sources from broad geographic areas with known regional enhancements (Frankenberg et al. 2016; Thorpe et al. 2017). The aerial-based Airborne Visible/Infrared Imaging

Spectrometer - Next Generation (AVIRIS-NG) has demonstrated the capability to identify point sources of CO_2 and CH_4 from the oil and gas sector (Thorpe et al. 2016, 2017). Frankenberg et al. (2016) used AVIRIS-NG and the Hyperspectral Thermal Emission Spectrometer (HyTES) to successfully identify and quantify 250 methane plumes from point sources in the San Juan Basin in the Four Corners region, an area with known methane enhancements based on satellite observations but with limited road access. Duren et al. (2019) also used AVIRIS-NG to survey more than 272,000 infrastructure elements (including landfills and dairies) in California and detect methane plumes from point sources. In a more recent study outside the timeframe of our review, Cusworth et al. (2021a) used AVIRIS-NG and a similar Global Airborne Observatory (GAO) hyperspectral imaging system to quantify large methane emitters in the Permian Basin. AVIRIS-NG allows for real-time aerial-based plume identification that can be relayed to ground teams to investigate the source of leaks for repair (Duren et al. 2019; Frankenberg et al. 2016; Thorpe et al. 2017). In another study outside the timeframe of our review, a controlled release evaluation of the Kairos LeakSurveyor, another aerial-based hyperspectral imaging system, suggests that under suitable conditions this system may be effective at the rapid detection of upstream and midstream super-emitters (Sherwin et al. 2021). Studies of Bridger Photonics' Gas Mapping LiDAR (GML), another emerging aerial-based technology, suggest it has comparable performance to OGI systems under appropriate conditions (Johnson et al. 2021) and has been used in combination with ground-based surveys to characterize emissions sources in British Columbia, Canada (Tyner and Johnson, 2021).

A pilot study in the Fayetteville Shale by Schwietzke et al. (2019) examined the use of aerial infrared CH_4 column imaging and in-situ mole fraction detection to identify leaks and guide ground-based OGI and Hi-flow LDAR programs. These aerial detection technologies are relatively small and inexpensive—facilitating commercial scaling—compared to AVIRIS-NG and GAO, which are relatively expensive and primarily used for scientific studies. Depending on method minimum detection limits (MDLs), aerial-guided LDAR was successful at detecting leaks that accounted for half or more of all fixable methane emissions and identified an order of magnitude greater fixable methane emissions per facility compared to ground-based methods. However, aerially-guided methods were limited by their inability to differentiate between fixable and non-fixable emissions and inherent delays between aerial measurements and ground-based verification (Schwietzke et al. 2019). Aerially-guided LDAR approaches are expected to be more cost-effective in wet gas regions, which exhibit larger overall emissions and more potential sources (e.g., liquid storage tanks), and in regions with larger distances between facilities (Schwietzke et al. 2019).

Satellite remote sensing

Since 2015, multiple satellites and satellite-based instruments capable of detecting and quantifying methane at high resolutions have been deployed, including: the GHGSat-D in 2016; GHGSat-C1 in 2020; GHGSat-C2 in 2021; the TROPOspheric Monitoring Instrument (TROPOMI) in 2017; the Greenhouse Gases Observing Satellite-2 (GOSAT-2) in 2018; and the PRecursore IperSpettrale della Missione Applicativa (PRISMA) in 2019. Additional details regarding these

satellites and instruments are summarized by Jacbo et al. (2016) and Cusworth et al. (2019). As new satellites and instruments were deployed, other instruments have been decommissioned, such as the Hyperion Imaging Spectrometer, which was decommissioned in 2017.

Despite recent satellite launches, many studies from 2015–2020 that utilize satellite retrieval data use previous generation GOSAT and SCanning Imaging Absorption spectroMeter for Atmospheric CHartographY (SCIAMACHY) retrieval data. Inversions of GOSAT satellite data have been used to estimate methane emissions on a country (Janardanan et al. 2020) and global scale (Maasakkers et al. 2019). Using surface observations and GOSAT data from 2010–2014, Turner et al. (2016) estimated a 30% increase in North American methane emissions from 2002–2014 and suggested the increase was primarily due to emissions from the oil and gas sector. Bruhwiler et al. (2017) argued that this apparent increase could have been due to atmospheric variability, sampling biases, choice of background, and the short time period of the GOSAT data. In response, Sheng et al. (2018) analyzed GOSAT data from 2010–2016 over North America. Although they found a decrease in Mexican emissions, no long-term trend in Canadian emissions, and a 2.5±1.4% increase in U.S. emissions, their analysis was still limited by the length of the GOSAT record and impacts of atmospheric transport. Within the United States, satellite data from GOSAT and SCIAMACY were used to estimate annual emissions from the southern San Joaquin Valley and the Four Corners region (Buchwitz et al. 2017).

Data from TROPOMI, which has a resolution of 7 km (4.3 mi), has been used to estimate emissions from the Permian Basin (Zhang et al. 2020) and emissions from a well blowout event in Ohio (Pandey et al. 2019). Multiple studies outside the timeframe of our review also utilized TROPOMI data. A study by Irakluis-Loitxate et al. (2021) used TROPOMI, along with China's Gaofen-5 and ZY1 satellites, to survey point sources in the Permian Basin. They found over 37 plumes with methane emission rates of >500 kilograms/hour (kg/hr) and that newer facilities are major emitters due to inefficient flaring operations. Another study used TROPOMI to identify oil and gas methane "ultra-emitters" in multiple countries. They found that "ultra-emitters" in the United States following a power-law distribution and were responsible for 5% of annual inventory emissions, excluding the Permian Basin (Lauvaux et al. 2021). Lyon et al. (2021) combined aerial surveys, TROPOMI, and tower-based observations to determine the effects of the COVID-19 pandemic on emissions from the Permian Basin. TROPOMI retrieval data was also used by Shen et al. (2021) and Zavala-Araiza et al. (2021) to quantify emissions from high-producing regions in Mexico.

The GHGSat-D and PRISMA have sub-kilometer spatial resolutions of 50 m (164 ft) and 30 m (98 ft), respectively, sufficient resolution to detect large point source emissions (Jacob et al. 2016; D. Varon et al. 2019). No studies captured in our systematic review examined GHGSat-D or PRISMA retrieval data. However, multiple studies outside the timeframe of our review utilize both GHGSat-D and PRISMA data. Varon et al. (2019; 2020) used GHGSat-D observations to identify large oil and gas production methane point sources in Asia and to quantify time-averaged methane emissions from a single coal mine vent in New Mexico. Cusworth et al. (2021b) combined observations of varying temporal and spatial scales from TROPOMI,

PSE

GHGSat-D, PRISMA, and the Visible Infrared Imaging Radiometer Suite (VIIRS) to quantify emissions from a well blowout in the Permian Basin. Early analysis of retrieval data suggest that PRISMA may be most effective at detecting point emissions in regions with bright, homogeneous surfaces, compared to regions with surface structures such as the Permian Basin (Guanter et al. 2021).

Advancements in satellite-based remote sensing have increased effective spatial resolution and allowed for the identification and quantification of large point sources on larger scales. Lower resolution satellite observations, such as those from TROPOMI, can identify methane hotspots and direct higher-resolution observations that can identify specific point source emissions for ground-based LDAR (D. Varon et al. 2019). Satellite remote sensing, when used in combination with surface- and ground-based measurements as part of a multi-tiered monitoring approach, allows for increased temporal and spatial sensitivity to guide sub-basin scale decision and policy (Cusworth et al. 2020). Future proposed satellite launches—such as MethaneSAT, NASA's GeoCarb, the French-German MERLIN, and Japan's GOSAT-GW—are expected to increase methane remote sensing capabilities and support future research.

Source apportioning

One method of source apportioning is to utilize in-situ ratios of alkanes, typically ethane:methane (C_2H_6:CH_4) or propane:methane (C_3H_8:CH_4), to differentiate between methane from the oil and gas sector and methane from biogenic sources (e.g., landfills, wastewater treatment plants, etc.). Ethane:methane and propane:methane ratios have been shown to be variable between regions (Cardoso-Saldaña et al. 2019; Peischl et al. 2015, 2018); within the same region (Cardoso-Saldaña et al. 2019; Townsend-Small et al. 2015); and along the natural gas supply chain (Allen et al. 2017), and have been shown to change overtime (Lan et al. 2019). Studies such as Franco et al. (2016) and Helmig et al. (2016) that use static ethane:methane enhancement ratios suggest significant increases in methane emissions in North America from oil and gas sources since 2007. However, Lan et al. (2019) found that the rate of increase in methane emissions from oil and natural gas sources is much smaller on a relative basis than emissions of both ethane and propane (Figure 3.5), reflecting the fundamental heterogeneity of the oil and gas sector. Due to changing ratios, studies that rely on static or non-region specific ethane:methane and propane:methane ratios may result in an overestimation of CH_4 emissions from biogenic sources, depending on the region (Barkley et al. 2019a; Lan et al. 2019; Townsend-Small et al. 2016a). Bruhwiler et al. (2017) found no statistically significant increase in North American methane emissions from 2000–2014, and although Lan et al. (2019) found an increase in emissions, the increase was much smaller than estimates that relied on static ethane:methane ratios. Similarly, Barkley et al. (2019a) found that discrepancies between methane-based and ethane-based estimates for a multi-basin top-down study covering the Anadarko, Eagle Ford, Fayetteville, Haynesville (Louisiana/Texas), Permian, and Barnett Basins were likely due to the assumed ethane:methane ratios being underestimates. This suggests previous studies that found increases in methane emissions over the course of years, but utilized static enhancement ratios, significantly overestimated emissions. Furthermore, spatial

and temporal differences in enhancement ratios reduce the reliability of using alkanes alone for source apportioning and estimating methane on a continental scale (Lan et al. 2019).

In addition to alkane compounds, non-alkane compounds have been used for methane source apportioning. Kille et al. (2019) used a network of ground-based solar absorption sensors to infer vertical column density of methane, ammonia (NH_3), and ethane for source apportioning in the Colorado Front Range. They found general agreement with past in situ-based measurements of the same region, with the advantages of solar absorption methods being measurements are independent of boundary layer height, the avoidance of sampling inlets, and good vertical sensitivity across the entire atmosphere. Ammonia measurements are generally less common than ethane, and although ammonia has been shown to be useful for apportioning agricultural and livestock sources, it has a relatively short atmospheric half-life which may impact transport distances and source apportionment (Kille et al. 2019).

Despite uncertainties associated with the use of ethane and propane to determine long-term trends in oil and gas methane emissions, they are still a useful tool to differentiate between multiple co-located methane sources that have unique signatures (Barkley et al. 2019b; Lan et al. 2019). Source apportioning studies in the Barnett Shale have found high variability in methane, but low variability in ethane (Allen et al. 2017; Karion et al. 2015; Smith et al. 2015). Studies suggest that this indicates that biogenic emissions, which are low in ethane compared to oil and gas emissions, are highly variable and a source of uncertainty in measurements. Variability due to changing alkane enhancements is unlikely due to the relatively short time frame of the study and the development of study specific ratios (Karion et al. 2015; Smith et al. 2015). However, a study by Allen et al. (2017) suggests that variability may be due to midstream sources, which also have low ethane:methane ratios in comparison to upstream sources. In the Marcellus, coalbed natural gas directly from coal mines or from wells drilled through coalbed layers are another potential source of low ethane:methane ratio emissions (Ren et al. 2019). Plume targeting aerial-based mass balance approaches are unable to differentiate emissions from multiple co-located sources if separate profiles are unavailable for each source (Baray et al. 2018).

In addition to alkanes, isotopic signatures including hydrogen-2 methane (δ^2H-CH_4), carbon-13 methane ($\delta^{13}C$-CH_4), and deuterated methane (δD-CH_4) have been used for methane source apportioning. Schwietzke et al. (2016) compiled an isotopic methane source signature database and found that fossil fuel based methane emissions are not increasing over time, but are underestimated by 20–60% on a global scale by current inventories. Isotopic measurements in the Four Corners region of New Mexico were used to differentiate between methane plumes from coal and natural gas extraction; high methane concentration events were from coal beds (Arata et al. 2016). Yang et al. (2019) measured $\delta^{13}C$-CH_4 to differentiate between anthropogenic sources in the Barnett Shale and Greater Houston areas. Repeated measurements of specific sources revealed natural variations in $\delta^{13}C$-CH_4 ratios, contributing to overall uncertainty. In the Colorado Front Range, Townsend-Small et al. (2016a) found that δ^2H-CH_4 signatures were the most consistent for landfills, cattle feedlots, and oil and gas wells,

as opposed to $\delta^{13}C$-CH_4 signatures. In the Barnett Shale region, δD-CH_4 was the most consistent tracer for natural gas production emissions, compared to alkanes and $\delta^{13}C$-CH_4 (Townsend-Small et al. 2015).

Due to uncertainties caused by inter- and intra-regional variability with both alkane- and isotope-based source apportioning, combining both tracer methods may better constrain source attribution (Allen, 2016b; Yang et al. 2019). Additional spatially resolved source signature profiles will be necessary for isotope- and alkane-based apportioning approaches to constrain multiple co-located sources and sectors with high uncertainty in the methane budget, clarify the roles of anthropogenic versus natural emissions in methane budgets, and help close the gap between methane observations and inventories (Townsend-Small et al. 2016a; Turner et al. 2016; Yang et al. 2019).
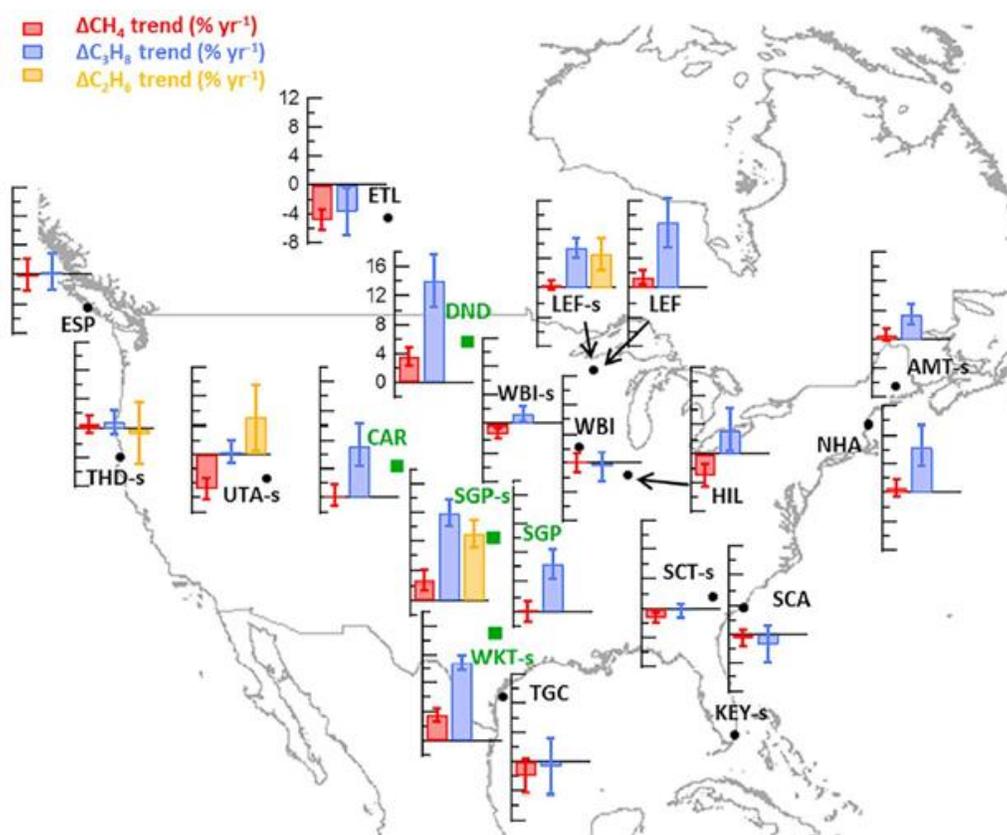
Figure 3.5. Trends in the enhancement of methane (CH_4), ethane (C_2H_6), and propane (C_3H_8) over North America from 2006–2015. Black dots represent non-oil and gas sources; green dots represent oil and gas sources. Each vertical tick represents a 2% change. Error bar shows 1σ uncertainty. Source: Reproduced[7] from Lan et al. (2019).

---

[7] Reproduced as permitted by the creative commons <u>license</u>. No changes were made to the content of the reproduced figure.

PSE

Orphaned and abandoned wells

Abandoned wells were not included in U.S. EPA GHGI until 2018. In 2018, the U.S. EPA added abandoned oil and gas wells to the GHGI based on a limited number of studies that suggested abandoned oil and gas wells contribute a small (2–4%) but still significant percentage of emissions for both regional and national inventories (Brandt et al. 2014; Kang et al. 2014, 2016; Townsend-Small et al. 2016b; US EPA, 2018). Since then, additional studies have been published that increase the state of knowledge around abandoned wells (Table 3.4). It is likely that upstream studies that measure emissions on a regional or basin scale also capture emissions from abandoned wells that are co-located with active wells and other production infrastructure. However, these studies typically do not apportion emissions on a component level or specifically identify abandoned wells as a source and are not included in Table 3.4.

Emissions from unplugged abandoned wells are on average higher than emissions from plugged wells (Kang et al. 2016, 2019; Riddick et al. 2019). Riddick et al. (2019) found average methane emissions from plugged and unplugged abandoned wells in West Virginia to be 0.1g/hr and 3.2 g/hr, respectively, compared to 138 g/hr for active conventional wells. Unplugged abandoned wells that were abandoned between 1993–2015 had much higher emissions (16 g/hr) compared to those abandoned before 1993 (3×10^{-3} g/hr) (Riddick et al. 2019). A study in Pennsylvania found that high-emitting abandoned wells tended to be unplugged abandoned gas wells or plugged and vented gas wells in coal areas, defined as wells that overlap a workable coal seam (Kang et al. 2016). Similarly, Townsend-Small et al. (2016b) found emissions from plugged wells were significantly lower than unplugged wells, although positive fluxes were only identified in one plugged and eight unplugged wells out of the 138 sampled abandoned wells. Kang et al. (2016) estimated abandoned wells contributed 5–8% of anthropogenic methane emissions for 2011 in Pennsylvania, while Townsend-Small et al. (2016b) estimated abandoned wells contributed 1.9–4.3% of U.S. national emissions in 2013.

Like other sectors, emissions from abandoned wells follow skewed emissions distributions and can be highly variable within the same geological formations (Kang et al. 2016; Riddick et al. 2020; Townsend-Small et al. 2016b). Riddick et al. (2020) measured emissions over a 24-hour period and observed variation in emissions ranging from 1.1 to 142, with a mean factor of 18. They did not find correlation between emission variability and magnitude of emission, temperature, relative humidity, or atmospheric pressure. They suggested that short-lived emissions sampling studies (i.e., <1 hr) may miss high-emitting events, leading to underestimates in emissions inventories, and recommended a sampling period of at least three hours to account for observed emissions variability. Kang et al. (2016) found that high-emitting abandoned wells (≥10 g/hr) in Pennsylvania emit methane at consistently high levels in multiyear measurements, while lower emitting wells had greater variability.

Using operator-reported data, Wisen et al. (2020) found that among seven leaking abandoned wells in Canada, the majority of cases (57%) were due to leakage originating from uncemented intervals below the surface casing. They noted that leakage from abandoned wells is likely underreported due to wells missing from leakage databases. Other factors, such as regulations

requiring venting in coal areas, are also likely the cause of some high-emitters (Kang et al. 2016). Overall, the lack of information regarding drilling and completion methods, surface casing depth, and abandonment methods for legacy abandoned oil and gas wells in particular prevent a thorough understanding of underlying causes of emissions from abandoned wells (Riddick et al. 2020). Measurements from a plugging program that targets high-emitting abandoned wells are needed in order to implement a broad plugging strategy that will reduce emissions (Riddick et al. 2020). Due to the relatively small emissions from abandoned wells, measuring emissions from abandoned wells generally require ground- based campaigns. An aerial survey by Pekney et al. (2018) using a cavity ring-down spectrometer was unable to identify small sources of methane associated with abandoned wells from background levels at a height of 40–50 m (131–164 ft).

The precise number of abandoned wells is not known for many states with a long history of oil and gas development, leading to increased uncertainty for national emission estimates from abandoned wells (Kang et al. 2016; Riddick et al. 2019). This issue stems from the lack of documentation for legacy wells, and inadequate well classification systems that categorize wells as "unknown" or "N/A" (Townsend-Small et al. 2016b). Kang et al. (2016) estimated the number of abandoned wells in Pennsylvania to be between 470,000 and 750,000. Riddick et al. (2019) estimated the number of abandoned wells in West Virginia to be between 63,000 and 760,000. Townsend-Small et al. (2016b) estimated that there are at least 2.3 million abandoned onshore wells in the United States, while Brandt et al. (2014) estimated up to 3 million abandoned wells nationwide. Estimates from more recent studies that were published outside the timeframe of our review put the number of abandoned wells in the United States and Canada to be at least 4 million and 370,000, respectively (Williams et al. 2021). In contrast, the most recent U.S. EPA GHGI estimates the number of abandoned wells in the United States to be 2.7 million (US EPA, 2021). Bottom-up component-level inventories like the U.S. EPA GHGI require accurate counts of abandoned wells to decrease uncertainties.

Table 3.4. Studies that measured emission rates, frequency, or enhancements from abandoned wells.

Mean Emission Rate per Well	Range	Number of Abandoned Wells Measured	Region	Study
N/A	N/A	5–15[1]	IN	Yin et al. 2020
33.4 g/hr[2]	N/A	17[3]	WV, PA	Riddick et al. 2020
0.1 g/hr	Background - 12 g/hr	112 (plugged)	WV	Riddick et al. 2019
3.2 g/hr	Background - 177 g/hr	147 (unplugged)		
18.6 t/yr	-	7[4]	British Columbia	Wisen et al. 2020
29.2 g/hr[5]	< 3.8–174 g/hr	31	Hillman State Park, PA	Pekney et al. 2018
6.4 mg/hr	1.1–16.8 mg/hr	4	Western PA	Bradshaw 2018
N/A[6]	N/A	228	Montney, British Columbia	Atherton et al. 2017
2 mg/hr	0.0006–145.7 g/hr[7]	119 (plugged)	Powder River Basin – WY; Denver-Julesburg Basin – CO; Uintah Basin – UT; Appalachian Basin – OH	Townsend-Small et al. 2016b
10 g/hr		19 (unplugged)		
15 g/hr	< 10^{-6} (Detection limit) - 350 g/hr	35 (plugged)	PA	Kang et al. 2016
22 g/hr		53 (unplugged)		
11 g/hr	6.3x10^{-4}–86 g/hr	19	PA	Kang et al. 2014

N/A – Not Available.

1. Exact number of abandoned wells is not stated.

2. 24-hour averaged emissions of selected high emission wells.

3. One abandoned well in this study was in the United Kingdom and was not included in this count.

4. Well leakage data from British Columbia Oil and Gas Commission.

5. Mean for 22 unburied abandoned wells. Nine buried abandoned wells flux rates were not significantly different from background measurements.

6. Measured emission frequency of 26%. Detection limit of 0.59 g/s was used as the emission factor for this study.

7. Range provided for nine wells with non-zero emission rates.

Detecting the locations of abandoned wells can be problematic even when modern records are available, due to the lack of above ground indications of their existence, lack of access to the location due to overgrowth, development, or property rights, and the presence of interfering metallic debris (Yin et al. 2020). Pekney et al. (2018) had success using a helicopter-based

PSE

magnetic survey followed by ground-based verification to confirm the locations of 51 abandoned wells without records, out of 88 identified sites in Pennsylvania.

Kang et al. (2019) analyzed potential mitigation strategies for abandoned wells including: plugging without venting; plugging with venting and flaring; plugging with venting and usage; flaring without plugging; and gas capture or usage without plugging. While options that do not include plugging may be more advantageous economically, they do not protect subsurface resources from potential contamination. In some locations, venting is required by law (e.g., coal areas in Pennsylvania). Using a 50-year plug lifetime, an average plugging cost of $37,000 per well, and methane emissions of 0.41 to 0.66 t/yr, all methods of emission reduction are economically beneficial if the social cost of methane—including air quality and other health or ecosystem impacts—is considered (Kang et al. 2019). If air quality and other impacts are excluded from the social cost of methane, then only gas flaring without plugging and gas usage without plugging, are economically beneficial. Of the plugging options examined, plugging without gas venting and plugging with gas venting and usage are the two most economical plugging options. The assumptions used for this analysis only represent a small fraction of abandoned wells; plugging costs in Pennsylvania range from $1,000 to $1 million per well and the methane emission rates used represent unplugged gas wells in non-coal areas and vented gas wells in coal areas, two high-emitting categories of abandoned wells.

Flaring

Only five studies focused on flaring in our review. Flaring in the oil and gas sector can be divided into three categories: natural gas production; natural gas processing; and associated gas production (gas co-produced with oil). Analysis of U.S. EPA Greenhouse Gas Reporting Program (GHGRP) data by Allen et al. (2016) found skewed emissions distributions for all flare types. Associated gas flares accounted for approximately 500 of the 21,000 reported flares but were responsible for 57% of emissions from all flaring types. The top 100 emitting associated gas flares accounted for more than half of all reported GHGRP flaring emissions. Actual flaring emissions distribution may be more skewed as reporting is not required for flares that fall below GHGRP emission thresholds (Allen et al. 2016).

A study of flares in the Bakken Shale (North Dakota) found that although median flaring destruction removal efficiency (DRE)—the percentage of fuel (i.e., methane) destroyed—was approximately 97%, close to the U.S. EPA default flaring DRE of 98% used for emissions inventories, DRE distribution and resulting methane emissions were highly skewed (Gvakharia et al. 2017). As such, using the standard flaring DRE is not representative and would underestimate methane emissions. The study found that incomplete combustion from flares were responsible for 21±4% of total methane emissions from the Bakken Shale, more than double the expected contribution than if the standard 98% DRE was applied. It should be noted that Gvakharia et al. (2017) observed lower DRE than a previous flaring study in the Bakken Shale, which found a median DRE of >99.97% (Caulton et al. 2014a), suggesting that the standard 98% DRE would instead overestimate flaring emissions (Allen et al. 2016; Caulton et

PSE

al. 2014a). Differences between measured DREs were likely due to the relatively few flares sampled in both studies and differences in measurement protocols.

In a study outside the timeframe of our review, Lyon et al. (2021) used data from the satellite-based Visible Infrared Imaging Radiometer Suite (VIIRS) to direct a helicopter-based infrared OGI survey of flares in the Permian Basin (Texas/New Mexico) and found on average 5% of flares were unlit and venting, and 6% of active flares had combustion issues. They estimated an overall DRE of 93%, with unlit flares accounting for 65% of the estimated 300 Tg annual flaring emissions in the Permian Basin. Unlit flares were also responsible for 13% of oil and gas emissions in British Columbia (Tyner and Johnson, 2021).

Willyard and Schade (2019) compared bottom-up flaring volumes from data self-reported to the Texas Railroad Commission (TxRRC) to top-down estimates from satellite-based National Oceanic and Atmospheric Administration (NOAA) data from 2012 to 2015 for the Eagle Ford Shale (Texas) and Permian Basin (Texas/New Mexico). Over this time period, venting and flaring volumes increased by a factor of 3.5 and 1.7, respectively, according to TxRRC and NOAA data. The majority of venting and flaring occurred in the Eagle Ford Shale in 2012, but shifted to the Permian Basin by 2015. On average, NOAA satellite-based estimates were twice that of TxRRC estimates. It is possible this difference is due to satellite calibration errors or the lack of daytime observations. However, the authors suggest that the TxRRC still underestimates total volumes due in part to reporting exceptions, such as not requiring reporting until after well completion.

Single-point failures

Single-point failures of upstream sources—such as accidents resulting in well blowouts—can have major impacts on regional- or state-scale reporting and inventories but are poorly understood and problematic to monitor. A TROPOMI satellite-based study of a single well blowout event in Ohio measured an hourly emission rate of 120±32 metric tons per hour (t/hr), double that of the midstream Aliso Canyon gas leak in California in 2015 (Pandey et al. 2019). Over the 20-day blowout period, this single well blowout event in Ohio may have been responsible for methane emissions equivalent to a quarter of Ohio's annual oil and gas methane emissions (Pandey et al. 2019). In a study outside the timeframe of our review, Cusworth et al. (2021b) used data from GHGSat, TROPOMI, and PRISMA at various spatial and temporal scales to estimate emissions from a flared well blowout in the Eagle Ford Shale (Texas) to be 4,830±980 metric tons over the course of the 20-day blowout period. Blowout events can be one of the largest point source emissions on a national scale, but their emissions are difficult to monitor and poorly understood. Satellite-based observations, such as those made by GHGSat, TROPOMI, and PRISMA, have the potential to detect, quantify, and monitor such events, closing this uncertainty gap (Cusworth et al. 2021b; Pandey et al. 2019).

3.1.2.   Upstream methane: key research gaps

Primary data collection and emission estimates

Research Gap: Emissions estimates from top-down studies do not agree with component-level bottom-up inventories. Studies suggest component-level bottom-up inventories routinely underestimate emissions.

Major studies that have reconciled top-down and bottom-up measurements suggest that the disagreement between bottom-up and top-down studies is a result of both modeling methods and uncertainties in the emission factors used to calculate bottom-up inventories. Bottom-up inventories likely miss a significant portion of emissions from abnormal operating conditions and other major sources that lead to heavy-tailed distributions (Alvarez et al. 2018; Zavala-Araiza et al. 2015a). A new component-level bottom-up inventory estimate suggests that emissions from storage tanks and other equipment leaks are a key contributor to the divergence seen between top-down studies and the U.S. EPA GHGI (Rutherford et al. 2021).

Research Gap: Accurately measuring emissions from upstream sources can be problematic due to the intermittent nature and spatial distribution of some sources.

The temporal variability of upstream methane emissions is poorly characterized due to the short duration of many upstream studies. With up to 90% of super-emitters displaying intermittent emission activity in some regions (Cusworth et al. 2021a), short duration studies may fail to adequately capture upstream emissions and heavy-tailed distributions. As such, emission factors derived from these studies may underestimate methane emissions.

Ground- and vehicle-based studies are further limited by site access and are subject to volunteer bias. Without cooperation from oil and gas operators, vehicle-based studies are often limited to measuring facilities in close proximity to public roads. In rural areas with limited public infrastructure, measuring emissions from a random and representative set of facilities is difficult without operator cooperation. Additionally, the spatial distribution of facilities may limit the number of sites that can be visited throughout the duration of a study, resulting in small sample sizes.

Access to high-resolution emissions activity data can assist in closing the gap between top-down and bottom-up estimates. However, when available, hourly resolution data was insufficient to resolve all emissions detected using top-down methods (Vaughn et al. 2018). Sub-hourly activity data would assist in accounting for large intermittent sources that occur as part of normal operations but may not be available or accessible for major oil and gas producing regions.

Research Gap: Methane emissions from Mexico's oil and gas sector are understudied compared to the rest of North America. In the United States, uncertainties remain for emissions between production basins as some regions have been more heavily studied compared to others.

Our review did not identify any major studies that collected primary data for upstream methane emissions in Mexico from 2015–2020. More recent studies outside the timeframe of our review have utilized aerial and satellite data to estimate methane emissions from Mexico's oil and gas sector. However, compared to the United States and Canada, the current understanding of emissions from Mexico remains limited. In the United States, upstream studies center around major basins and shale plays (e.g., Permian Basin, Bakken Shale, Barnett Shale, Fayetteville Shale, Eagle Ford Shale, Marcellus Shale, etc.). Few studies have measured emissions from less prominent oil and gas regions (e.g., Antrim Shale, Niobrara Shale, etc.).

Research Gap: Known emissions sources from the oil and gas sector, such as well blowout events and mud degassing, are not included in the current U.S. EPA GHGI (US EPA, 2021)**, resulting in underestimates from bottom-up component-level inventories.**

In recent years, the U.S. EPA has included new emission sources (e.g., abandoned wells) in the GHGI and updated emissions estimates as new emissions data has become available. However, some known emission sources have yet to be included in recent inventories due to the lack of emissions data.

Research Gap: Few upstream studies have monitored changes in methane emissions throughout the well drilling, completion, and production process. In some cases, regulatory agencies do not require data collection of emissions during development and construction activities.

A vehicle-based study found that methane emissions from two hydraulic fracturing wells primarily occurred during the flowback phase—the initial stage of production from a well—with minimal emissions during drilling and completion (Williams et al. 2018). However, a tower-based monitoring study found methane spikes during vertical drilling and hydraulic fracturing phases, but not during horizontal drilling (Russell et al. 2020). Both studies were limited to a single well pad and it is unclear how representative these results are of well drilling and completion activities on a larger scale. Regional data collected by states may also provide an incomplete picture of emissions due to exceptions in reporting requirements. For example, the Texas Railroad Commission (TxRRC) does not require venting and flaring data collection during equipment startup, well drilling, completion, or mud-gas separating (Willyard and Schade, 2019).

Super-emitters

Research Gap: The term super-emitter is not standardized and is employed differently depending on the context.

Zavala-Araiza et al. (2015b) suggested a definition of functional super-emitters as components that are in the 85th percentile of proportional loss rates for wells, however, this definition has not been widely adopted in literature since publication as most studies use absolute emission rates. Caulton et al. (2019) defined super-emitters as the top 10% of emitting sites by either proportional loss rates or emissions rates and identified different sites depending on the definition used. In other studies, these sources are simply referred to as high-emitters. Balcombe et al. (2018) suggested that because super-emitters are a continuously changing set, as opposed to a discrete set of equipment or facilities, there should not be a quantitative definition of super-emitters.

Research Gap: Emission factors and component-based bottom-up inventory methods do not adequately capture skewed emission distributions caused by super-emitters. The root causes of some super-emitters remain poorly understood.

Development of more accurate emission factors requires understanding the entire distribution of sources, including low-probability, high-emitting sources known as super-emitters (Zavala-Araiza et al. 2015a). Studies with small sample sizes may not capture highly skewed emission distributions (Omara et al. 2016). Furthermore, the reasons why sources become super-emitters is not clear, though operational practices or equipment failure due to age or improper maintenance is suggested in some cases (Allen, 2016a; Balcombe et al. 2018).

Research Gap: Mitigation of super-emitters is not incentivized within the context of the current U.S. EPA Greenhouse Gas Inventory.

This is largely due to how emissions factors are used to calculate methane emissions and the use of "regulatory reductions" instead of actual measurements to demonstrate reductions. Kemp and Ravikumar (2021) suggest that non-performance-based (i.e., mass- or rate-based) regulations risk not meeting emissions goals if operators aim to only meet minimum requirements at the lowest costs. While other countries and U.S. industrial sectors have benefitted from measurement-based regulation frameworks, it remains unclear how a similar system could be managed given the vast, disparate, and highly variable oil and gas supply chain.

Source apportioning

Research Gap: Uncertainties remain regarding intra- and inter-basin variability of ethane:methane, propane:methane, and isotopic signatures used for source apportioning. Source identification is hindered in regions where multiple types of methane source are co-located.

The rate of change of ethane:methane (C_2H_6:CH_4) and propane:methane (C_3H_8:CH_4) ratios over extended periods of time have not been adequately characterized in major oil and gas producing basins, possibly resulting in overestimates in top-down studies that use them for source apportioning. Isotopic source signatures for oil and gas methane vary, with some

PSE

isotopic signatures being more consistent than others in specific regions. Regions with large coalbed methane emissions or significant sources of biogenic natural gas need to be adequately characterized to reduce uncertainties. Regions where multiple source types of methane are co-located require additional characterization of individual sources to reduce uncertainties with source apportioning.

Abandoned wells

Research Gap: The available emissions data for abandoned wells is limited and may not represent national trends, resulting in high uncertainties in emissions for both the United States and Canada.

Representative surveys of emissions from abandoned wells are difficult due to their relatively small emissions, accessibility of abandoned wells on private property, lack of records regarding well locations, and difficulties finding wells in the field (Kang et al. 2014, 2016; Pekney et al. 2018; Riddick et al. 2019, 2020; Townsend-Small et al. 2016b; Yin et al. 2020). There are currently no long-term requirements to monitor emissions from leaking abandoned wells in both Canada and the United States (Wisen et al. 2020).

Research Gap: The number of abandoned wells in the United States and Canada is currently unknown and remains a large uncertainty in current emissions inventories.

The lack of records for historical and legacy abandoned wells results in a high degree of uncertainty in the number of abandoned wells. Estimates for the number of abandoned wells in some states with a long history of oil and gas development range by up to an order of magnitude.

Research Gap: The underlying causes of emissions from plugged and unplugged abandoned wells are not well characterized.

Many factors are likely to influence emissions from abandoned wells, including age, plugging status, plugging procedures, depth and production, proximity to horizontal drilling or hydraulic fracturing, and regulations regarding venting (Kang et al. 2016; Pekney et al. 2018). Records of well drilling, construction, completion, and abandonment are often unavailable for historic wells, limiting potential analysis of the root causes of emissions.

Flaring

Research Gap: Standard flaring destruction removal efficiencies used in bottom-up inventories are not representative of flaring activities and may underestimate methane emissions in some regions.

Emission inventories typically use a U.S. EPA default flaring destruction removal efficiency (DRE)—the percentage of fuel (i.e., methane) destroyed—of 98%, which may not accurately

account for the skewed distributions seen in both flaring DRE and emissions (Allen et al. 2016; Gvakharia et al. 2017; Lyon et al. 2021). As such, emissions from flaring could be more than double current estimates in some regions. Regional data collected by states used for bottom-up estimates may also provide an incomplete picture of emissions from flaring due to exceptions in reporting requirements.

3.1.3. Upstream methane: recommendations

Primary data collection and emission estimates

Recommendation: Additional national, regional, sector, and source-specific emission studies and monitoring are necessary to better quantify bottom-up inventories and close the gap with top-down estimates.

A multi-tiered approach to characterizing a single basin or producing field over the same timescale, such as the Barnett Coordinated Campaign, would provide the most robust emission estimates but may not be feasible for all areas. At a minimum, higher-frequency measurements are needed to understand shifts in emissions or repeated emission states (Englander et al. 2018). Continuing satellite, airborne, vehicle, and ground-based measurements over longer timescales, and during the same time frames, will complement short-term studies to reduced uncertainties, provide a more robust dataset to estimate methane emissions at varying spatial and temporal scales for upstream sources, and provide regional scale decision support for methane mitigation policies (Barkley et al. 2017; Cui et al. 2019b; Cusworth et al. 2021a; Cusworth et al. 2020). Measurements over weeks and months, rather than days, will reduce the effects of temporal variability on emission estimates and better capture high-emitting events and skewed emissions distributions.

Recommendation: Future studies should work with oil and gas operators to improve the representativeness of collected data and reduce uncertainties.

Many ground- and vehicle-based studies are limited by a lack of site access and suffer from volunteer bias. Working with oil and gas operators to gain site access will improve the overall representativeness of facility and component measurements. Access to operator activity data is also important for evaluating emissions from high-emitting events that occur during normal operations. High-resolution operator data on hourly or sub-hourly timescales would assist in accounting for temporal variations in emissions and closing the gap between top-down and bottom-up comparisons (Vaughn et al. 2018). Data provided by operators and regulatory agencies should also be checked for accuracy and consistency. Independent monitoring of emissions from the upstream sector is necessary for capturing single-point failure sources of methane that may otherwise be overlooked by methane inventories (Pandey et al. 2019).

PSE

Recommendation: Efforts to characterize methane emissions from understudied regions and processes should continue.

Methane emissions in Mexico remain understudied compared to the United States and Canada. Additional data collection is needed to develop Mexico-specific emission factors, update Mexico's national emissions inventory, and reduce uncertainties (Zavala-Araiza et al. 2021). Within the United States, the majority of upstream studies have focused on a few major oil and gas producing basins, with few studies in lower producing regions (e.g., Antrim Shale). Regions with lower production are likely to contain super-emitters which could contribute significant methane emissions to regional budgets, warranting additional investigation. Methane emission rates in these areas should be evaluated to guide regional mitigation strategies (Yin et al. 2020).

Region specific factors such as oil or gas development, the age of infrastructure, operational practices, geological characteristics, and implementation of best practices can influence methane emissions from the upstream oil and gas supply chain (MacKay et al. 2019; Zavala-Araiza et al. 2018). Additional studies would provide insight into the relationship between methane mass emissions and throughput normalized methane emissions (Robertson et al. 2017). Understanding emissions during upstream well drilling, completion, and other development activities is important to identify high emitting processes. Future studies are needed to determine emissions throughout the lifespan of upstream infrastructure in order to identify and mitigate phases of upstream development that have high methane emissions (Baillie et al. 2019; Russell et al. 2020; Williams et al. 2018).

Recommendation: Bottom-up component-based emissions inventories should be updated to include observed emissions distributions, missing emission sources, and updated emission factors in order to more accurately estimate methane emissions and inform mitigation policies.

Current component-level bottom-up inventory methods would be improved by incorporating new emissions data from recent studies and revising bottom-up emission modelling approaches (Alvarez et al. 2018; Rutherford et al. 2021). Emissions inventory modeling approaches that incorporate emission factors based on observed emissions distributions may better capture the effects of intermittent high-emitting sources and day to day variability compared to previous inventories (Allen et al. 2017; Alvarez et al. 2018). Component- and equipment-level emission factors should be regularly validated with measurements from randomly sampled sources at various spatial scales (Rutherford et al. 2021).

Super-emitters

Recommendation: Super-emitters are responsible for a disproportionate amount of methane emissions. Efforts to prevent, identify, and mitigate super-emitting sites and equipment should continue to be a top priority.

Although super-emitters cannot be eliminated entirely, identifying and reducing emissions from super-emitters in a timely manner is one of the largest opportunities for methane emission mitigation and should be prioritized (Atherton et al. 2017; Balcombe et al. 2018). If super-emitters are largely random in nature, studies with large sample sizes and frequent measurements may be necessary to effectively characterize super-emitter emission (Caulton et al. 2019). Balcombe et al. (2018) suggests that emissions mitigation technologies alone are inadequate to fully address the impacts of super-emitters. Mitigation must be used in conjunction with better LDAR programs, preventative maintenance, and multi-platform emissions monitoring and predictions to reduce the impacts of spatio-temporally dynamic super-emitters (Alvarez et al. 2018; Balcombe et al. 2018; Zavala-Araiza et al. 2018). High-resolution satellites, commercial airborne remote sensing systems, and continuous on-site methane and HDAP monitoring systems can all improve detection capabilities; however, prevention and mitigation steps require other types of management and operational practices to be in place in order to take advantage of these emerging technologies. Reducing emissions from super-emitters requires that specific sources be identified or accurately predicted (Cardoso-Saldaña et al. 2019). Aerial survey technologies and airborne methane remote sensing systems can augment further study of super-emitters and should continue research and development in detecting, quantifying, and apportioning emissions throughout the supply chain. Increased understanding of super-emitting sources will allow for more economic and efficient mitigation programs (Brandt et al. 2016a).

Recommendation: The term super-emitter must be standardized, whether based on proportional loss rates or absolute emission rates.

If super-emitters are to be explicitly targeted for mitigation, a consensus on the definition of super-emitters needs to be established by both regulatory agencies and the scientific community.

Source apportioning

Recommendation: Additional characterization of alkane and isotopic source signatures are needed to help constrain emissions from regions with multiple co-located sources of methane.

Alkane and isotopic signatures should be monitored for changes over time in regions that use these methods for source apportioning. Sources of methane that are of thermogenic and/or biogenic nature, such as coalbed methane, may decrease the effectiveness of alkane-based methods. Additional characterization using both alkane and isotopic methods can reduce

these uncertainties; however, region-specific source signatures are necessary for isotope- and alkane-based apportioning approaches to provide maximum value (Turner et al. 2016). Methods utilizing sensor networks have shown promise in reducing uncertainties in quantifying and attributing methane emissions on regional scales (Kille et al. 2019).

Abandoned wells

Recommendation: Additional studies are needed to adequately characterize the extent of leakage from abandoned wells throughout North America in order to update national methane emission budgets and inform monitoring programs and mitigation solutions.

Current studies of abandoned wells are limited in geographic coverage or scope but provide evidence that emissions from abandoned wells are a widespread problem and likely underreported (Townsend-Small et al. 2016b; Wisen et al. 2020). Agencies with jurisdiction should implement long-term monitoring requirements for abandoned wells in both Canada and the United States to determine emission rates and identify high-emitting abandoned wells for remediation.

Plugging of wells does not guarantee emissions reductions. In coal areas where regulations require the venting of abandoned wells, alternatives that reduce methane emissions while maintaining safety should be considered (Kang et al. 2016). Potential alternatives include a combination of plugging, venting with flaring, and venting with usage; however, other alternatives should be explored to reduce costs (Kang et al. 2019). Plugging and abandoning wells can be cost effective for high-emitting, average cost wells when the full social cost of methane emissions is taken into account (Kang et al. 2019).

Flaring

Recommendation: Additional studies are needed to evaluate flaring destruction removal efficiencies, methane emissions, and emissions distributions in major oil and gas producing regions and under a variety of operating conditions.

Recent studies indicate that the standard flaring destruction removal efficiency (DRE) of 98% used in bottom-up inventories is not representative of flaring activities in some regions (Gvakharia et al. 2017; Lyon et al. 2021). Additional studies are needed to evaluate flaring DREs and methane emissions more broadly in the upstream oil and gas sector to reduce uncertainties. In addition, efforts should be made by regional and state regulators to increase reporting requirements for flaring to close key data gaps and improve regional inventories.

PSE

Regulatory recommendations

Recommendation: Based on the current scientific understanding of upstream methane emissions, emissions control technologies and approaches should be swiftly implemented. The need for additional scientific study to close existing data gaps should not delay deployment of emission controls and monitoring systems for well characterized sources.

Given the short-term global warming potential of methane, it is imperative that methane prevention, detection, and mitigation strategies be swiftly and aggressively deployed. The prospect of future research to qualify emerging monitoring technologies and refine our understanding of upstream methane emission should not preempt swift action using proven technologies to reduce emissions from known sources. Pneumatic controllers, tank flashing, liquid storage tanks, and unlit flares are some of the largest upstream sources of methane emissions and the implementation of improved controls and regulations would be an effective component of emission reduction (Cardoso-Saldaña et al. 2019; Lavoie et al. 2017a; Lyon et al. 2016; Ravikumar et al. 2020; Rutherford et al. 2021; Tyner and Johnson, 2021). Leaks from sources of unreported emissions, such as fugitive emissions, should be addressed via regulations that require replacement of offending components and additional LDAR programs (Zavala-Araiza et al. 2018).

Recommendation: Regulatory agencies should implement performance targets based on either proportional loss rates or absolute emission rates.

Performance-based regulations would allow for increased operator flexibility to implement LDAR programs and mitigation technologies based on facility characteristics and regional emission-size distributions (Ravikumar and Brandt, 2017). Kemp and Ravikumar (2021) suggest that LDAR programs can balance detection thresholds and survey frequency based on regional emission-size distributions while maintaining equivalent emissions mitigations. Regions with newer upstream infrastructure that incorporate best practices exhibit low leakage rates, and may require different approaches for older fields with legacy infrastructure and active development (MacKay et al. 2019). Non-performance-based regulations risk not meeting emissions goals if operators aim to only meet minimum requirements at the lowest costs. Performance-based regulations also allow for increased operator incentives for exceeding performance goals and may encourage faster reductions than typical LDAR programs. Operators who fail to meet performance goals may be fined based on actual emission rates and the social cost of methane.

Recommendation: Methane reduction and LDAR regulations should be technology agnostic to improve cost effectiveness and operator flexibility.

Technology agnostic regulations would allow for the use of new and emerging technologies for monitoring and leak detection once equivalence to approved technologies is demonstrated (Kemp and Ravikumar, 2021; Ravikumar and Brandt, 2017). Optical gas imaging (OGI)

technologies—the standard for LDAR programs—are capable of detecting large leaks, such as super-emitters, but are costly, labor intensive, and susceptible to environmental conditions and operator judgement (Ravikumar and Brandt, 2017). Emerging technologies can be used in combination with OGI-based LDAR programs to increase screening speeds while reducing costs, but need to demonstrate equivalent emissions mitigation to existing technologies before implementation (Fox et al. 2019; Kemp and Ravikumar, 2021; Ravikumar et al. 2019). Such technology- and methodology-agnostic policies may expedite research towards more effective reduction strategies (Barkley et al. 2019b). One example of such regulations is the Alberta Energy Regulator's Alternative Fugitive Emissions Management Program (Alt-FEMP), where pilot or full-scale programs using new and emerging detection technologies can be proposed for evaluation (Alberta Energy Regulator, 2021). Additional studies should include targeted campaigns that test the effectiveness of control technologies and validate emerging sensing technologies. These data are critical for supporting policies around continuous monitoring that can underpin a move towards performance-based emissions targets

Recommendation: Stricter regulations are needed for methane monitoring and reporting from oil and gas development activities.

Pennsylvania currently requires operators to report fugitive and process gas losses from active and unplugged abandoned oil and gas wells (Ingraffea et al. 2020). All active and unplugged abandoned wells must also be tested for mechanical integrity on a quarterly basis (Ingraffea et al. 2020). Other states should consider adopting similar abandoned well policies. Other regulatory agencies, such as the Texas Railroad Commission (TxRRC), allow exceptions for emissions reporting during well construction and completion, which may result in underestimates by regional bottom-up inventories. Agencies and regulators should consider requiring reporting during all phases of upstream development where there is evidence of emissions.

3.2.   Review of midstream methane studies

The midstream sector primarily consists of facilities that support transmission and processing of hydrocarbons through pipeline systems from production regions into consumption networks. In comparison to the U.S. EPA Greenhouse Gas Inventory (GHGI), the midstream sector contains the gathering and boosting segment of the production sector, and the entirety of the processing and transmissions and storage sectors. We included gathering and boosting in the midstream to address all compressor station studies in one place, even though gathering and boosting sources are technically considered production as per the U.S. EPA GHGI. Related to this categorization decision, Marchese et al. (2015) recommended that gathering and boosting systems be a separate sector within the U.S. EPA GHGI, noting that gathering-related emissions represent a sizable portion of the total supply chain (1,631 Kt methane as per most recent U.S. EPA GHGI).

PSE

3.2.1. Midstream methane: detailed findings

A total of 71 methane studies spanning 2015 to August 2020 addressed some portion of the midstream sector and were included in the final review. Of the 71 total studies, 45 contained primary data collection for methane. The studies that did not contain primary data either relied on previously collected primary data or presented a systematic or critical review. Even when considering our broad inclusion criteria for the midstream sector, only 13 studies focused on the midstream sector exclusively, with the remaining 58 studies focused on other portions of the supply chain.

Prior to 2015, very little direct measurement or atmospheric sampling of midstream facilities had been performed outside of direct measurements used to compile initial emission factors (EFs) (e.g., 1996 U.S. EPA/GRI study). For the midstream sector, 2015 marked a critical data discovery period. Five major national-level studies were published in 2015 alone: three focusing on methane emissions from gathering and processing (i.e., compressors, dehydrators, metering) (Marchese et al. 2015; Mitchell et al. 2015; Roscioli et al. 2015) and two for transmission and storage (Subramanian et al. 2015; Zimmerle et al. 2015). In 2020, Zimmerle et al. (2020) provided an updated methane estimate for all gathering compressor stations in the United States, derived from a national sampling campaign that used measures from 180 stations across 28 production basins and nine industry operators, as well as newly available activity information for gathering stations from the EPA Greenhouse Gas Reporting Program (GHGRP)[8]. The review timeframe (2015–2020) also witnessed a host of methane studies related to underground gas storage (UGS) emissions following the wake of the Aliso Canyon, California, well blowout incident.

Due to the co-located nature of many gathering and processing facilities, multiple studies collected methane emissions data for all systems within these segments (Marchese et al. 2015; Mitchell et al. 2015; Roscioli et al. 2015). Our review also groups gathering and boosting with processing. Additionally, in this section we report on studies related to the transmission and storage sector and conclude with key research gaps and recommendations. Notably, compressor stations are heavily featured throughout these segments. Compressor stations emissions made up approximately 2,370 Kt or ~38% of all emissions according to the most

---

[8] The GHGRP (codified at 40 CFR Part 98) requires certain industrial operators to report greenhouse gases and other activity information if certain emissions thresholds are met (e.g., 25,000 metric tons CO_2e). Approximately 7,600 facilities report emissions annually encompassing about 50% of total emissions.

PSE

recent U.S. EPA GHGI[9]. This represents a slight decrease in the relative contribution compared to the ~43% compressor station share noted by Heath et al. (2015), referring to the 2014 U.S. EPA GHGI. Nonetheless, when including the gathering and boosting segment, the U.S. midstream sector makes up approximately 57% of the total supply chain emissions[10]. Overall, we identified 17 studies of compressor stations and 13 studies of underground gas storage that reported primary emissions estimates during 2015–2020. These are the two largest focus areas within the midstream.

Gathering and boosting systems and processing

Collectively, gathering and boosting systems and processing include all infrastructure that supports the movement and processing of hydrocarbons to prepare them for delivery to gas-transmission or distribution systems. Methane emissions from gathering operations were largely unknown and therefore unaccounted for in the U.S. EPA GHGI prior to the direct measurement studies published in 2015 (Marchese et al. 2015; Mitchell et al. 2015; Roscioli et al. 2015) and in 2021 (Vaughn et al. 2021). Because many of the gathering operations involve compressors, early estimates for these unaccounted sources were upwards of ~20% of the total oil and natural gas (ONG) supply chain (Alvarez et al. 2018).

The 2015 studies address one of the key research gaps identified preceding this literature review—uncertainty related to facility and component counts in operation. This research gap was the main driving force for the host of 2015 studies collecting primary methane data in the midstream. Subsequent results of these efforts directly informed future versions of the U.S. EPA GHGI. Overall, the gathering and boosting segment is estimated to contribute a third of total emissions according to the latest U.S. EPA GHGI—though much fewer emissions data points have been collected compared to upstream production.

From the initial work by Marchese et al. (2015), the U.S. EPA for the first time adopted a per-gathering station facility-level emissions factor, which was derived from measurements from 114 of the estimated 738 such facilities nationwide and 16 processing plants. This approach was somewhat unique at the time, because the U.S. EPA GHGI typically estimated emissions by multiplying counts of source-types (i.e., activity data) by an established emissions factor—not facility-level emissions factors (Zimmerle et al. 2020). Overall Marchese et al. (2015) estimated that methane emissions from the U.S. gathering and processing segment was 2,421 (+245/-237) gigagrams (Gg) of methane in comparison to 1,296 Gg derived from the U.S. EPA GHGI—a nearly

---

[9] Compressor emissions total from U.S. EPA GHGI categories: production well pad compressors, production compressor exhaust vented, gathering and boosting compressors, processing compressors (recip. and centrifugal, processing compressor exhaust, transmissions and storage compressors (transmission and storage locations), and transmissions and storage compressor exhaust.

[10] Midstream U.S. categories include gathering and boosting.

100% underestimate. The 2013 EPA Greenhouse Gas Reporting Program (GHGRP)[11] methane emissions estimate for gathering stations showed an even larger discrepancy—underestimating the Marchese et al. (2015) estimate by over 1,500% (Figure 3.6). However, this comparison predated the 2016 updates to the U.S. EPA GHGRP, which required gathering facility operators to begin reporting emissions and activities at the facility level, and, if they met certain emissions thresholds, data that became the basis of new gathering compressor station estimates by Zimmerle et al. (2020).
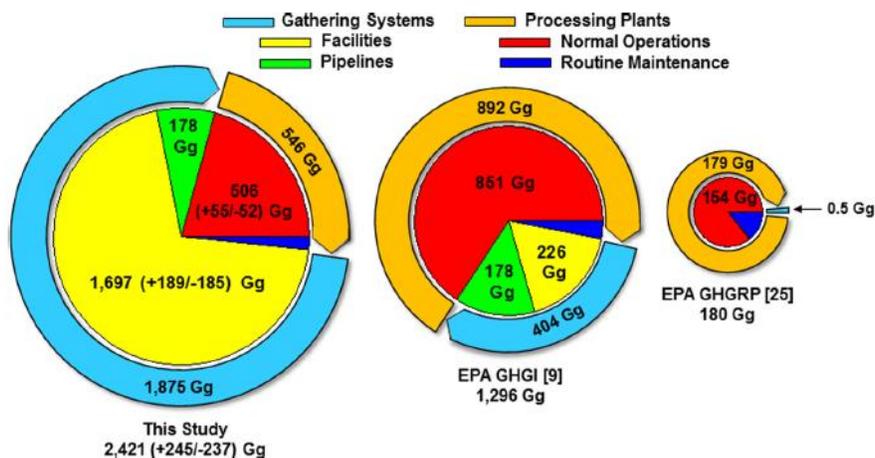
Figure 3.6. Gathering system and processing plant methane emissions derived from Marchese et al. (2015) with comparisons to the U.S. EPA GHGI and GHGRP. Source: Reproduced with permission[12] from Marchese et al. (2015).

In comparing emissions estimates from Marchese et al. (2015) to the U.S. EPA GHGI, large disagreements were observed related to the relative contributions from gathering facilities vs. processing plants. The U.S. EPA GHGI overpredicted contributions from processing plants and substantially underestimated contributions from gathering facilities. Marchese et al. (2015) noted that these disagreements could be largely attributed to missing counts of facilities and misclassified types of facilities (and associated components in operation). Additionally, the

---

[11] The GHGRP (codified at 40 CFR Part 98) requires certain industrial operators to report GHGs and other activity information if certain emissions thresholds are met (e.g., 25,000 metric tons CO_2e). Approximately 7,600 facilities report emissions annually, encompassing about 50% of total emissions. CO_2e is a standardized metric that reflects a pollutants contribution to climate change by adjusting for its global warming potential, or its ability to trap extra heat in the atmosphere over time relative to carbon dioxide.

[12] Reproduced with permission from ACS Publications. Anthony J. Marchese et al. 2015, "Methane Emissions from United States Natural Gas Gathering and Processing". *Environmental Science & Technology 2015*. https://pubs.acs.org/doi/10.1021/acs.est.5b02275. Direct any further request for this material to ACS Publications.

PSE

significant disagreements with the U.S. EPA GHGRP estimate were driven by the fact that the majority of processing plants and gathering stations were not required to report to the U.S. EPA GHGRP.

For processing plants, the authors noted that more centrifugal compressors and fewer reciprocating compressor stations were observed in the field relative to what was predicted for processing plants by the U.S. EPA GHGI. Because reciprocating compressors emit on average 75% more methane than centrifugal compressors—as confirmed by other direct measurement studies (Mitchell et al. 2015; Subramanian et al. 2015; Zimmerle et al. 2015)—the overestimation of reciprocating compressors by the U.S. EPA GHGI likely led to an overestimation of processing plant emissions.

A similar disagreement—but in the opposite direction—was observed for gathering systems as shown in Figure 3.6 from Marchese et al. (2015). However, direct comparisons across the gathering sector are confounded by the fact that gathering systems are technically nested within the production sector (Marchese et al. 2015). In addition, some components—gathering compressors, engine exhaust, pneumatic devices, etc.—are common to both gathering sites and production sites (Allen et al. 2013; Allen et al. 2015b). Therefore, assumptions must be made to determine how to apportion overlapping activity data, inviting uncertainty and a clear research limitation regarding direct measurement studies for production- and gathering-related source categories. Nonetheless, Marchese et al. (2015) apportioned activity data by using equipment data provided by the 738 partner facilities, and detailed equipment surveys from the 114 sampled gathering facilities. From these assumptions (and as shown in Figure 3.6), a total of 404 Gg of methane emissions were assigned to the 2014 U.S. EPA GHGI. Thus, in comparison to the 1,697 Gg estimated from Marchese et al. (2015), the U.S. EPA GHGI substantially underestimated emissions from gathering stations.

Three recommendations were put forth by Marchese et al. (2015). The first included the recommendation to separate out gathering-related emissions in the U.S. EPA's GHGI (which has not been adopted to date). The second was a proposed rule change to require activity and emissions for gathering and boosting stations to U.S. EPA's GHGRP. This recommendation was adopted for the 2016 U.S. EPA GHGRP, and data therein has been used to provide new estimates for gathering compressors (Zimmerle et al. 2020). The final recommendation noted a key research gap in their gathering sector emissions estimates—that methane emissions were not measured for the estimated 445,135 miles (mi) of gathering pipelines as per the U.S. EPA's 2014 activity data. Notably, the data used to develop the national emissions estimates, which were originally presented in Mitchell et al. (2015), used only one sampling method (tracer-tracer). Other studies, such as Subramanian et al. (2015) and Vaughan et al. (2017), deployed multiple concurrent measurement methods—such as on-site direct measurement with tracer-tracer measurements—which were crucial in better understanding observed source regimes and overall certainty in emissions estimates.

After the U.S EPA's GHGRP altered its reporting requirements as per recommendations by others (Subramanian et al. 2015; Zimmerle et al. 2015), in combination with new on-site measurements, Zimmerle et al. (2020) used U.S. EPA GHGRP activity data to produce new national emissions estimates for gathering compressor stations. These new estimates were later used as the basis for the 2020 U.S. EPA GHGI. Zimmerle et al. (2020) was the first study to have access to U.S. EPA GHGRP reports for gathering stations, which included information on 15,895 compressors. While the number of stations are not reported, the study estimated the number of stations based on a dataset from 1,687 partner stations, resulting in a nationally-weighted ratio of 2.8 compressors per station. This resulted in an estimate of about 500 more stations than currently estimated by the U.S. EPA GHGI. Despite this increased station count, national emissions estimates were 66% lower than the U.S. EPA GHGI, and nearly half (57%) of the emissions estimated by Marchese et al. (2015) as shown in Table 3.5, below. The largest contributor to total emissions came from compressor exhaust, also called combustion slip—making up nearly 38% (30–43%) of total emissions across all stations measured. Although not shown in Table 3.5, large variability was also observed across sites measured—with both station throughput and emissions spanning five orders of magnitude.

Table 3.5. National methane emissions estimates from gathering compressor stations. Source: Reproduced with permission[13] from Zimmerle et al. (2020).

emission category	activity basis	activity data source	emissions estimate source	methane emissions Gg/year	activity estimate	emission factor kg/h/unit
current GHGI				1955.1	5241	42.6
study total	stations			1290 [+4/−3.4%]	6110 [+4.5/−4.3%]	24.1 [+5.8/−5.2%]
			Fugitive[a]			
station equipment	stations	study[b]	study[d]	72.3 [+41/−28%]	6110 [+4.5/−4.3%]	1.35 [+41/−28%]
AGRUs	units	EPA[c]	study[d]	0.0895 [+46/−34%]	149 [+1.1/−0.86%]	0.0683 [+46/−34%]
compressors	units	EPA[c]	study[d]	275 [+14/−10%]	17,092 [+0.97/−0.81%]	1.84 [+14/−11%]
dehydrators	units	EPA[c]	study[d]	1.84 [+13/−13%]	3686 [+0.9/−0.78%]	0.0569 [+13/−13%]
separators	units	study[b]	study[d]	1.14 [+35/−25%]	12,397 [+34/−22%]	0.0105 [+12/−11%]
tanks	units	EPA[c]	study[d]	201 [+9.8/−8.6%]	35,859 [+1.6/−1.2%]	0.64 [+9.9/−8.7%]
			Vented and Combusted			
			Normal Operation			
AGRU vents	not estimated					
dehydrator vents	units	EPA[c]	calc[e]	52.1 [+1.2/−0.97%]	3686 [+0.87/−0.77%]	1.61 [+1.4/−1.3%][g]
flaring	units	EPA[c]	calc[e]	9.32 [+1.4/−1.1%]	4643 [+1.3/−1%]	0.229 [+1.8/−1.6%][g]
compressor exhaust	units	EPA[c]	study[d]	366 [+2.6/−2.5%]	17,971 [+3/−2.9%]	2.32 [+4/−3.9%]
			Pneumatic Devices			
(high bleed)	units	EPA[c]	EF[e]	34.2 [+8.5/−8.4%]	6692 [+1.3/−1.1%]	0.583 [+8.6/−8.4%]
(intermittent bleed)	units	EPA[c]	EF[e]	201 [+7.1/−6.9%]	108,684 [+1/−0.87%]	0.211 [+7.1/−7.1%]
(low bleed)	units	EPA[c]	EF[e]	7.07 [+7.2/−7.3%]	37,101 [+1.1/−0.89%]	0.0218 [+7.4/−7.3%]
			Routine Maintenance/Upsets			
blowdowns[f]	events	EPA[c]	calc[e]	69.4 [+1.2/−0.98%]	492,461 [+1.1/−0.91%]	0.0161 [+1.5/−1.5%][g]

[a]As per GHGI definition, category includes both fugitive emissions and some vented sources, such as rod packing vents. [b]Station count estimated from the GHGRP reported compressor count and study factor for compressors per station. The separator count estimated from a similar ratio of separators per compressor. [c]GHGRP reported units or events, scaled for operations that do not report to GHGRP [V3-S3-6.1].[20] [d]Major equipment emission factor developed from study. [e]Calculated emissions, "calc", or emission factor, "EF", taken directly from GHGRP. [f]Includes compressor unit blowdowns, emergency shutdowns, and blowdown of pig launchers and receivers, facility piping, and other equipment meeting the GHGRP reporting threshold. [g]Emissions estimate taken directly from GHGRP without a confidence interval. The confidence interval here reflects only scaling for operations that do not report to GHGRP.

PSE

Multiple factors were associated with the large underestimates predicted by Zimmerle et al. (2020) compared to the U.S. EPA GHGI and Marchese et al. (2015). First, the effect of super-emitters differed between the two studies; however, even after artificially increasing the effect from large emitters by 50%, the total estimated emissions did not make up the gap between the two. Next, the population of partner stations to sample from was much larger in Zimmerle et al. (2020) than in previous studies (1,705 vs. ~700). This suggests gathering stations were much smaller, simpler, and lower through-put than indicated in Mitchell et al. (2015). Relatedly, many more low-emitting station types were in operation than predicted by Mitchell et al. (2015). One example is all-electric stations, which were some of the lowest emitting stations measured, with all stations in the lowest 11th percentile. Finally, Zimmerle et al. (2020) noted that availability of U.S. EPA GHGRP reports provided a level of internal consistency and required minimal assumptions and associated estimations of station size and equipment that were present in many previous studies.

Zimmerle et al. (2020) provided three recommendations to continue to improve and update gathering sector emissions. These included: (1) require all operators to report counts of stations and separators to the U.S. EPA GHGRP; (2) clarify the definition of yard piping to include all ancillary equipment not included in other major categories, and; (3) report the driver type and loading for all compressors (e.g., electric, rich- and lean-burn engines, turbines, etc.). The authors also noted that additional on-site measurements are still lacking for numerous components at gathering facilities. These include flares, dehydrators, still/reboil vents, acid gas removal units, and pneumatic controllers—all of which are likely to have long-tail emissions distributions (which remain a key research gap).

Three other studies contributed directly-measured data assigned to gathering and boosting activities (Luck et al. 2019; Vaughn et al. 2017; Zimmerle et al. 2017). Vaughn et al. (2017) measured gathering stations, Zimmerle et al. (2017) measured gathering pipeline emissions, and Luck et al. (2019) measured pneumatic controllers. These three studies uniquely deployed three independent teams and methods: measuring methane from 17 gathering stations located in the Fayetteville Shale, Arkansas, using direct on-site measurement, aircraft-based measurement, and tracer-tracer measurements using a mobile laboratory. Thus, the study by Vaughn et al. (2017) could be considered one of the first hybrid and concurrent bottom-up and top-down measurement studies. Sampling durations were three weeks, with tracer measurements made in weeks 1-3 and on-site measurements in weeks 2-4, allowing for both contemporaneous and concurrent comparisons between methods. On-site measurements included Bacharach Hi Flow samplers and optical gas imaging for device-level measurements of flanges, unions, valve stem packing, rod packing vents, open-ended lines, pneumatic devices and controllers, and other like sources. The three studies also performed simulated direct measurements in lieu of direct measurements for compressor station combustion slip, crankcase vents, and dehydrator regenerator vents due to inaccessible sampling conditions. Simulated measurements entailed previously collected exhaust test data for 111 compressors (combustion slip), and statistical simulations (e.g., Monte Carlo) for the crankcase vents and dehydrator regenerator vents.

Results of the sampling found that simulated combustion slip contributed 78% to the total methane emissions using the on-site estimates. In comparing on-site measures to the tracer-tracer, 11 of 14 facilities had overlapping estimates with 95% confidence intervals. Notably, the three facilities that exhibited immeasurable sources were excluded from the comparison, in part due to the magnitude of on-site emissions that were not safe to sample (i.e., super-emitters). The sources in question were tank venting emissions at both gathering stations. Emissions estimates were produced via tracer and aircraft measures instead. Emissions from these two tanks alone were greater than all other emissions (measured and simulated combined) observed at their respective facilities. Outlier tank venting emissions were also observed in Mitchell et al. (2015), who noted that venting from liquids storage tanks produced emissions rates four times higher on average than similar facilities without venting. While this seemed like a large discrepancy, Vaughn et al. (2017) found that methane emissions from gathering and boosting stations were 73% greater than the estimates by Marchese et al. (2015), which were based on the measurements in Mitchell et al. (2015). However, these estimates are in direct contradiction to estimates derived in later studies such as Zimmerle et al. (2020) and Vaughn et al. (2021).

Key research gaps and recommendations from Vaughn et al. (2017) included:

● Regarding comparisons between on-site measurement emissions estimates and tracer estimates, the authors noted that tracer predictions were lower than on-site estimates. One potential explanation, first noted by Roscioli et al. (2015), is the invalid assumption that tracer gases released at a facility undergo identical plume dynamics. The methane released from the facility may behave differently, due to additional plume buoyancy that ultimately results in a portion of unmeasured methane downwind relative to the tracer gas. Subramanian et al. (2015) also suggested that unmeasured lofted exhaust emissions may have systematically biased their tracer flux estimates in measuring emissions from compressor stations. Ultimately, Vaughn et al. (2017) recommended future tracer-tracer studies should release one tracer gas directly into compressor station exhaust, and a second tracer gas on the ground nearby in a typical location.
● In comparing the top-down to bottom-up estimates using the aircraft cylinder flight path mass balance method (Conley et al. 2017), the study found an observed upward bias in aircraft estimates over on-site measurements, which was significant at the 90% confidence interval level, but not 95%. The authors noted that the differences were likely due to the inclusion of other methane sources in addition to the target gathering facilities, which could not easily be disentangled. The authors noted that aircraft facility measurements should be used with caution when emissions from nearby sources may confound results. Additional uncertainty comes from the altitude of the flight path, which must be extrapolated downward. One recommendation for future aircraft studies is to equip the aircraft with tracer gas measurements and simultaneously deploy an aloft tracer-tracer method. Notably, this was later applied to the Aliso Canyon leak in California, by treating the leak as a natural tracer experiment with aircraft sampling (Gourdji et al. 2018).

- Information related to methane emissions contributions from maintenance, episodic, and malfunction events are critical to reduce uncertainty in total oil and natural gas methane emissions. In some cases, on-site observers (or the equivalent sensing technologies) are necessary to fully characterize events that contribute disproportionately to total emissions. Nonetheless, large emissions sources present a double-edged sword. On the one hand, their identification and characterization are critical in fully understanding emissions regimes and in informing mitigation policies. On the other hand, due to the explosive nature of methane, large emissions sources on-site can present hazardous conditions, limiting on-site direct measurement activities. This axiomatic feature was identified by multiple authors during this period of time (Subramanian et al. 2015; Vaughn et al. 2017; Zimmerle et al. 2015).

Two studies that reported gathering and boosting emissions occurred outside of our initial search window, but are included here for completeness. Vaughn et al. (2021) provided a key update to compressor combustion slip emissions estimates—further verification that compressor emissions are likely overestimated in the current U.S. EPA GHGI, and that compressor emissions have likely decreased over the years due to deployment of emissions controls. In taking their own recommendations, Vaughn et al. (2021) deployed a novel in-stack tracer gas method to measure combustion slip at 67 individual gathering and boosting compressor stations. Their results showed that mean methane emissions rates for both lean-burn and rich-burn compressor engines were 12% lower than U.S. EPA GHGI underestimates, further confirming that previous U.S. EPA GHGI estimates for compressor stations were likely overestimated by a factor of two. As such, the authors concluded that the U.S. EPA GHGRP combustion slip emissions factor is inadequate and should not be used in the U.S. EPA GHGI. Driving these differences are likely to be different assumptions on compressor counts, operating loads, and utilization rates. Additionally, current emissions factors in USEPA Method AP-42 for 4-stroke rich-burn engines likely do not reflect additional emissions controls that have become standard on certain systems, such as exhaust after treatment and the use of pre-chambers for ultra-lean combustion engines. Vaughn et al. (2021) provides an updated method to estimate national gathering and boosting combustion slip emissions by basin.

Cusworth et al. (2021a) performed repeated fly-overs in the Permian Basin to evaluate emissions intermittency at the facility level and, notably, found 3,067 unique methane plumes from 1,756 distinct sources. Gathering compressor stations were found to contribute 19% of total emissions in the Permian Basin. The authors noted that in combination with gathering pipeline emissions, gathering and boosting emissions comprised 38% of total emissions observed in the Permian Basin. This represents a 20% relative shift from upstream to midstream emissions compared to other U.S. basins (Alvarez et al. 2018). There is some indication that the Permian Basin may be somewhat unique in its inability to process high levels of hydrocarbon production, leading to increased venting gathering and boosting venting emissions, particularly during the recent COVID-19 pandemic (Lyon et al. 2021).

Gathering pipelines

By 2017, there was still very little direct measurement data for gathering pipelines—a network consisting of an estimated 400,000 miles (~643,737 km) of pipelines and ancillary infrastructure connecting production to gathering and processing sectors (US EPA, 2020). In addition to pipelines, gathering pipeline systems consist of pig launchers and receivers, blocking valves, and a variety of other less common components such as knock out bottles used to remove liquids common in older pipeline systems. Zimmerle et al. (2017) performed a four-week, vehicle-based sampling campaign in the Fayetteville Shale. The authors collected direct measurements from 96 km (~60 mi) of gathering pipelines (10 years old on average), 56 pigging facilities, and 39 block valves across two partner companies. Using Monte Carlo methods, the methane emissions were scaled to an area consisting of 4,684 km (~2,911 mi) of gathering pipeline. The authors note that extrapolating emissions from 96 km (~60 mi) of pipe to an estimated 4,684 km of pipe is a major study limitation, particularly given their results and how heavy-tailed their distribution of emissions was. Nonetheless, the study detected 98 total leaks, 72% of which originated from valve packing infrastructure, with the remaining 13% from pig launchers, 12% from flanges, and 2% from gauges. In terms of methane emissions, valve packing contributed 49% of the total methane, pig launchers 47%, flanges 3%, and gauges 1%. Notably, no super-emitter events were captured for the auxiliary components—either planned episodic events or otherwise. Moreover, total methane emissions from above-ground equipment were small compared to other portions of the gathering system, with estimated emissions across the 65x80 km (40x50 mi) study area of 402 kg methane per hour.

However, Zimmerle et al. (2017) did capture one large gathering pipeline leak (measured at 4 kg methane/hr) that dominated total emissions from gathering pipelines. Scaling this leak to the total study area resulted in pipeline leaks dominating the total emissions from the gathering pipeline system, with an estimated 93% of emissions. The authors noted some caution based upon their small sample size and detection of only a single leak using their vehicle-based sampling method. However, the authors did perform sensitivity tests, noting that the sampling system was likely capable of capturing leaks one to two orders of magnitude less than the single leak detected. Nonetheless, the authors concluded that U.S. EPA's current emissions factor for gathering pipelines, which was based on distribution network measurements performed in the 1990s, may be significantly underestimating gathering pipeline emissions, particularly in areas where older pipeline systems exist. The authors suggested that significant uncertainty remains for leak frequencies per length of pipeline, and without better understanding of these leak frequencies, designing a sufficient sampling campaign to capture enough leaks to characterize the system remains a challenge. More frequent periodic screenings could help constrain *a priori* estimated leak frequencies, informing future sampling campaigns using similar vehicle-based sampling methods.

Cusworth et al. (2021a) performed repeated fly-overs in the Permian Basin to evaluate emissions intermittency at the facility level. The authors found gathering pipelines contributed 19% of total emissions, on par with compression stations (19%) and a greater contribution than

was found for well sites (17%). The total contribution of gathering and boosting emissions was estimated to be 38% in the Permian Basin, with 65% of methane sources occurring at production sites from (Cusworth et al. 2021a). This value represents a much higher estimate than the national breakdown by Alvarez et al. (2018), which estimated 24% from gathering and boosting. The authors suggested that emissions associated with gathering and boosting systems in the Permian Basin may have been unable to keep pace with production, resulting in excess venting and inefficient flaring at gathering and processing stations. There are some field-based measures supporting the hypothesis that the Permian Basin was operating at overcapacity during the time of sampling, which overlapped with the COVID-19 pandemic (Lyon et al. 2021). In addition to overcapacity issues, the authors also note that gathering pipeline corrosion, material defects, and fitting failures are also likely culprits given the magnitude and overall emissions regime observed from gathering pipelines.

Pneumatic controllers

Pneumatic controllers (PCs) and equipment leaks are the largest emission sources in the oil and natural gas production segment—not including potential missing emission sources—with malfunctioning controllers contributing that large majority of total pneumatic controller emissions (Alvarez et al. 2018). Like compressor stations, PCs are present throughout the supply chain. From the 2021 U.S. EPA GHGI, over 1.1 million PCs were estimated in operation in the United States contributing approximately 20% of total methane emissions from the entirety of natural gas operations (Alvarez et al. 2018) – an estimate that was recently found to match oil and gas emissions in British Columbia, Canada (Tyner and Johnson, 2021). It is also worth noting that PCs fall into the venting and fugitive emissions category, and some systems may emit raw unprocessed natural gas that has been shown to contain relatively higher ratios of HDAP non-methane VOCs. Similar contributions from PCs have been observed in many production regions, including in Canada, though there is some indication that overall emissions from PCs may be decreasing due to the movement to replace high-bleed PCs with zero-bleed PCs (Tyner and Johnson, 2021).

Luck et al. (2019) was one of the first to report on long-duration direct measurements of pneumatic controllers (n = 72) at 16 midstream gathering facilities, with each sampled on average for 76 hours. Studies preceding Luck et al. (2019) had generally found bottom-up emissions estimates of PCs to underestimate direct measurements, with many suspecting that some degree of abnormal operating conditions were to blame (Allen et al. 2013; Allen et al. 2015b; Thoma et al. 2017). However, Luck et al. (2019) pointed out that to better understand the presence of abnormal operating conditions of PCs, direct measurements of PCs needed to be longer than 15 minutes—the average time duration that previous PC studies had used. This is a particularly troublesome sampling design for intermittent controllers that may only actuate a few times in 15 minutes (if at all) and could potentially lead to an under sampling of both "normal" and abnormal operating conditions.

Luck et al. (2019) found that 42% of the 72 measured devices demonstrated abnormal emissions behavior, defined as emissions substantially higher than the emissions of those operating normally. For example, intermittent-vent PCs had nearly 8x the emissions than normally operating intermittent-vent PCs. Notably, 25 of 40 intermittent-vent PCs exhibited abnormal emissions behavior, compared to five of 24 low-bleed PCs and zero of the 8 high-bleed PCs. The authors found that sampling durations of at least 24 hours would be required to capture a similar distribution of mixed-type PC emissions with an error tolerance within 20%. High- and low-bleed continuous-vent PCs can tolerate shorter sampling durations to capture a representative long-term average as compared to intermittent-vent PCs. which were shown to vary widely even within the 3-day measurement duration. The authors concluded that their findings suggest a need to use longer-duration, direct measurements to develop updated emissions factors that take into account abnormal operating conditions that appear fairly normal but have not been captured previously due to short sampling durations.

Transmission and storage

The transmission and storage sector consists of a network of compressor stations, pressurized high-volume pipelines, and underground gas storage systems. Similar to the gathering and boosting sector, very little primary methane emissions data had been collected for transmissions and storage systems prior to 2015. Early estimates of the ~2,500 compressor stations within the in the transmissions and storage sector suggested the sector contributed a previously unaccounted ~20% of the total emissions from the oil and natural gas supply chain (Alvarez et al. 2018). The companion studies Zimmerle et al. (2015) and Subramanian et al. (2015) were the first to report contemporary direct methane measurements in the transmissions and storage sector, reporting data on 45 compressor stations located in 16 states.

Sampling entailed comprehensive onsite measurement methods, including a two-stage leak detection and measurement including on-site direct measurement and a downwind tracer-tracer approach deployed on a mobile sampling vehicle (Subramanian et al. 2015). Again, the dual deployment of two measurement techniques works in a complimentary way, in which shortcomings of one method are addressed by the other. For example, the downwind tracer can provide robust target facility-level emissions; however, it can not provide specific source apportionment onsite. Direct measurement data collected onsite can pinpoint sub-facility sources, but not all sources are accessible for direct sampling, especially at large compressor stations that contain hundreds of individual valves, vents, and other emissions points.

From Subramanian et al. (2015), for the 38 sites that exhibited methane emissions less than 200 standard cubic feet per minute (SCFM), the tracer flux method resulted in systematically higher measures at the lower-emitting sites, while the opposite was observed at higher-emitting sites. No single explanatory factor was identified. One potential reason for the difference between the two data sets for lower-emitting facilities is that a subset of on-site sources could not be measured due to inaccessibility and may have been underestimated

based on study activity factors used to estimate them. The authors also noted that the discrepancy for the higher emitting sources could be due to an overestimation with AP-42 emissions factors or perhaps the downwind tracer did not fully capture the lofted exhaust emissions from those sites. The authors also observed that sampling performed at one site resulted in a very large discrepancy between the onsite estimate and the tracer—a difference of 15 SCFM vs. 880 SCFM. This discrepancy was likely due to a pipeline venting event related to site maintenance. While the venting event occurred for 15–20 hours, and an onsite emissions estimate was made using the open ended line study factor, the onsite team could not safely measure the event, ultimately highlighting the challenges in accurately capturing these types of large emissions events.

In comparing the tracer results to on-site measurements, the authors noted the non-parametric nature of the observed uncertainties that exists throughout these systems (Subramanian et al. 2015). In this situation, a distribution of emissions rates exists for a component; however, the bounds or parameters of that full distribution are impossible to quantify. This observation is underscored in part by the inherent tradeoffs in sampling methodologies that capture emissions, and by the inherent long-tailed distributions of emissions that are captured by the current sampling and inference methodologies. Like many other studies to this point, Subramanian et al. (2015) noted that the skewness of emissions from transmission and storage compressor stations appears similar to other natural gas sectors, noting that the highest 10% of sites contributed over 50% of the aggregated emissions. The authors also noted the inverse, whereby the lowest 50% of sites contributed less than 10% of total emissions.

From Subramanian et al. (2015), site-level emissions estimates varied substantially across the 45 sites—varying by almost three orders of magnitude, and with much greater variations between sites vs. within sites (see Figure 3.7). Unsurprisingly, operational status is highly predictive of total emissions, whereby operating facilities emit more than facilities on standby. However, it is worth noting that both super-emitting sites identified by Subramanian et al. (2015) were at facilities that were on standby (likely due to leaky isolation valves, rod-packing vents, or other leaks from pressurized equipment). The authors noted that there was not enough data to conclude whether super-emitters are more common in standby or operating modes, but natural gas demand can play a large role in operating mode trends. For example, many northern latitude facilities may often operate in standby mode in the summer, when gas demand is low, compared to winter, when gas demand peaks. The authors additionally noted that the methodologies behind the operational mode weighting factors used in the U.S. EPA GHGI are not well documented. Nonetheless, identifying factors such as contributions of operating mode status that may predict the likelihood of super-emitters remains a key research challenge.
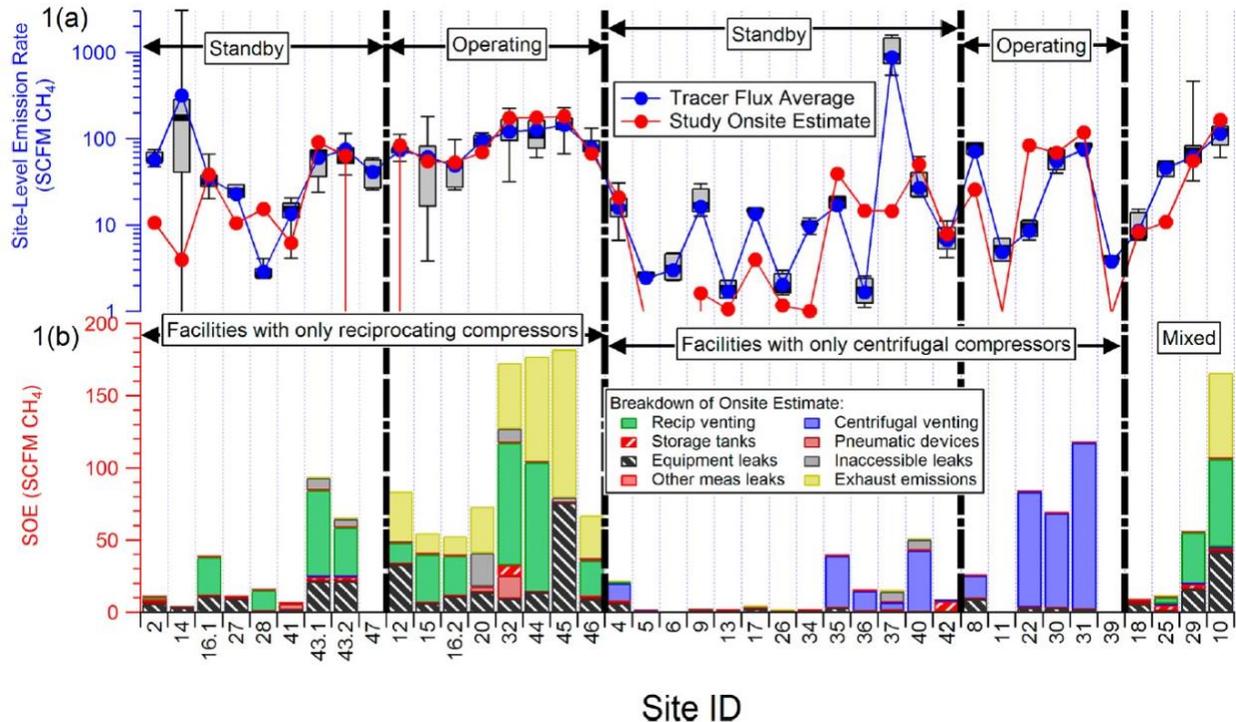
Figure 3.7. Methane emissions estimates for the 40 sites sampled by Subramanian et al. (2015) showing: (1a) tracer flux-based methane emissions; and (1b) study on site emissions. Results are broken out by reciprocating vs. centrifugal vs. mixed compressor station types on site, in addition to operational conditions—either standby or operating. In 1a, the mean flux-based estimates are depicted by the blue line with box and whisker plots showing confidence intervals. The mean SOE emissions are also shown in 1a as a red line. Direct measurements in 1b are apportioned to respective components as shown in the bar graphs with accompanying legend. Source: Reproduced with permission[14] from Subramanian et al. (2015).

The measurement data presented by Subramanian et al. (2015) was used to develop updated methane emissions estimates for the entire transmissions and storage sector that was found to underpredict U.S. EPA's GHGI estimates. Zimmerle et al. (2015) estimated total methane emissions at 1,503 [1,220–1,950] Gg/yr (95% confidence interval) for the transmissions and

---

[14] Reproduced with permission from ACS Publications. R. Subramanian et al. 2015, "Methane Emissions from Natural Gas Compressor Stations in the Transmission and Storage Sector: Measurements and Comparisons with the EPA Greenhouse Gas Reporting Program Protocol". *Environmental Science & Technology 2015*. https://pubs.acs.org/doi/10.1021/es5060258. Direct any further request for this material to ACS Publications.

PSE

storage sector compared to the 2012 U.S. EPA GHGI estimate of 2,071 [1,680–2,690] Gg/yr. While the confidence intervals overlapped, the differences between the estimates resulted from many significant but offsetting factors, which can greatly impact how emissions reductions strategies are focused. Factors that reduced the study estimate included a lower facility count, more lower-emitting compressor engines, and the reduction of natural gas-driven pneumatic devices. Factors that increased the study estimate (relative to the U.S. EPA GHGI) included updated emissions rates, and incorporation of skewed emissions distributions to account for the two super-emitters identified across the 45 sampled sites. Most notably, long-tail distributions and super-emitters account for nearly 40% of the emissions estimated by Zimmerle et al. (2015). While accounting for these non-parametric emissions distributions remains a challenge, the authors noted that more tractable issues related to the U.S. EPA GHGI activity data may provide additional uncertainty reductions. Errors in the U.S. EPA GHGI activity data—both in counts and technologies deployed—should be easy to rectify if all sites were required to report to the GHGRP, not just 28% of sites. As mentioned above, the U.S. EPA GHGRP was changed in 2016 requiring operators to report activity and emissions if they meet the threshold of 25,000 metric tons CO_2e.

Underground gas storage

Following the 2015 Aliso Canyon, California, well blowout incident, the five years between 2015–2020 witnessed a host of methane studies related to underground gas storage (UGS) emissions. Our systematic review identified 13 studies focused on UGS methane emissions, and likely many other studies were published in this time period related to the structural integrity of UGS systems (Long et al. 2018; Michanowicz et al. 2017). Using an instrumented aircraft and ground-based whole-air canister sampling, Conley et al. (2016) estimated that Aliso Canyon released 97,100 metric tons (5.0 billion SCF) of methane and a proportional amount of benzene using a benzene-to-methane enhancement ratio of $(5.2\pm0.1)\times10^{-6}$ throughout the duration of the leak (118 days). At an average leak rate of 53±3 metric tons of methane per hour for the first six weeks, the Aliso Canyon leak was the largest known anthropogenic point source of methane in the United States at that time (Conley et al. 2016). In the wake of the Aliso Canyon disaster, a renewed focus was placed on midstream natural gas infrastructure across the United States, and methane leakage across California. Outside of California, very little target sampling of UGS facilities has occurred.

Staying in California, Fischer et al. (2017) first identified discrepancies between reported emissions vs. measured via airborne surveying at nine of California's storage fields. At this time, very little direct measurement had been performed at gas storage fields in California and nationwide. Addressing this data gap, Thorpe et al. (2020) provided an in-depth sampling campaign covering all 12 California UGS facilities, providing both spatially and temporally-resolved methane emissions estimates. The study deployed two airborne remote sensing methods: remote sensing using an Airborne Visible/Infrared Imaging Spectrometer (AVIRIS-NG) and mass balance in situ sampling via a cylindrical flight pattern around each individual facility. Samples spanned January 2016 to November 2017. Overall, 229 unique facility samples

PSE

were collected. While the authors found that UGS methane emissions contributed only a small fraction of the statewide methane budget (only about 10% of the natural gas portion of the inventory), significant discrepancies were observed between reported and measured emissions. Measured emissions from the seven facilities that report to the state were about five times greater than reported, largely driven by non-routine emissions activity such as blow-downs and higher-than-expected compressor exhaust (rod pack venting was the suggested culprit). Significant temporal variability was also observed as is evident in the time series of methane estimates at the McDonald UGS facility, where a range of 84–760 kg methane per hour was observed (Figure 3.8).
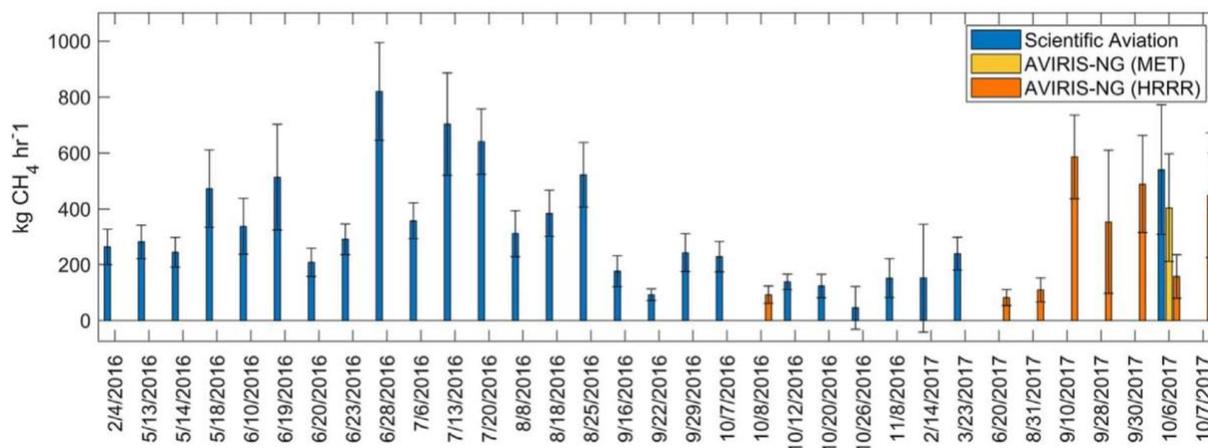
Figure 3.8. Time-series of methane emissions estimates from the McDonald Island UGS facility. Source: Reproduced[15] from Thorpe et al. (2020).

While the discrepancies between measured and reported emissions were large, in addition to the significant temporal variability, the majority of those emissions are likely due to malfunctions on readily-identifiable equipment (Thorpe et al. 2020). However, without a protracted sampling design, isolating operational conditions remains elusive in one-off sampling campaigns. The authors strongly recommended the need for both spatially- and temporally-resolved observational strategies moving forward, particularly in a concurrent fashion. The authors also noted the significant advancement in remote sensing systems—particularly with spatial resolutions now capable of 1–3 m (3.3–9.8 ft) and differentiating between facility-level source components. Recognizing there may be resource constraints in deploying similar dual systems elsewhere, the authors suggested deployment of systems with less spatial resolution, but more frequent regional monitoring frameworks designed to capture temporal trends and subsequent abnormal operating conditions.

---

[15] Reproduced as permitted by the creative commons license. No changes were made to the content of the reproduced figure.

PSE

3.2.2. Midstream methane: key research gaps

Primary data collection

Research Gap: Even when considering our broad inclusion criteria for the midstream sector (i.e., gathering and boosting and processing), only 18 studies focused on the midstream sector exclusively—with very little study pre-dating our review.

Overall, relative to upstream sectors, very little primary methane emissions data have been collected for midstream sources, but significant advances have been made in the past five years. Many of the similar research gaps identified in the upstream are in midstream sources as well, namely the contribution of—and inability to fully characterize—large emitting sources and their underlying root causes. Prior to 2015, very little direct measurement or atmospheric sampling of midstream facilities had been performed outside of direct measurements used to compile initial emission factors. A total of 85 studies spanning 2015 to August 2020 addressed some portion of the midstream sector and were included in the final review. Of the 85 total studies, 64 contained primary data collection for methane. Five of these 64 also included health-damaging air pollutant (HDAP) measures. The studies that did not contain primary data either relied on previously collected primary data or presented a systematic or critical review.

Many of the early studies from 2015–2020 relied on sampling campaigns that last only a few weeks. While even these studies identified issues and tradeoffs between sampling methodologies (e.g., direct measures vs. tracer-tracer), only later have we seen the emergence of longer-term sampling campaigns that identified significant variability of emissions over time—calling into question the robustness of previous shorter-term sampling studies (Luck et al. 2019; Thorpe et al. 2020). Sampling duration also has an impact on the likelihood of capturing large emission events, particularly associated with abnormal operating conditions, and particularly for certain components such as intermittent pneumatic controllers that were found to be particularly prone to abnormal operating behavior (Luck et al. 2019).

Research Gap: Measurement studies of specific midstream source-types are lacking relative to upstream studies and are concentrated in only two basins. Specific calls for additional measurement data have been made for gathering pipelines, facility-level yard piping, pneumatic controllers, and multiple components at gathering stations.

Marachese et al. (2015) was the first to study gathering station emissions exclusively and noted that methane emissions were not measured for the estimated 445,135 miles (mi) of gathering pipelines as per the U.S. EPA's 2014 activity data. Subsequent studies of gathering pipelines have found conflicting results with Zimmerle et al. (2017) reporting significantly lower gathering pipelines emissions in the Fayetteville Shale Basin compared to estimates by Cusworth et al. (2021a) performed in the Permian Basin. The consensus around gathering pipelines is that significant uncertainty remains for leak frequencies per length of pipeline, and without better understanding of these leak frequencies, designing a sufficient sampling campaign to capture enough leaks to characterize the system remains a challenge.

With over 1.1 million estimated pneumatic controllers (PCs) in operation throughout the U.S. supply chain, and an estimated 20% proportional methane share, PC direct measures are severely lacking. A recent study in British Columbia, Canada noted a decreasing share of emissions from PCs (Tyner and Johnson, 2021); however, no such follow-up study has been conducted since Luck et al. (2019). Zimmerle et al. (2020) also noted that additional on-site measurements are still lacking for numerous components at gathering facilities. These include flares, dehydrators, still/reboil vents, acid gas removal units, and pneumatic controllers—all of which are likely to have long-tail emissions distributions.

While significant progress has been made in estimating contributions from compressor stations (in part from primary data collection and in part due to updates to the U.S. EPA GHGRP), uncertainties remain. The overall arc in the literature seems to suggest that compressor emissions are either not as large as first predicted, or emissions controls are now in place that were not previously. In sum, both over- and underestimations of methane emissions derived from observations from compressor stations were produced in 2015–2021, with some studies suggesting the U.S. EPA GHGI was underpredicting (Marchese et al. 2015; Mitchell et al. 2015; Vaughn et al. 2017), and other studies finding the opposite (Subramanian et al. 2015; Zimmerle et al. 2015, 2020).

Outside of compressor stations, significant uncertainties in emissions from other midstream components remain in part due to lack of primary data collection. There remains very poor accounting of gathering pipelines both in terms of sheer length and location. Overall, relatively poorly constrained emissions estimates remain for gathering pipelines, compressor station yard pipelines, flares, dehydrators, acid gas removal units, underground gas storage facilities outside of California, and intermittent pneumatic controllers.

Research Gap: Temporal characterization of emissions is lacking.

Relative to spatial variability, less is known related to temporal variability of emissions. Temporal aspects of emissions, including intermittent or persistent emissions adds complexity and uncertainty in characterizing emissions and subsequently designing mitigation strategies. Most studies performed in the midstream have taken place over very short durations (e.g., 2-4 weeks)—something not uncommon elsewhere across the supply chain. Shorter duration studies have clear limitations related to generalizability of findings and representativeness of study sample measures. Given the large degree of heterogeneity across the North American oil and gas landscape, the presence of long-tailed distributions (i.e., super-emitters), and other temporal variation (e.g., persistent vs. intermittency), this limitation warrants particular attention. The few studies in the midstream that have assessed temporal aspects of emissions have been particularly informative (Cusworth et al. 2021a; Luck et al. 2019). Cusworth et al. (2021a) for example, performed multiple flyovers demonstrating the difference between persistent and intermittent emissions sources and how their relative emissions contributions can help constrain emissions discrepancies and help design more targeted mitigation approaches.

PSE

In regard to Pneumatic controllers, Luck et al. (2019) pointed out that to better understand the presence of abnormal operating conditions, direct measurements needed to be longer than 15 minutes—the average time duration that previous PC studies had used. This is a particularly troublesome sampling design for intermittent controllers that may only actuate a few times in 15 minutes (if at all) and could potentially lead to an under sampling of both "normal" and abnormal operating conditions.

Super-emitters

Research Gap: The presence of super-emitters is poorly constrained for many source-types throughout the midstream are largely drives differences between studies of similar source-types.

While significant progress has been made in identifying component-level methane emissions contributions at midstream facilities (e.g., combustion slip and tank venting), large variations in emissions profiles exist between facilities. This variability is somewhat unexpected, at least in relation to upstream facilities where operators are much more numerous (which likely explains some of that variability). Regarding gathering pipelines, Zimmerle et al. (2017) captured only a single large leak during their sampling campaign of ~96km length of pipe. Component-level studies of compressor stations and gathering and boosting stations have also identified various degrees of super-emitter behavior. Large emissions sources, or super-emitters, present a double-edged sword. On the one hand, their identification and characterization are critical in fully understanding emissions regimes and in informing mitigation policies. On the other hand, due to the explosive nature of methane, large emissions sources on-site can present hazardous conditions, limiting on-site direct measurement activities. This axiomatic feature was identified by multiple authors in this period of time (Subramanian et al. 2015; Vaughn et al. 2017; Zimmerle et al. 2015). High-resolution satellite remote sensing and continuous on-site methane and HDAP monitoring systems are already being deployed with success in quickly identifying and mitigating super-emitting events, which can occur even when emissions controls are installed and working as designed.

Research Gap: Information related to methane emissions contributions from maintenance, episodic, and malfunction events are critical to reduce uncertainty in total oil and natural gas methane emissions.

Subramanian et al. (2015) found that super-emitters can be present at facilities (compressor stations in this case) that are on standby (likely due to leaky isolation valves, rod-packing vents, or other leaks from pressurized equipment). The authors noted that there was not enough data to conclude whether super-emitters are more common in standby or operating modes, but operational status should be studied further as an explanatory variable in relation to super-emitting events. While aerial sensing platforms will continue to provide valuable detection and characterization data related to super-emitters, cooperation with operators will be necessary to further study any relationships between abnormal emissions and operational status.

3.2.3. Midstream methane: recommendations

Primary data collection

Recommendation: Existing studies provide numerous recommendations to help inform future study.

Given the co-located nature of much of the midstream infrastructure, multiple studies have demonstrated the utility of deploying concurrent emissions measurement techniques such as on-site direct measurement paired with downwind tracer-tracer measures. Both Subramanian et al. (2015) and Vaughan et al. (2017), deployed on-site direct measurement with tracer-tracer measurements—which were crucial in better understanding observed source regimes and overall certainty in emissions estimates. Vaughn et al. (2017) recommended future tracer-tracer studies from exhaust type emissions sources should release one tracer gas directly into compressor station exhaust, and a second tracer gas on the ground nearby in a typical location. This additional tracer can help discern potential plume loft that may impact downwind measurement assumptions. Vaughn et al. (2021) deployed this novel in-stack tracer gas method to measure combustion slip at 67 individual gathering and boosting compressor stations and showed that compressor emissions have likely decreased over the years due to deployment of emissions controls. As such, the authors concluded that the U.S. EPA GHGRP combustion slip emissions factor is inadequate and should not be used in the U.S. EPA GHGI. Additionally, current emissions factors in USEPA Method AP-42 for 4-stroke rich-burn engines likely do not reflect additional emissions controls that have become standard on certain systems, such as exhaust after treatment and the use of pre-chambers for ultra-lean combustion engines. Emissions estimates by Vaughn et al. (2021) provides an updated method to estimate national gathering and boosting combustion slip emissions by basin that should be considered in the subsequent GHGI.

In addition to calls for additional direct measurement studies, Zimmerle et al. (2020) provided three recommendations to continue to improve and update gathering sector emissions. These included: (1) require all operators to report counts of stations and separators to the U.S. EPA GHGRP; (2) clarify the definition of yard piping to include all ancillary equipment not included in other major categories, and; (3) report the driver type and loading for all compressors (e.g., electric, rich- and lean-burn engines, turbines, etc.).

Related to gathering pipelines, multiple authors have determined that significant uncertainty remains for leak frequencies per length of pipeline, and without better understanding of these leak frequencies, designing a sufficient sampling campaign to capture enough leaks to characterize the system remains a challenge. More frequent periodic screenings could help constrain *a priori* estimated leak frequencies, informing future sampling campaigns using similar vehicle-based sampling methods.

PSE

Super-emitters

Recommendation: Future study of super-emitters is needed; however, studies need to weigh feasibility with generalizability.

While further component-level root cause study would be valuable, volunteer bias in on-site methane surveys is an unavoidable confounding factor that undoubtedly limits generalizability of findings. Aerial survey technologies and airborne methane remote sensing systems can augment further study of super-emitters and should continue research and development in detecting, quantifying, and apportioning emissions throughout the supply chain. Recognizing there may be resource constraints in deploying large-scale studies, multiple authors have suggested deploying systems with less spatial resolution, but more frequent regional monitoring frameworks designed to capture temporal trends and subsequent abnormal operating conditions.

Temporal characterization

Recommendation: Studying temporal aspects of emissions (over spatial variability) has been particularly valuable.

The few studies that have assessed temporal aspects of emissions have been particularly informative. Cusworth et al. (2021a) most recently showed evidence of substantial temporal intermittency in source strengths from a few high-frequency sampling campaigns, including from many midstream facilities. They found that the large majority (~90%) of these super-emitter events were not persistent-type emissions sources occurring as one-offs or at relatively low frequency. While it is unclear if these types of low frequency events indicate that the leak was repaired, it has also been documented that many super-emitter events are easily fixable and preventable issues such as with unlit flares or open thief hatches on storage tanks.

The recommendations from Subramanian et al. (2015) still stand today in that more research is needed related to methane emissions associated with operational modes, and to ascertain whether any associations exist in better predicting conditions that cause certain emissions regimes or large emitting events.

3.3. Review of downstream methane studies

In this section, we will discuss the research gaps, recommendations, and details of the findings from 28 downstream-specific papers (1 critical review; 27 research articles) that were published between January 2015 and August 2020 and identified as focusing on downstream methane emissions from oil and gas, with an additional 36 papers reporting on emissions from downstream and overlapping with at least one other sector. While all the studies were considered in aggregate to produce the findings, research gaps, and recommendations presented here, not every paper will be mentioned explicitly.

PSE

We found that studies from downstream methane can be generally classified into three main categories, each elaborated within the following sections:

- Emissions pre-meter, focusing mainly on emissions from natural gas distribution pipelines (typically bottom-up);
- Emissions post-meter, focusing on emissions from natural gas appliances during all phases of operation and leaks from the indoor gas distribution (typically bottom-up); and
- City-wide emissions giving a big-picture overview of emissions from regional population centers (typically top-down).

3.3.1. Downstream methane: detailed findings

Methodology for measuring downstream methane

As technology has advanced in the past several decades and with more research identifying major research gaps in the downstream methane emissions, we noticed a shift in methodology towards primary data collection, mainly focused on quantifying methane emissions rather than simply characterizing them. Of the 28 downstream-specific studies, 93% were reported to collect primary data on methane emissions, although some studies were more methodological in nature and focused on measuring controlled emissions rather than leaks. Hopkins et al. (2016) reported on the most common measurement techniques in 2016; these techniques and others are summarized in Table 3.3 (upstream methane). In general, papers that provided data collection for downstream methane use one of several methods:

- Mobile measuring, usually in vehicles or airplanes;
- Site-based data collection, typically utilizing chambers or high-flow equipment; and
- Tower-based measurements, which can capture emissions basin-wide.

We describe findings from our analysis below in the following categories of possible sources of emissions: street emissions from the distribution network, post-meter emissions from residential buildings and appliances, emissions from power plants and refineries, and whole-city emissions using a top-down method which captures the entire air basin.

Measurements of street emissions

Prior to 2015, much work was performed both atmospherically and from point sources of leaks to identify the frequency and relative intensity of urban distribution leaks (Lamb et al. 2015; McKain et al. 2015; Subramanian et al. 2015). In the early 2010s, some studies began to systematically map urban distribution leaks using vehicles equipped with methane analyzers. They found that leaks were far more prevalent in areas served by aged, cast-iron mains, and suggested that cities accelerate their pipeline replacement programs to more quickly install new infrastructure (Jackson et al. 2014; Phillips et al. 2013).

Between 2015-2020, we found 10 studies which used mostly vehicle-based and some chamber-based measurements to investigate downstream emissions and are analyzed in this section. These studies advanced the field of emissions measurement for urban distribution systems by transitioning from qualitative leak detection to more quantitative measurements—either through a chamber-based approach, or by using a mobile plume integration system to quantify emissions from the data collected while driving. Additionally, some studies worked to apportion the sources to better understand where the gas was originating from—and to identify the gas as thermogenic or biogenic. Table 3.6 below outlines each of the studies, the region(s) that they measured, and the general scope of the measurements taken.

Table 3.6. Studies measuring urban distribution system methane leaks.

Location(s)	Scope	Study
12 cities in United States	Direct measurements of emissions from vehicles and modeling of overall emissions	Weller et al. 2020
Los Angeles, CA 4 unreported cities	Vehicle and walking measurements on streets, chambers to determine leak rate	Weller et al. 2018
Houston, TX	Mobile monitoring from residential areas, comparing median house age and density of gas heating units	Sanchez et al. 2018
Boston, MA Staten Island, NY Syracuse, NY Burlington, VT Indianapolis, IN	Driving surveys, some plume quantifications	von Fischer et al. 2017
Bakersfield, CA Sacramento, CA Berkeley, CA Oakland, CA	Compressed natural gas fueling stations, single-family houses, roadways	Fischer et al. 2017
Fairbanks, AK Los Angeles, CA San Diego, CA Salt Lake City, UT	Some mobile methane street data shown	Hopkins et al. 2016
Ithaca, NY	Pipelines, street pavement, manhole	Chamberlain et al. 2016
Boston, MA	Chamber-based measurements of metro street leaks	Hendrick et al. 2016
Bryan and College Station, TX	Vehicle-based measurements on road and near natural gas plant	Koch et al. 2016

PSE

Location(s)	Scope	Study
13 locations in the United States	Direct measurements of leaks, including distribution leaks, customer meters, and meter and regulating stations	Lamb et al. 2015
Manhattan, NY	Street sources (general category)	Payne and Ackley, 2015
Durham, NC Cincinnati, OH Manhattan, NY	Street sources and infrastructure, including manhole covers, valve boxes, and other locations	Gallagher et al. 2015

Weller et al. (2020) is the most recent and most comprehensive of the studies published in the last 6 years on urban distribution emissions. Weller used an advanced mobile leak detection platform to measure leaks from the urban natural gas distribution systems and estimates that there are 630,000 leaks from distribution means in the United States. In total, these leaks emit 0.69 Tg of methane per year. This estimate is approximately five times greater than the current U.S. EPA's estimate of emissions from the greenhouse gas inventory. As with other studies measuring emissions from pipeline leaks, this study also identifies that most emissions arise from those in the long tail (super-emitters). This study also identified that there were overall more leaks than previously thought. Data gaps and recommendations from all these studies are presented in more detail in the following sections.

Downstream post-meter emissions

Comparatively little research has been done on downstream post-meter emissions. To date, four studies (Fischer et al. 2018; Lebel et al. 2020; Merrin and Francisco, 2019; Sanchez et al. 2018) and one critical review (Saint-Vincent and Pekney, 2019) were identified that were published in peer-reviewed journals for emissions specifically from homes in North America. Lebel et al. (2020) and Fischer et al. (2018) both measured homes in California; Merrin and Francisco (2019) measured homes in Boston, Indianapolis, Illinois, and New York; and Sanchez et al. (2018) measured emissions from Houston. Fischer et al. (2018) and Merrin and Francisco (2019) both looked at whole-house emissions by measuring a subset of appliances in each house, and scaled emissions based on national inventory estimates. Fischer et al. (2018) also measured quiescent emissions (baseline slow-bleed emissions) from the house using a whole-home depressurization system to capture all the emissions, which included emissions from the appliances while they were off, emissions from any pilot lights present in the home, and emissions from leaks in natural gas infrastructure in the home. Fisher estimated that methane emissions from residences in California were equal to 15% of the state's overall methane emissions from natural gas. Lebel et al. (2020) narrowed the scope to water heating appliances, focusing on the difference between tankless and storage natural gas water heaters. They found that tankless water heaters on average emit 0.93% of the natural gas they use, more than double the relative emissions of storage water heaters, which emit just 0.39%. Most of these

storage water heater emissions arise from incomplete combustion of the pilot light, while tankless water heater emissions arise from pulses of unburned gas during on/off pulses.

While these studies were all direct measurements of sources, Sanchez et al. (2018) measured elevated concentrations from the street level in Houston during the heating season, stratifying samples across various residential zones. They measured methane concentrations from various residential zones and found elevated methane concentrations on residential streets in Houston and found that most of the emissions were thermogenic in nature. However, it wasn't entirely clear whether the emissions were coming from street leaks, home emissions, or even emissions from natural gas vehicles on freeways. Their vehicle-based method to collect the data didn't measure emissions directly from appliances, but they tested if emissions from homes affected local outdoor concentrations of methane. No correlation between elevated emissions and median house age and density of natural gas heating was observed.

Methane emissions from power plants and oil refineries

We reviewed two studies that specifically measured methane emissions from power plants and oil refineries (Hajny et al. 2019; Lavoie et al. 2017b). Both studies used a flyover approach, where they measured emissions using a Picarro Cavity Ring Down Spectrometer downwind of facilities in various locations throughout the United States. Lavoie et al. (2017b) suggests that emissions from refineries and power plants were measured to be larger than facility-reported emissions by a factor of 21–120 for power plants and 11–90 for refineries, but with the caveat that they were only able to measure a very small sample size. Hajny et al. (2019), measured stack plumes from 14 natural gas power plants in a snapshot of time using an aircraft to quantify the emissions.

Emissions were measured in a snapshot of time as the aircraft was taking its measurement and did not achieve long term measurements. Hajny et al. (2019) noted that this could be an issue because transient phases of power plant operations are not fully captured. For instance, emissions are only able to be measured fully when the facility is either on or off, and periods of ramping or start-up or shutdown are not captured. There is the potential for a large amount of emissions during these periods, as initial evidence collected by this study suggests.

City-wide methane measurement studies

Another group of studies attempted to better quantify overall urban emissions with various top-down approaches, either using towers positioned in the city or using aircraft or vehicles to collect concentration data over a wide spatial area. In total, we identified 10 studies that broadly measured citywide emission data since 2015, summarized in Table 3.7 below. The data gaps and recommendations from these studies are discussed in more detail in the following sections.

PSE

Table 3.7. Studies quantifying methane emissions from urban centers.

Location(s)	Approach	Study
Washington, D.C. Baltimore, MD	Two airborne platforms	Lopez-Coto et al. 2020
Washington, D.C. Baltimore, MD	Tower measurements and modeling	Huang et al. 2019
Washington, D.C. Baltimore, MD Philadelphia, PA New York, NY Providence, RI Boston, MA	Aircraft observations downwind of each city	Plant et al. 2019
Washington, D.C. Baltimore, MD	Aircraft measurements and grab samples for methane:ethane ratios	Ren et al. 2018
Indianapolis, IN	Aircraft mass balance to measure urban emissions	Heimburger et al. 2017
Los Angeles, CA	Ground-based stationary sites	Verhulst et al. 2017
Indianapolis, IN	Aircraft and tower measurements	Davis et al. 2017
Indianapolis, IN	Process-based estimation of emissions compared to atmospheric estimates	Lamb et al. 2016
San Francisco Bay area, CA	Flask samples or continuous monitoring at five locations; tracers used for source apportioning	Jeong et al. 2016
Boston, MA	Atmospheric transport modeling	McKain et al. 2015
Indianapolis, IN	Aircraft mass-balance approach	Cambaliza et al. 2015

3.3.2. Downstream methane: key research gaps

Broadly, research gaps from downstream methane emission studies focus primarily on the limitations of collecting data and the need to collect additional data to fill in gaps. In most cases, data is very limited, making it difficult to impossible to accurately assess the total magnitude of emissions from various downstream sources. Multiple strategies for additional data collection were presented in this literature, focusing on resolving the themes presented below.

PSE

Seasonality of emissions and temporal variability

Research Gap: More data needs to be collected in different seasons, particularly from top-down studies, and care should be taken to appropriately interpret the spatiotemporal variability in these measurements.

Studies have remarked on the difficulty of obtaining year-round, seasonally-resolved data, and the importance this data will play in better understanding downstream methane emissions (Lamb et al. 2016; Ren et al. 2018). Lopez-Coto et al. (2020) measured power plants and traffic counts hourly and demonstrated that 97% of the daily variability was explained by accounting for the sampling in time and space from sources that had large hourly variability. In some cases, the daily variability was larger than the uncertainty attributed to the method itself. Therefore, caution must be used when interpreting variability resulting from sampling conditions that are irregular spatiotemporally. Similarly, a technical paper by Huang et al. (2019) suggested that emissions from Baltimore and Washington, D.C. are 41% lower in the summer months than in the winter. The authors suggest that urban emissions are underestimated given the limited studies that quantify urban methane emissions, particularly the lack of studies that attempt to address seasonal and year-round emissions.

The studies reviewed here show that aircraft are primely situated to measure across a large spatial scale, and tower-based measurements are best for measurements across a long temporal scale. For example, Temporal variation is difficult to capture in aircraft studies because aircraft typically are only able to capture a snapshot of emissions at the time of flight. However, aircraft can capture emissions from a very wide spatial range in a short amount of time. In contrast, tower-based data collection points can capture a wide temporal variability because they are easily able to collect data for weeks or months at a time, but they are immobile and cannot capture the spatial variability of emissions.

Research Gap: Due to a lack of data, there isn't a consensus on whether seasonal trends exist from urban methane sources.

Currently, some studies predict that there is little to no seasonality from urban sources, while others hypothesize that modest seasonality may exist (McKain et al. 2015; Plant et al. 2019; Wong et al. 2016).

Post-meter

Research Gap: Post-meter emissions studies commented on the need for additional measurements from this line of research.

Saint-Vincent and Pekney (2019) comment on current data gaps which need to be filled in from the post-meter emissions: Sources of fugitive emissions from residential, commercial and industrial customers need to be identified, and it should be determined if there is a connection between methane emissions and housing type, or between appliance type or equipment age.

PSE

Research Gap: More data is needed on baseline emissions from interior natural gas infrastructure and emissions while appliances are off, with a focus on the largest emitters.

Other studies with primary data collection (Fischer et al. 2018; Lebel et al. 2020) suggest that leakage while the appliances are off plus baseline leakage from the interior natural gas infrastructure in the building have the potential to be major contributors to overall building emissions, in some cases emitting more than 90% of overall emissions. Importantly, the emissions from these types of leaks are independent of the total times the appliances are used, suggesting that the mere existence of natural gas appliances contributes to leaks. Because of the diffuse nature of these leaks, they are extremely difficult to identify and quantify, resulting in a major data gap identified by this group of studies.

Distribution pipeline and urban emissions

Research Gap: Leaks from distribution systems need to be identified and the relative magnitude of these emissions needs to be measured to understand the impact of the leaks to the environment and their contribution to overall urban emissions.

Much more work is needed in the area of urban methane emissions from distribution pipelines. These leaks are a hazard to safety, the climate, and even to the health of trees and other vegetation in the vicinity of the leaks (Schollaert et al. 2020). Currently, most of the focus is placed on the safety concerns of these leaks, which prioritize the detection and repair of the leaks which are closest to buildings and other infrastructure in confined spaces. Obtaining more measurements which focus solely on the quantity of methane emissions is still a gap that is beginning to be bridged through additional measurements. Quantified measurements of methane emissions are especially important so that the largest leaks can be identified and repaired quickly and efficiently.

Research Gap: More research on the source characterization needs to be performed, particularly from downstream urban emissions.

Biogenic methane from street sewers could be easily misidentified as leaks from distribution mains, potentially skewing the results higher than they are in reality (Fries et al. 2018). This could be done using ethane and or ^{13}C-CH_4 (carbon-13 methane), both of which have been utilized for this purpose and are commonly used throughout the oil and gas industry for source attribution.

Research Gap: Additional data on the distribution, frequency, and rate of fugitive methane across urban systems are needed. In addition, an understanding of how different management practices influence emissions rates is currently lacking.

Since data sources from urban environments are typically extremely diffuse and numerous, it is difficult to understand the distribution of individual leaks in cities. There are too many to

PSE

easily characterize by a bottom-up method and emissions are too diffuse to attribute emissions to particular sources using a top-down method.

3.3.3. *Downstream methane: recommendations*

With the downstream papers, many of the mitigation measures suggested are linked with the research gaps presented by these studies. Here, we discuss several of the themes presented in these papers and explicitly lay out the research recommendations that we identified as major themes from the group of downstream studies. We group the recommendations first by overarching theme and then by the specific component of downstream emissions.

Temporal and spatial variability in the sampling

Recommendation: To address the gap of lack of seasonality in measurements, additional observations of urban regions covering multiple regions/cities and seasons would provide further valuable, direct observational assessment of possible seasonality. Aircraft measurements utilizing multiple flights over different hours, days, months, and seasons, as well as multiple aircraft flying together with well-coordinated flight plans will maximize the spatial coverage of urban measurements at all times. Additionally, tower-based approaches are best suited for measuring temporal variability.

Lopez-Coto et al. (2020) discusses this research gap in using aircraft measurements and recommends multiple flights over a region over different hours, days, months, and seasons, as well as. This recommendation is echoed by many of the other aircraft-based studies we reviewed, as flights are resource-prohibitive and it is difficult to fully capture all the emissions from a city across long time scales.

Successfully capturing temporal variability is equally as important and is better achieved using sampling methods other than aircraft. McKain et al. (2015)measured emissions in Boston using a tower-based approach, coupled with modeling to calculate regional emissions. The authors concluded that it is imperative to sample both high emission events and diffuse low-emission sources. To achieve this, continuous or repeated samples need to be taken to understand the true distribution of the emission rates (McKain et al. 2015). Not having enough resolution in a chosen sampling method would cause sources of low flow to not be captured, and not having enough temporal coverage would result in missing short duration events with possible high emissions. Both types of gaps could result in an underestimate of emissions.

Recommendation: Greater amounts of data sharing between researchers would better assist at performing calculations and reduce some of the error associated with top-down measurements.

One issue with many of these methods is that it is difficult to quantify the emission rate from the concentration data collected, often leading to a large error term in the measurements when quantification is attempted. McKain et al. (2015) measured emissions from the greater

Boston area using tower-based concentration and modeling and stressed the importance of better quantification methods to understand the emission rate of methane from many of these sources. One of the easiest ways to achieve these better quantification results would come from a higher cooperation of data sharing, synthesis and dissemination between researchers collecting these datasets.

This level of quantification and data sharing will ultimately enable a better understanding of component-level emissions, which is one of the ultimate goals of these urban measurements. Cambaliza et al. (2015) states in their work from Indianapolis that this level of understanding will help with inventory development and emissions mitigation approaches, a process that was already underway in Indianapolis at the time of their publication. Data sharing will also help inform policies, especially at the urban level, and it is advisable to create open data. These methods should be fully disclosed and documented in a timely manner, which will enable the best use of these atmospheric data for informing policy (Verhulst et al. 2017).

Source attribution

Recommendation: Carbon-13 methane and ethane data should be collected in conjunction with methane data. This will be used for source attribution.

A major issue raised from these studies is that it is very difficult to attribute sources to the emissions that were observed. Isotopic measurements and measurements of methane:ethane ratios can provide insight into whether the methane observed is biogenic or thermogenic. It is possible that some measurements were confounded by biogenic sources such as sewage gas (Fries et al. 2018) and ensuring that the measurements are somehow able to reconcile the sources of the methane, particularly those which measure top-down, are of the utmost importance.

Recommendation: Better characterize the diffuse sources of urban methane emissions to better reconcile differences in top-down versus bottom-up inventories.

Many studies suggested that a potential source of unaccounted for gas, particularly from the bottom-up studies, is low-level, diffuse emissions (Lamb et al. 2016). This was particularly apparent in Lamb et al. (2016), as they measured methane emissions using both top-down and bottom-up approaches. They were unable to find any major point sources that were missing from the bottom-up inventory, so concluded that the residual likely came from widespread and relatively small diffuse sources. One possible source of these low-level emissions are post-meter emissions from buildings and appliances, discussed in more detail below.

PSE

Post-meter

Recommendation: Increase the sample size to create a nationwide emissions factor estimate for all post-meter emissions, including from individual appliances and from buildings overall.

Some ways that were recommended in the literature to achieve these recommendations is to first determine appropriate measurement conditions and measurement devices, which will help to focus the measurements on this goal. These measurement devices should be designed to best characterize the long-tail nature of these emissions and should focus on attaining the best results in the most efficient manner possible. Possible partnerships could be formed to share resources and spread out the geographic locations of sampling (Saint-Vincent and Pekney, 2019).

Recommendation: Characterize and reduce emissions from the pilot light and while the appliance is off.

Two studies (Fischer et al. 2018; Lebel et al. 2020) in particular discuss the need to reduce emissions from incomplete combustion of the pilot light. For instance, Lebel found that converting pilot light ignitions to electric sparkers would substantially reduce emissions from these appliances. Fischer et al. (2018) found that nearly 30% of the total appliance emissions are estimated to be emitted from the pilot lights. Merrin and Francisco (2019) did not measure pilot light emissions in their study.

Recommendation: Improving efficiency of the appliances will reduce natural gas emissions and will assuage safety and reliability concerns.

Two studies (Fischer et al. 2018; Merrin and Francisco 2019) discuss the need to improve the efficiency of the appliances. Merrin and Francisco (2019) points out that the emissions from some appliances may arise from safety or reliability concerns. Fischer et al. (2018) suggests that there is value in promoting a transition to renewable energy and high efficiency technologies, particularly for appliances for residential water, heating, and cooking.

Recommendation: Regular inspection of home fittings and appliances can reduce emissions.

All three studies (Fischer et al. 2018; Lebel et al. 2020; Merrin and Francisco, 2019) found that emissions from post-meter emissions had long-tail distributions, meaning that a small number of homes or appliances were responsible for a disproportionate amount of emissions. Therefore, inspecting homes for the largest emitting sources—either from loose connections in the gas lines or from the appliance itself—can greatly reduce emissions by eliminating the largest emitters.

PSE

Recommendation: A more thorough investigation of usage patterns should be undertaken to better understand a pathway for emissions reductions.

For example, tankless water heaters emit most of their methane during transient phases (i.e., turning from on to off or off to on), reducing the number of times that the water heater comes on can reduce methane emissions. This could be accomplished by having a small storage tank near the water heater to supply sufficient hot water for "short draws" or by eliminating a single handle faucet where the hot water heater is triggered even when the user did not intend to draw hot water. Whether other appliances also emit most of their emissions during on/off pulses is an area of future research (Lebel et al. 2020).

Urban distribution leaks

Recommendation: Create a framework to identify leaks most efficiently from urban distribution systems which evaluate both the safety concerns of the leak and the hazard level to the climate (i.e., the overall magnitude of emissions from the leak).

Regarding urban distribution leaks, in addition to the climate concerns of the leaked gas, the other major concern is safety of the leaks. If a leak is large enough and in a confined, poorly ventilated space, it has the possibility to cause an explosion. In fact, this has happened on several occasions; some notable instances were in Lawrence, Massachusetts[16] in 2018 and San Bruno, California[17] in 2010, highlighting the need to quickly address the issue of corroding pipes.

Gas companies tend to prioritize leaks based on safety concerns, and many use a three-stage system to classify leaks based on the danger level of the leak, considering both the amount of gas in the leak and proximity to infrastructure. Given that the magnitude of the leaks are only a component of the classification system, many of the studies in this group have recommended that large, but not necessarily dangerous leaks, also be prioritized in gas companies' leak detection and repair programs. Even for the largest leaks from the distribution system, it is very possible that many leaks have gone undetected as certain meteorological and physical subsurface conditions may prevent some of the leaks from being picked up, even by walking surveys (Ulrich et al. 2019).

One specific framework for prioritizing leaks was suggested by Hendrick et al. (2016) (Figure 3.9). In this figure, the deeper red colors represent leaks that are a higher safety hazard and are large leaks, therefore emitting more methane to the atmosphere. Deeper blue colors indicate

---

[16] News report from The New York Times, February 26, 2020:
https://www.nytimes.com/2020/02/26/us/columbia-gas-massachusetts.html

[17] News report from SFGATE, September 11, 2010:
https://www.sfgate.com/bayarea/article/San-Bruno-explosion-Some-victims-identified-3174910.php

PSE

low risk to both safety and climate. While it is easy to measure the concentration in the atmosphere because of these pipeline leaks, it is more challenging to fully quantify the emissions from the leaks. Many studies suggest that a quicker, more efficient method of leak quantification will be useful in helping prioritize the largest emitters.
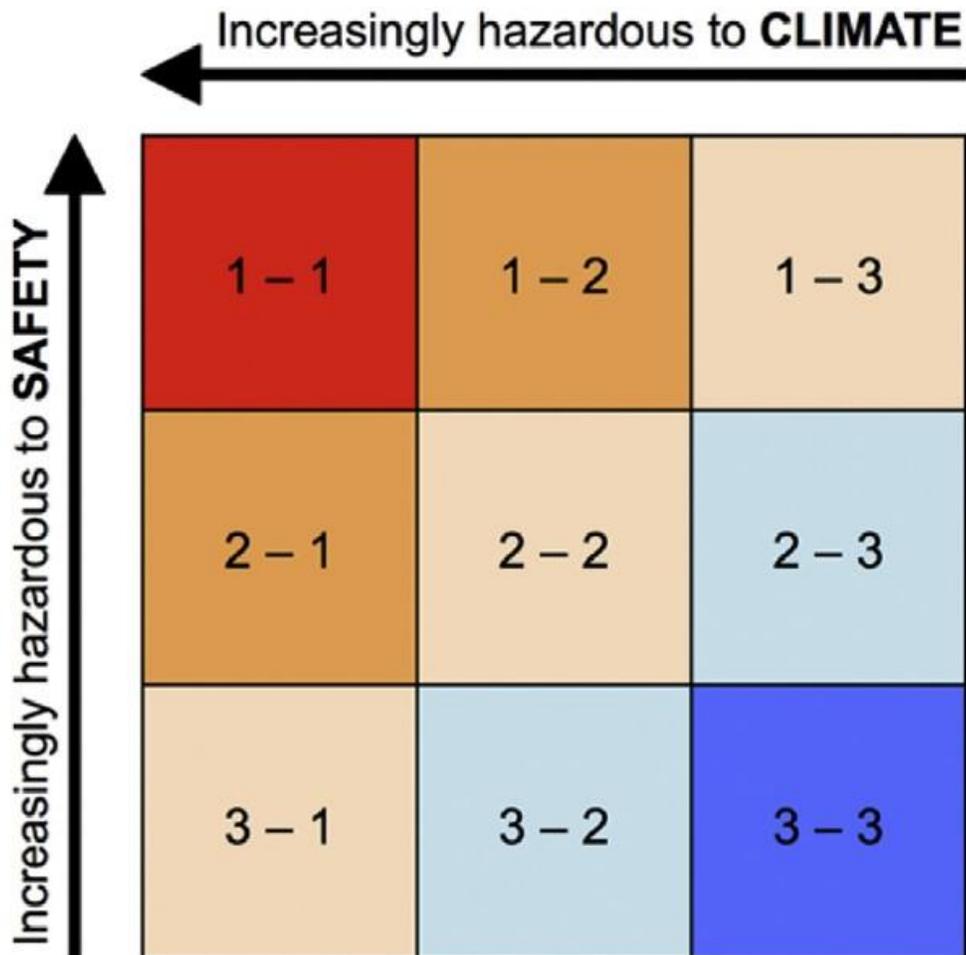
Figure 3.9. Framework to balance risk of natural gas leaks on climate and for local safety concerns. Deeper red colors represent leaks that are both a higher safety hazard and are large leaks. Deeper blue colors indicate overall lower risk to both safety and climate. Source: Reproduced with permission[18] from Hendrick et al. (2016).

---

[18] Reprinted from Environmental Pollution, Volume 213, Margaret F. Hendrick et al.
Fugitive methane emissions from leak-prone natural gas distribution infrastructure in urban environments, Pages 710-716, Copyright 2016, with permission from Elsevier.

PSE

Recommendation: Pipeline repair and replacement programs should be accelerated in many cities, particularly those with old, cast iron pipes. This will result in lower emissions over time.

These studies found that older pipelines, typically those made of cast iron, are most prone to leaking (Chamberlain et al. 2016; Lamb et al. 2015). Replacing these pipelines with more modern infrastructure, such as plastic pipelines will reduce overall leakage from the system.

Urban emissions

Recommendation: Implement an urban monitoring program to better understand the large scale and long temporal effects of natural gas leaks in urban areas.

This is important not only for urban distribution system leaks, but also for all downstream emissions. We need to tackle urban methane emissions by identifying mitigation targets, developing and implementing mitigations strategies, and monitoring levels of methane in urban areas to ensure that these mitigation efforts are effective (Hopkins et al. 2016).

Recommendation: Future studies will require additional measurements over a range of scales of emissions and will likely need to use multiple techniques.

These measurements should take place at facility-levels and should include source attribution to determine where the methane is originating from and whether it is thermogenic or biogenic. Atmospheric measurements coupled with activity-based estimates have been used in the past by governments which recent research has shown to underestimate true emissions. Going forward, a new interdisciplinary approach is required to study the effects of the economy, population, climate, geography, policy and development on methane emissions. Mitigation work should be undertaken to understand the connection of governance structures, culture, economy, and emitting sectors on the effectiveness of mitigation activities in different regions (Hopkins et al. 2016).

Recommendation: Understanding the sources of emissions is imperative before designing any research project aiming to quantify these emissions. When designing research projects involving direct measurements, scientists should understand the overall policy discussions taking place between regulators and the gas companies.

As Hopkins et al. (2016) noted, there is a cycle between regulators, researchers, and the parties that emit methane (i.e., end users, oil and gas companies, etc.) (Figure 3.10). This conceptual framework is a great starting point for understanding how the variety of measurement techniques discussed in the papers in this section for primary data collection fit into the big picture of overall urban methane emissions. Researchers should understand the partnerships needed between themselves, the emitters, and the policymakers to best structure their research in these locations. There are a variety of methane sources in urban environments, ranging from oil and gas fossil-based (thermogenic) sources to biogenic sources originating

from wastewater, landfills, or agriculture, and a substantial portion of urban emissions come from the burning of fossil fuels (Hopkins et al. 2016).
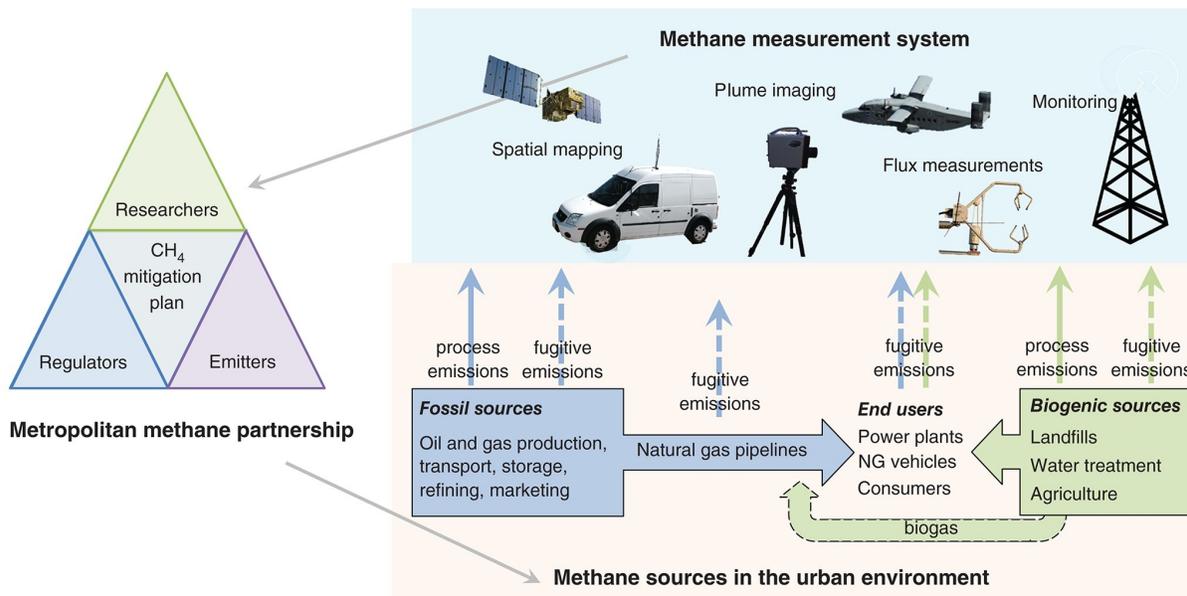
Figure 3.10. Conceptual framework for urban methane mitigation. Methane emissions in the urban environment originate directly from either fossil or biogenic sources or escape unintentionally from engineered systems and end users as fugitive emissions. These observations can inform a shared methane mitigation plan developed by a metropolitan methane partnership, consisting of emitters, researchers, and regulators, with the shared goal of adaptive management of urban methane for safety and climate mitigation. Source: Reproduced[19] from Hopkins et al. (2016), Figure 1.

The studies reviewed in this section have all advanced the science of methane emissions from downstream oil and gas operations between 2015–2020. It is evident that more data is needed in this sector, but it should be thoughtful about how resources are allocated for this data collection going forward. Data collection should be structured to influence policy decisions most effectively, such as whether to transition away from natural gas to all-electric energy infrastructure. As these decisions are made, it is imperative to have evidence-based information to inform these policy decisions, particularly from downstream sources. As these discussions are rapidly gaining momentum, it is essential to understand the dynamics of emissions from a system in transition.

---

[19]Reproduced as permitted by the creative commons license. No changes were made to the content of the reproduced figure.

4.0 Results: Health–damaging air pollutant (HDAP) studies (2015–2020)

Health–damaging air pollutants (HDAPs) – particulates and gaseous volatile or semi–volatile compounds that are hazardous to human health – are also emitted from the oil and gas sector. Broadly, HDAPs associated with the upstream oil and gas sector include many naturally occurring volatile compounds found in petroleum products; products of complete or incomplete combustion processes used for hydrocarbon development or transport; proppants used for well stimulation; and other chemical additive compounds used to facilitate hydrocarbon development or odorize final natural gas products. The detailed characterization of HDAPs associated with the oil and gas industry in the context of public health is provided in Appendix A.

While methane emissions hold implications at a global scale in the context of a changing climate, HDAP emissions contribute to more locally– and regionally–realized health risks and impacts. A wide array of populations may be exposed to HDAP emissions from the oil and gas sector, including but not limited to those residing near oil and natural gas extraction sites (e.g., upstream), near storage facilities (e.g., midstream), or further down the distribution line as fuel sources make their way into industrial, residential, or commercial settings (e.g., downstream). The literature prior to 2015 provides ample evidence that health–damaging air pollutants are emitted by the oil and gas sector (see Appendix A).

We identified 105 HDAP studies published between 2015–2020. Due to the rapid pace of developments in the science examining HDAP emissions, select studies from 2020–2021 that are outside the timeframe of our systematic review are incorporated throughout this section to provide additional context about recent developments. Below we discuss HDAP studies by oil and gas sector (upstream, midstream, and downstream) and identify key findings, research gaps, and mitigation recommendations from the peer–reviewed literature.

4.1. Review of upstream HDAP studies

The majority of studies evaluating HDAP emissions from the oil and gas industry focus on the upstream oil and gas sector. We identified 87 studies published between 2015–2020 that focused on HDAPs and upstream oil and gas development in North America. Of the 87 studies, 27 studies (~31%) report quantitative methane data from the upstream oil and gas sector. The studies that evaluate both upstream HDAPs and methane are discussed in detail in Section 5 of this report.

PSE

4.1.1.   Upstream HDAPs: detailed findings

Of the 87 studies examining HDAPs and upstream oil and gas development from 2015–2020, 22 studies (~25%) evaluated HDAPs in the context of public health (Table 4.1). These studies discuss HDAPs in specific oil and gas regions across the United States; a handful of studies focus on Canada. Sixteen studies compared ambient air pollutant concentrations to health–based guidance values or standards (e.g., National Ambient Air Quality Standards, NAAQS), or discuss HDAP contributions from upstream oil and gas sector to NAAQS exceedances. Two studies evaluate the health impact and economic impacts of HDAPs from upstream oil and gas development considering premature mortality attributable to air pollution. Four studies conducted quantitative health risk assessment using measured or estimated HDAP concentrations in air to estimate non–cancer and/or cancer risk.

Comparison to health–based guidance values and standards

When comparing measured or estimated air pollutant concentrations to health–based guidance values or standards, it is important to note that health effects may be observed at concentrations lower than health–based guidance values, and that these values may need to be updated over time as additional evidence on pollutant toxicity is made available. The health–based guidance values and standards most commonly used in the reviewed studies are:

- U.S. Environmental Protection Agency (US EPA) National Ambient Air Quality Standards (NAAQS) for criteria air pollutants;[1]
- Texas Commission on Environmental Quality (TCEQ) Effects Screening Levels (ESL);[2]
- Occupational Safety and Health Administration (OSHA) Permissible Exposure Limits (PEL);[3]
- National Institute of Occupational Safety and Health (NIOSH) Recommended Exposure Limits (REL);[4] and
- California Environmental Protection Agency (CalEPA) Office of Environmental Health Hazard Assessment (OEHHA) Reference Exposure Levels (REL).[5]

---

[1] US EPA. National Ambient Air Quality Standards. https://www.epa.gov/criteria–air–pollutants/naaqs–table

[2] TCEQ. Effects Screening Levels. https://www.tceq.texas.gov/toxicology/esl

[3] OSHA. Permissible exposure limits. https://www.osha.gov/annotated–pels

[4] NIOSH. Pocket guide to chemical hazards. https://www.cdc.gov/niosh/npg/pgintrod.html

[5] CalEPA OEHHA. Acute, 8–hour and Chronic Reference Exposure Levels. https://oehha.ca.gov/air/general–info/oehha–acute–8–hour–and–chronic–reference–exposure–level–rel–summary.

PSE

Broadly, appropriate comparison of air pollutant concentrations to health–based guidance values and standards requires that the air sampling methodology aligns with the duration of exposure being evaluated. For acute (short–term) exposures, that means air pollutant concentrations may be measured and compared over shorter durations, while chronic (long–term) exposures are appropriate to compare to average pollutant concentrations over longer periods of time (e.g., annual average). Additionally, occupational health–based guidance values are designed to apply to worker–related exposures during the typical work period (8 hours per day).

The majority of the 16 studies that compare measured or estimated air pollutant concentrations do not report exceedances of health–based guidance values or standards (Table 4.1). Evaluations of criteria air pollutants (e.g., ozone, particulate matter of <2.5 microns in diameter or $PM_{2.5}$) largely do not report exceedances of NAAQS near upstream oil and gas sites (Allshouse et al. 2019; Khalaj and Sattler, 2019; McCawley, 2015; Prenni et al. 2016). However, in Pennsylvania, Banan and Gernand (2018) note that at the current statewide setback distance of 500 feet (152 meters), $PM_{2.5}$ exposure limit exceedances occur frequently nearby well sites with simulated, higher–than–average emission rates and/or greater number of wells per well pad. Also in Pennsylvania, Long et al. (2019) reported $PM_{2.5}$ concentrations that exceeded chronic health–based air comparison values (i.e., U.S. EPA annual NAAQS) near oil and gas development sites. Additionally, while Richards and Brozell (2015) did not find respirable crystalline silica concentrations around frac sand–producing facilities in Wisconsin to exceed CalEPA OEHHA chronic reference exposure levels (RELs), Walters et al. (2015) did report exceedances of primary annual NAAQS for $PM_{2.5}$ (12 micrograms per cubic meter, $\mu g/m^3$) in samples collected around frac sand mines and processing sites in the same geographic region.

Studies that evaluated volatile organic compounds (VOCs) commonly associated with upstream oil and gas sites also largely did not find that ambient air concentrations observed exceeded health–based guidance values. Benzene was the most commonly evaluated VOC in the context of health–based guidance values.

In Texas, Hildenbrand et al. (2016) found that while benzene concentrations did not exceed OSHA 8–hr time–weighted average permissible exposure limit (PEL) of 1,000 parts per billion (ppb) at extraction sites, benzene concentrations did exceed the OSHA action level of 500 ppb at a few well pads. These findings suggest that individuals at extraction sites for an extended period of time (>8 hours) could be subjected to potentially harmful levels of ambient benzene if the detected concentrations persisted and additional mitigation measures were not taken. Also in Texas, Khalaj and Sattler (2019) found that modeled benzene concentrations exceeded the 1–hr TCEQ ESL in a scenario with strong sloped terrain, and exceed the annual TCEQ ESL across all modeled terrain types. These modeled maximum benzene emissions likely represent a reasonable worst–case scenario in the Barnett Shale, but likely underestimate emissions for areas with wetter gas (Khalaj and Sattler, 2019).

PSE

Long et al. (2019) used air quality measurements for 11 key air pollutants collected across 200 sampling locations nearby oil and gas development sites in the Marcellus Shale in Pennsylvania and New York to assess potential risks to human health. Few benzene, hydrogen sulfide, formaldehyde and $PM_{2.5}$ measurements exceeded chronic health–based comparison values (HBACVs), indicating that air pollutant measurements are typically below HBACVs. However, episodic spikes in emissions do lead to HBACV exceedances and warrant further site–specific investigations. Additionally, few air monitoring sampling locations were located <1,000 feet (<305 meters) from oil and gas sites, indicating the potential for higher concentrations to occur closer to these sources cannot be ruled out. Additional studies did not find measured or estimated benzene concentrations to exceed health–based guidance values at upstream oil and gas sites in Texas (Marrero et al. 2016; Olaguer et al. 2016a; Sablan et al. 2020; Schade and Roest 2018) and in Pennsylvania (Maskrey et al. 2016).

Instead of directly measuring air pollutants, Thompson et al. (2017) modeled scenarios with and without oil and production in the western United States to determine how emissions from oil and gas development contribute to ambient ozone (O_3) and $PM_{2.5}$ concentrations in the context of NAAQS. Broadly, emissions from oil and gas development were responsible for contributions to daily maximum 8–hr average ozone concentrations across vast geographic space, while modeled contributions to $PM_{2.5}$ were more localized to major oil and gas basins.

The maximum modeled contribution of oil and gas emissions to the non–attainment standard[6] in Colorado, Utah, New Mexico, Kansas, and Oklahoma ranged from 10–15 ppb, with the highest ozone concentration observed in eastern Texas (63 ppb). The largest modeled impact of oil and gas emissions to annual average of $PM_{2.5}$ was 1.6 µg/m^3 and was observed in New Mexico, with additional contributions >1 µg/m^3 observed in western Colorado, the Front Range in eastern Colorado, eastern Utah, central California, Wyoming, Oklahoma, Kansas, and northern New Mexico. These results suggest that emissions from the upstream oil and gas sector are contributing to a violation of air quality and health standards and threshold exceedances in certain regions in the western United States (Thompson et al. 2017).

Health impact and economic assessment of air pollution from the oil and gas sector

Two studies used the U.S. EPA Environmental Benefits Mapping and Analysis Program (BenMAP) to assess premature mortality and economic cost of air pollution from the upstream oil and gas sector (Fann et al. 2018; Roohani et al. 2017). Another study (Nsanzineza et al. 2019)

---

[6] Regarding the National Ambient Air Quality Standards (NAAQS) for ozone, an area is designated as nonattainment when the 3–year average of the fourth highest daily maximum 8–hr ozone concentration at a regulatory air quality monitor is greater than the standard (Thompson et al. 2017).

PSE

evaluated premature mortality and economic costs for both the upstream and downstream oil and gas sector; results from Nsanzineza et al. (2019) are discussed in Section 4.3.

Fann et al. (2018) estimated the number of air pollution–related deaths and adverse health symptoms attributable to the oil and natural gas sector in the United States projected for the year 2025. The authors define the oil and gas sector as including drill rigs, workover rigs, well completions, well hydraulic fracturing, heaters, storage tanks, mud degassing, dehydration, pneumatics, well venting, fugitives, truck loading, wellhead engines, pipeline compressor engines, flaring, artificial lifts, and gas actuated pumps, which, for the purposes of this report, primarily fall within the upstream subsector. Annual mean $PM_{2.5}$ concentrations from oil and gas activities by region ranged from <0.001 µg/m³ to 5.27 µg/m³, with Alabama, Colorado, Illinois, Louisiana, North Dakota, Ohio, Oklahoma, Pennsylvania, Texas, and Wyoming experiencing the largest $PM_{2.5}$ concentrations (Fann et al. 2018). Similarly, the authors found average 8–hr ozone concentrations to range from 8.12 ppb to 0.003 ppb, with Alabama, Louisiana, Nebraska, Oklahoma, Texas, and West Virginia experiencing the greatest summer–season ozone concentrations from the oil and natural gas sector. The $PM_{2.5}$– and ozone–related excess mortality burden was greatest in Texas, Pennsylvania, Ohio, Oklahoma, Illinois, California, Michigan, Colorado, Indiana, and Louisiana. In California, 59 $PM_{2.5}$–related and 14 ozone–related premature deaths are attributable to emissions from the oil and natural gas sector in 2025 (Fann et al. 2018).

Roohani et al. (2017) assessed the potential impacts of rapid natural gas development in the Marcellus and Utica Shales (across Ohio, Pennsylvania, New York, West Virginia) on regional ozone and particulate matter levels using three emissions scenarios (i.e., high, medium, low). In the medium–emissions scenario, 8–hr ozone values were predicted to increase by 2.5 ppb by 2020, and average annual $PM_{2.5}$ concentrations by 0.27 µg/m³, in areas with the most natural gas development. Premature deaths in the medium–emissions scenario were also predicted to increase by 200–460 annually, with health impacts and changes in air pollutant concentrations primarily driven by NO_X emissions (Roohani et al. 2017).

Quantitative health risk assessment

Four studies conducted quantitative health risk assessments by estimating non–cancer and/or cancer risk from exposure to HDAPs among populations in proximity to upstream oil and gas development (OGD) (Holder et al. 2019; McKenzie et al. 2018; McMullin et al. 2018; Paulik et al. 2016). Each study estimated health risks by distance from upstream OGD sites using measured or modeled HDAP concentrations. Three of these studies were focused on Colorado (Holder et al. 2019; McKenzie et al. 2018; McMullin et al. 2018), and one focused on Carroll County, Ohio (Paulik et al. 2016).

Colorado

Quantitative health risks assessments conducted in Colorado suggest that HDAP emissions from upstream oil and gas development contribute to health risks. McKenzie et al. (2018) used

air pollutant measurements collected 152–1600 m (500–5,249 ft) from upstream oil and gas development sites. The authors found that air pollutant concentrations and associated health risks increased with proximity to upstream oil and gas sites. Acute non–cancer health risks (with hazard indices >1), including neurological, hematological (i.e., blood), and developmental health effects, were noted for populations living within 152 m (500 ft) of an upstream oil and gas site. These health risks were largely driven by benzene, n–nonane and n–pentane. Chronic non–cancer health risks including hematological and developmental health effects were noted for people living within 152 m of a site, largely driven by benzene. Given the measured air concentrations of benzene and ethylbenzene, McKenzie et al. (2018) estimated that lifetime excess cancer risks exceeded the U.S. EPA de minimis threshold (1 case in one million) at all locations (including background) and began to increase over background at 501–610 m (1,673 ft to ~2,000 ft). While cancer risk associated with exposure to benzene exceeded the U.S. EPA de minimis threshold across all distances examined, lifetime excess cancer risk clearly increases with proximity from oil and gas development (McKenzie et al. 2018).

Using air monitoring data collected within 107–1128 m (350–3,700 ft) of oil and gas sites in Colorado, McMullin et al. (2018) found that combined hazard indices (HI) for all detected VOCs were slightly above one for acute (HI = 1.2) and chronic (HI = 1.3) exposures. While the authors do not present hazard indices by target organ system (e.g., respiratory system, nervous system), the authors note that most of the total risk was driven by n–nonane and benzene, which do not have similar target organ system effects. Considering concentrations of benzene and ethylbenzene, lifetime excess cancer risks were estimated at 4.3 cases per 100,000 individuals, exceeding the U.S. EPA de minimis threshold by more than an order of magnitude (McMullin et al. 2018). The lifetime excess cancer risk estimate reported by McMullin et al. (2018) falls within the range reported by McKenzie et al. (2018) within similar distances from oil and gas sites (5.7 cases per 100,000 to 1 case per 10,000).

Holder et al. (2019) clearly demonstrates a reduction in cancer risks and noncancer health risks associated with acute, subchronic, and chronic exposures as distance from oil and gas sites increases. Holder et al. (2019) also found potential for noncancer adverse health effects associated with acute exposures to 2-ethyltoluene, 3-ethyltoluene, toluene, and benzene, and for respiratory, nervous, and hematologic target organ systems. These results applied to the highest–exposed hypothetical individuals and were found to persist out to 610 m (2,000 ft) for benzene exposure, as well as for neurologic and hematologic effects.

It is important to note that Holder et al. (2019) only considered cancer risks associated with exposure to benzene and did not consider exposures to other possible or probable carcinogens. (2018). Because Holder et al. (2019) only considered benzene, total cancer risks were likely underestimated, although the degree of underestimation is unknown. Despite this limitation, the authors found that excess lifetime cancer risk below the U.S. EPA de minimis threshold was only achieved at a distance beyond 549 m (1,800 ft) from the well pad when considering various combinations of benzene exposure and risk estimate scenarios.

PSE

Ohio

Paulik et al. (2016) deployed 23 passive air samplers in rural Ohio for three to four weeks to quantify polycyclic aromatic hydrocarbon (PAH) concentrations by distance from at least one active natural gas well pad. Sampling sites were excluded if they were near other known sources of PAHs (e.g., airports, within city boundaries). Using PAH ratios, authors determined that PAH concentrations were from primarily petrogenic sources, not from combustion sources. As observed in other studies, health risk decreased with distance. Excess lifetime cancer risks associated with exposure to PAHs at sites closest to active wells (<100 m, or 328 ft) were estimated at 0.04 in one million and decreased 30% at beyond 1 mile (1,609 m) (0.027 in a million). Lifetime cancer risks observed were below the U.S. EPA upper threshold of one case per 10,000. Of note, passive sampling methods are useful to calculate average concentrations over deployment period, but do not characterize episodic spikes in emissions.

Summary of findings

Twenty–two studies have evaluated the public health impact of HDAPs associated with the upstream oil and gas sector. The majority of studies that compare measured or modeled air pollutant concentrations to health–based guidance values or standards have not found that ambient concentrations exceed standards. However, a few studies report NAAQS exceedances for $PM_{2.5}$ near well pads in Pennsylvania and near frac sand mining and processing sites in the upper Midwest (Banan and Gernand 2018; Long et al. 2019; Walters et al. 2015). Exceedances for benzene were noted in studies in Texas (Khalaj and Sattler, 2019). Additionally, while exceedances were not observed in the majority of studies, many authors acknowledged that oil and gas sites contribute to local and regional air quality concerns. Economic and health impact assessments of upstream oil and gas–associated air pollutant emissions conducted nationwide and in the Marcellus and Utica Shales indicate increases in ozone and $PM_{2.5}$ that contribute to additional premature mortality (Fann et al. 2018; Roohani et al. 2017). The majority of peer–reviewed quantitative health risk assessments noted non–cancer and cancer risks associated with upstream oil and gas emissions, with non–cancer risks driven by benzene, n–nonane, n–pentane, 2–ethyltoluene, 3–ethyltoluene, and toluene, and cancer risks driven by benzene and ethylbenzene (Holder et al. 2019; McKenzie et al. 2018; McMullin et al. 2018).

PSE

Table 4.1. Summary of studies that assess HDAPs in the context of public health (2015–2020).

Type of health assessment	State / Region	Pollutants considered	Main findings related to HDAPs	Study
Compare to health–based guidance values/standards (NAAQS)	CO	$PM_{2.5}$; black carbon	- Daily average $PM_{2.5}$ concentrations did not exceed NAAQS. - Mean $PM_{2.5}$ concentrations measured at the closest monitor (4.7 miles or 1.6 kilometers east of multi well pad site in an urbanized area along a major road) were 6.0, 9.6, 9.9, and 11.4 µg/m³ during drilling, hydraulic fracturing, flowback, and production, respectively. - Differences observed between phases of development are small in comparison to regional and seasonal variability.	Allshouse et al. 2019
Compare to health–based guidance values/standards (NAAQS)	PA	$PM_{2.5}$	- At current statewide setback distance of 500 ft (152 m), $PM_{2.5}$ exposure limit exceedances occur frequently with simulated higher than average emission rates and/ or greater number of wells per well pad. - Setback distances should be 2,415 ft (736 m) to ensure compliance with the daily average concentration of $PM_{2.5}$, and a function of the number of wells to comply with the annual average $PM_{2.5}$ exposure standard.	Banan and Gernand, 2018
Compare to health–based guidance values/standards (OSHA/NIOSH)	TX	benzene, toluene	- Benzene concentrations did not exceed OSHA 8–hr TWA PEL (1,000 ppb), but exceedances of the OSHA action level (500 ppb) were observed at a few well pads. - Toluene and total xylene isomer concentrations did not exceed OSHA PELs and/or NIOSH RELs. - These findings suggest that individuals at extraction sites for an extended period of time (>8 hours) could be subjected to potentially harmful levels of ambient benzene if the detected concentrations persisted and additional mitigation measures were not taken.	Hildenbrand et al. 2016

PSE

Type of health assessment	State / Region	Pollutants considered	Main findings related to HDAPs	Study
Compare to health–based guidance values/standards (Provincial Air Quality Objective)	Canada	NO_2, SO_2	- NO_2 and SO_2 concentrations over a 3–month period were below Provincial annual ambient air quality objectives.	Islam et al. 2016
Compare to health–based guidance values/standards (NAAQS; TCEQ ESL)	TX	CO, NO_X, PM_{10}, SO_2, benzene	- CO, NO_X, PM_{10}, SO_2 maximum concentrations were less than the 1–hr NAAQS. - Benzene exceeded the 1–hr ESL for strong sloped terrain and annual ESL for all terrain types. - Maximum benzene emissions modeled likely represent a reasonable worst–case scenario in the Barnett Shale, but likely underestimate emissions for areas with wetter gas.	Khalaj and Sattler, 2019
Compare to health–based guidance values/standards (NAAQS, US EPA RfC)	PA	$PM_{2.5}$, NO_2, SO_2, BTEX, H_2S, acetaldehyde, formaldehyde, n–hexane	- Few benzene, H_2S, formaldehyde and $PM_{2.5}$ measurements exceeded chronic health–based comparison values (HBACVs), indicating that air pollutant measurements are typically below HBACVs; however, episodic spikes in emissions do lead to HBACV exceedances and warrant further site–specific investigations. - Additionally, few air monitoring sampling locations were located <1,000 ft from oil and gas sites, indicating that the potential for higher concentrations to occur closer to these sources cannot be ruled out.	Long et al. 2019
Compare to health–based guidance values/standards (OSHA/NIOSH)	TX	hexane, benzene, toluene, ethylbenzene, and xylene	- Benzene and toluene concentrations did not exceed NIOSH RELs or OSHA 8–hr workplace exposure limits. - No explicit comparisons to standards were provided for other pollutants measured.	Marrero et al. 2016

PSE

Type of health assessment	State / Region	Pollutants considered	Main findings related to HDAPs	Study
Compare to health–based guidance values/standards (EPA RSLs)	PA	62 VOCs, including benzene, hexane	- Maximum concentrations from 24–hr air samples did not exceed subchronic/chronic RSLs for each detected VOC (1,4–dioxane, 2–butanone, 2–propanol, acetone, benzene, chloromethane, cyclohexane, dichlorodifluoromethane, ethanol, heptane, hexane, methylene chloride, toluene, trichlorofluoromethane, m,p–xylene).	Maskrey et al. 2016
Compare to health–based guidance values/standards (NAAQS)	WV	Respirable silica, $PM_{2.5}$, PM_{10}, ammonia, CH_4, O_3, NO_X, SO_2	- $PM_{2.5}$ and PM_{10} concentrations at drill sites did not exceed 24–hr NAAQS. - Average concentrations of ammonia, NO_X, O_3, SO_2 "did not indicate a concern for ambient or occupational exposures," though the author did not offer direct comparison to standards for these pollutants.	McCawley, 2015
Compare to health–based guidance values/standards (TCEQ)	TX	BTEX	- Ambient benzene concentrations from a modeled worst-case scenario did not exceed TCEQ short–term ESL (54 ppb). - Benzene concentrations from a modeled normal wind variability scenario did not exceed TCEQ's long–term ESL (1.4 ppb). - Similar findings were noted for toluene and ethylbenzene.	Olaguer et al. 2016
Compare to health–based guidance values/standards (NAAQS)	ND	$PM_{2.5}$, NH_3, HNO_3, SO_2, BC, O_3, NO_X, NO, NO_2, CO, methane, acetylene, EC, ethane, propane, n–butane, n–pentane	- Authors reported that observed concentrations "fall well below the NAAQs, they are elevated for a remote area, and in some cases are increasing".	Prenni et al. 2016

PSE

Type of health assessment	State / Region	Pollutants considered	Main findings related to HDAPs	Study
Compare to health–based guidance values/standards (OEHHA REL)	WI	Respirable crystalline silica	- Mean respirable crystalline silica concentrations at the fence line of frac sand–producing facilities were less than 10% of the CalEPA OEHHA chronic REL and consistent with background concentrations throughout the region.	Richards and Brozell, 2015
Compare to health–based guidance values/standards (TCEQ AMCV)	TX	Benzene, toluene, n–hexane, 2–methlypentane, 3–methylpentane	- Maximum concentrations of all measured pollutants did not exceed the TCEQ long–term air monitoring comparison values (AMCV).	Sablan et al. 2020
Compare to health–based guidance values/standards (TCEQ)	TX	H_2S, NO_X, ethene, toluene, benzene, isopentane, pentane, butane, propane, ethane and other NMHCs	- Average annual benzene concentrations at site did not exceed TCEQ long–term ESL (1.4 ppb). - No comparison to standards were presented for other pollutants.	Schade and Roest, 2018
Assess contribution of oil and gas emissions to health–based guidance values/standards (NAAQs)	Western half of United States	$PM_{2.5}$, O_3	- The maximum modeled contribution of oil and gas emissions to the fourth–highest daily maximum 8–hr O_3 concentration in Colorado, Utah, New Mexico, Kansas and Oklahoma ranged from 10–15 ppb, with the maximum O_3 concentration observed in eastern Texas (63 ppb). - The largest modeled impact of oil and gas emissions to annual average of $PM_{2.5}$ is 1.6 $\mu g/m^3$ and was observed in New Mexico, with additional contributions >1 $\mu g/m^3$ in western Colorado, the Front Range in eastern Colorado, eastern Utah, central California, Wyoming, Oklahoma, Kansas and northern New Mexico.	Thompson et al. 2017
Compare to health–based guidance values/standards (NAAQS)	Multistate (WI, MN)	$PM_{2.5}$	- $PM_{2.5}$ concentrations around frac sand mines and processing sites in some cases exceeded primary annual NAAQS (12 $\mu g/m^3$).	Walters et al. 2015

PSE

Type of health assessment	State / Region	Pollutants considered	Main findings related to HDAPs	Study
Health impact and economic assessment of air pollution	USA	$PM_{2.5}$, O_3	- Annual mean $PM_{2.5}$ concentrations from oil and gas activities by region ranged from <0.001 µg/m³ to 5.27 µg/m³ with AL, CO, IL, LA, ND, OH, OK, PA, TX, and WY experiencing the largest $PM_{2.5}$ concentrations. - Similarly, the authors found average 8–hr ozone concentrations to range from 8.12 ppb to 0.003 ppb, with AL, LA, NE, OK, TX, and WV experiencing the greatest summer–season ozone concentrations from the oil and natural gas sector. - The $PM_{2.5}$– and ozone–related excess mortality burden was greatest in TX, PA, OH, OK, IL, CA, MI, CO, IN and LA.	Fann et al. 2018
Health impact and economic assessment of air pollution	Multistate (PA, OH, NY, NYC, DC)	$PM_{2.5}$, O_3, NO_x, VOC, elemental carbon	- In the medium emissions scenario, 8–hr ozone values were predicted to increase by 2.5 ppb and average annual $PM_{2.5}$ concentrations by 0.27 µg/m³ in areas with the most natural gas development. - Premature deaths in the medium emissions scenario were also predicted to increase by 200–460 annually, with health impacts and changes in air pollutant concentrations primarily driven by NO_x emissions.	Roohani et al. 2017
Proximity–based quantitative health risk assessment	CO	47 VOCs (including BTEX)	- Noncancer adverse health effects were associated with acute exposures to 2–ethyltoluene, 3–ethyltoluene, toluene, and benzene, and for respiratory, nervous, and hematologic (i.e., blood) target organ systems. - These results applied to the highest–exposed hypothetical individuals and were found to persist out to 2,000 feet (610 m) for benzene exposure, as well as for neurologic and hematologic effects.	Holder et al. 2019

PSE

Type of health assessment	State / Region	Pollutants considered	Main findings related to HDAPs	Study
Proximity–based quantitative health risk assessment	CO	BTEX, total alkanes, n–hexane, etc.	- Within 500 ft (152 m) of active oil and gas development, the cancer risk estimate was 8.3 cases per 10,000, exceeding the US EPA upper threshold for acceptable risk (1 case per 10,000).	McKenzie et al. 2018
Proximity–based quantitative health risk assessment	CO	56 VOCs, including BTEX	- Acute and chronic non–cancer hazard quotients were not elevated for any individual VOCs. - Hazard indices (HI) combining exposure for all VOCs were elevated for acute (HI=1.2) and chronic (HI 1.3) exposures. - Lifetime excess cancer risk estimate for combined exposures was 4.3 cases per 100,000 individuals for 500 ft (152 m) and beyond.	McMullin et al. 2018
Proximity–based quantitative health risk assessment	OH	PAHs	- Acute noncancer health risks (with HIs >1) were noted for neurological, hematological, and developmental health effects for populations living within 500 ft (152 m) of an upstream oil and gas site, which were largely driven by benzene, n–nonane and n–pentane. - Chronic noncancer health risks were noted for hematological and developmental health effects for those living within 500 ft (152 m), which was largely driven by benzene. - Within 328 ft (100 m) of an active gas well, excess cancer risk estimate was 0.04 cases per 1,000,000, not exceeding the U.S. EPA upper threshold for acceptable risk (1 case per 10,000).	Paulik et al. 2016

BTEX – benzene, toluene, ethylbenzene and xylene; **NAAQS** – National Ambient Air Quality Standard; **CalEPA OEHHA REL** – California Environmental Protection Agency Office and Environmental Health Hazard Assessment Recommended Exposure Limit; **OSHA PEL**– Occupational Safety and Health Administration Permissible Exposure Limit; **NIOSH REL** – National Institute of Occupational Safety and Health Recommended Exposure Limit; **TCEQ ESL**; Texas Commission on Environmental Quality Effects Screening Level; **TCEQ AMCV**– Texas Commission on Environmental Quality Air Monitoring Comparison Values.

PSE

Peer–reviewed literature published in 2021

We also identified six additional peer–reviewed studies, published in 2021, that evaluate HDAP emissions from upstream oil and gas processes. Studies were conducted in California (Johnston et al. 2021; Okorn et al. 2021), Pennsylvania (Long et al. 2021), Texas (Holliman and Schade, 2021), West Virginia (Orak et al. 2021), and the U.S. (Buonocore et al. 2021). While outside the scope of the review period, due to the relevancy of these findings, we include a discussion of preliminary findings in this report.

Five of the six studies evaluated emissions of both HDAPs and methane from upstream oil and gas processes, the results of which are discussed in Section 5. Findings from the remaining study, Long et al. (2021), which evaluated HDAP concentrations near an oil and gas production site, are summarized below.

Conducted in Pennsylvania, Long et al. (2021) consistently detected 14 non–methane VOCs near oil and gas production sites, including: acetone, benzene, 2–butanone, carbon tetrachloride, chloromethane, dichlorodifluoromethane, ethanol, Freon 113, methanol, methylene chloride, n–hexane, propylene, toluene, and trichlorofluoromethane. For a subset of these VOCs, maximum concentrations were observed during the same well activity period for the majority of sites sampled. Benzene concentrations, for example, were highest during site construction and set–up, n–hexane and propylene concentrations were highest during flowback activities, and ethanol concentrations were highest during production (Long et al. 2021).

However, results summarized in Long et al. (2021) found all measurements of $PM_{2.5}$ and VOC monitoring at the three locations (approximately 1,000–2,800 ft (304–853 m) away from a well pad) were below health–based air comparison values, "and thus do not provide evidence of either 24–hr or long–term air quality impacts of potential health concern…"

4.1.2.   Upstream HDAPs: key research gaps

Many of the HDAP studies that evaluated pollutant releases from upstream oil and gas sources identified important research gaps in the peer-reviewed literature related to HDAP emissions. Below is a summary of key research gaps, as highlighted in the upstream HDAP studies.

Research Gap: There is a lack of thorough source characterization among the peer reviewed literature.

Improved characterization of source–specific emissions from upstream oil and gas development are needed (Ahmadov et al. 2015), including for primary organic aerosols and intermediate–volatility organic compounds (IVOCs). There are unknowns around evaporation of flowback wastewater, compounded by the fact that wastewater composition is generally unknown and may vary by site (Bean et al. 2018). Roest and Schade (2017) also evaluated emissions from liquid storage tanks in Texas and found that major gaps in data severely limit a

PSE

more precise, top–down assessment of emissions. The authors suggest that information related to the composition of both raw natural gas and vented gases from liquid storage tanks would be necessary for such an assessment to be conducted. Improved source characterization would improve estimates of HDAP emissions from upstream oil and gas development (OGD) and also allow for improved assessment of health risks and impacts associated with upstream OGD activities.

Research Gap: There is a lack of, and uncertain accuracy in emission factors for upstream oil and gas sources with known HDAP releases.

Emissions factors may not account for unintentional emissions as a result of malfunctioning equipment and fugitive leaks (Lan et al. 2015). A key research gap highlighted by numerous studies is the need for improved emissions factors for HDAPs from a variety of upstream oil and gas sources and activities, including flaring (Roest and Schade 2020; Schade and Roest 2016; Weyant et al. 2016) and liquids unloading and wastewater processing (Hecobian et al. 2019).

Research Gap: There is a limited assessment of temporal variability in emissions monitoring efforts in the literature among different upstream OGD sources.

Studies identified the need for additional ambient air monitoring at upstream oil and gas sites. This includes field campaigns designed to capture temporal variations in emission rates, which could be used to model additional temporal variations in atmospheric concentrations (Goetz et al. 2015; Hecobian et al. 2019; Khalaj and Sattler, 2019; Swarthout et al. 2015). Additional monitoring should also include monitoring at nighttime, because emissions do not disperse as much as during the daytime (McKenzie et al. 2018).

Additionally, continuous monitoring over time may be warranted, especially because there is potential for long–term exposure among populations living in close proximity to upstream OGD (Marrero et al. 2016). Continuous monitoring is important given the potential for temporal variation in emissions over time. For example, black carbon emissions may be influenced by gas composition, which may vary over time and across different upstream OGD sites (Weyant et al. 2016). Monitoring efforts should also be designed to identify sources of air pollution and quantify contributions of sources to ambient air pollutant levels with high density of monitors to increase the spatial resolution of air quality data (Field et al. 2015).

Brantley et al. (2015) found fugitive VOC emissions from liquid storage tanks to be highly variable, illustrating the need for more effective, economically feasible measurement strategies to identify and properly mitigate intermittent spikes in emissions. The authors proposed two primary research recommendations to address this data gap. First, the consistency in results seen between on–site samples and remote measurements using the U.S. Environmental Protection Agency's Other Test Method (OTM) 33A in Brantley et al. (2015) suggests that OTM 33A could serve as an effective inspection technique for identifying and properly mitigating large fugitive leaks at the well pad (Brantley et al. 2015). Second, results from the commercial high–volume sampler also suggest that emissions from condensate tanks

PSE

can be severely underestimated if results are not verified by corresponding canister measurements, a factor that should be considered when determining which measurement strategy to implement.

Systematic ongoing monitoring should include source tracers and air pollutants associated with upstream oil and gas activities to allow for measured and modeled exposure estimates of key health risk drivers (e.g., BTEX) (McKenzie et al. 2018). Additionally, monitoring should consider different phases on development (e.g., pad development through production) (Maskrey et al. 2016), including as drilling of new wells decreases and overall production in oil and gas basins and the potential impact of aging production infrastructure over time (Goetz et al. 2017). In addition to monitoring during upstream oil and gas activities, it's also important to have access to local background concentrations of HDAPs of interest (Maskrey et al. 2016). Studies conducted over multiple years are necessary to assess the influence of changes in upstream oil and gas activities on ambient levels of VOCs and criteria air pollutants (e.g, NO_X) (McDuffie et al. 2016).

Research Gap: The evidence suggests that HDAP emissions from different upstream oil and gas sites and oil and gas regions have questionable generalizability, meaning that findings in one region may not necessarily apply to a different oil and gas region, for example.

Studies raise the concern about generalizability of study findings across different upstream OGD sites within the same region and across different oil and gas regions (Goetz et al. 2015; Paulik et al. 2016). Efforts are needed to identify reductions in emissions from upstream oil and gas development (Prenni et al. 2016).

Research Gap: Current research is limited in scope with regards to the HDAPs considered in air monitoring and modeling efforts.

Future research efforts should examine a wide range of health–damaging air pollutants. For example, more research is needed on particulates, including trace metals bound to PM, ultrafine PM, respirable crystalline silica, and black carbon (Amoatey et al. 2019; Richards and Brozell, 2015; Weyant et al. 2016). And although secondary organic aerosols and ammonia from vehicle emissions have been examined for their role as criteria air pollutant precursors, these compounds are not evaluated in the oil and gas literature. Volatile chemical additives are also not evaluated in the literature, likely due to the site–specific use of chemicals, the lack of standardized methods for quantifying emissions in air, and the resultant lack of emission factors for these compounds. Nonetheless, studies highlight the need for additional research to support selection of chemical additives that minimize risks to the human health and the environment while also maximizing benefits to oil and gas development (Chen and Carter, 2020). Additionally, there are limited report measurements of radioactive compounds (e.g., radon) associated with the upstream oil and gas sector, which may be a particularly relevant consideration in certain oil and gas regions (e.g., Marcellus Shale) (Tian et al. 2017). Certain

PSE

assessments in the Marcellus Shale have not included measurements for important and relevant HDAPs, including formaldehyde and acetaldehyde (Schade and Roest, 2016).

Modeling should be conducted of combined impacts of various upstream oil and gas infrastructure that operate in proximity of each other, such as well pads and nearby gathering stations, and in different terrains (Holder et al. 2019; Khalaj and Sattler, 2019). Additionally, modeling efforts should consider a range of emission scenarios, including worst–case and average conditions (Hecobian et al. 2019; Khalaj and Sattler, 2019). In many cases, emission estimates require additional validation using other top–down or bottom–up methodologies. For example, one study of top–down satellite observations of tropospheric nitrogen oxides (NO$_X$) states the additional need to validate these findings using bottom up methodologies (Majid et al. 2017)

Research Gap: There is a need for further assessment of exposures to oil and gas-associated HDAP emissions, including an evaluation of the health risks and potential adverse health impacts that may occur among populations living near upstream oil and gas sites.

Exposure assessment of long–range transport of compounds is needed (Amoatey et al. 2019; Nye et al. 2020), particularly given the epidemiological literature that has shown impacts up to 10 miles (16.1 km) from upstream oil and gas sites (Deziel et al. 2020) and given that various criteria air pollutants will travel regionally (Fann et al. 2018). More research is needed to assess human exposures to upstream oil and gas–associated HDAPS (Marrero et al. 2016; McKenzie et al. 2018; McMullin et al. 2018; Paulik et al. 2018). Studies are needed that consider the cumulative health effects from multiple stressors, including but not limited to air pollution, for people living near upstream OGD facilities (Allshouse et al. 2019). Additionally, real–time monitoring should be coupled with real–time symptoms reporting, allowing nearby residents to be alerted of potential elevated concentrations of HDAPs and enabling symptom reporting in the context of measured ambient air pollutant concentrations. Furthermore, these air monitoring and symptom reporting efforts may then be used to assess individual exposures using biomonitoring (Olaguer et al. 2016b).

Risk assessment efforts should consider key health risk drivers (e.g., BTEX, other known human carcinogens) (Elliott et al. 2017; McKenzie et al. 2018; McMullin et al. 2018; Schade and Roest, 2018), and air monitoring efforts aimed at assessing health risk should be conducted to coincide with time frames that are compatible with health–based guidance values, including assessments of both acute and chronic exposures (McMullin et al. 2018; Walters et al. 2015) and incorporating low limits of detection to assess low–level exposures to various oil and gas–associated HDAPs (Maskrey et al. 2016). The potential additive or synergistic effects of exposure to compounds, including known human carcinogens, should also be considered (McCawley, 2015)

PSE

4.1.3. Upstream HDAPs: recommendations

HDAPs emitted from the upstream oil and gas sector pose risks to public health, particularly for populations that live in close proximity to oil and gas development activities and for those with underlying susceptibilities or vulnerabilities. While the health risks posed by HDAPs may be realized more intensely at the local scale, HDAP emissions from upstream oil and gas sources may also impact regional air quality through the long–range transport of gases and particles, and from the secondary formation of compounds that interact in the atmosphere (e.g., ground–level ozone formation, $PM_{2.5}$ secondary formation) (Lim et al. 2019).

Numerous studies acknowledge that the potential for adverse health impacts from exposure to HDAPs is the primary impetus for mitigating emissions from the upstream oil and gas sector (Elliott et al. 2017; Marrero et al. 2016; Paulik et al. 2018; Rich and Orimoloye, 2016). Mitigating HDAP emissions is of utmost importance for HDAPs that are known or suspected human carcinogens (e.g., benzene), for which exposure at any concentration or duration may increase the risk of developing cancer (Elliott et al. 2017). Super–emitters should also be identified and prioritized to realize the health and climate benefits of emission reductions. Super–emitters in the context of methane emissions are discussed in detail in Section 3.

Below we discuss explicit recommendations included in the peer–reviewed literature from 2015–2020 regarding mitigating HDAP emissions from upstream oil and gas sources and reducing exposure to HDAPs associated with the upstream oil and gas sector.

Recommendation: Advanced oil and gas practices and technologies, such as the use of closed loop systems to capture and transport flowback and recovered fluids, or strategies to control emissions from well completion activities, could further reduce emissions from upstream oil and gas development. While advanced emission control technologies are one strategy to reduce emissions, ongoing maintenance and monitoring (e.g., leak detection and repair efforts) are also necessary to mitigate HDAP emissions from upstream oil and gas sources.

Increased emission control measures are broadly recommended in the peer–reviewed literature because upstream oil and gas sources have been identified as significant sources of non–methane volatile organic compounds (NMVOCs), and these sources contribute to organic carbon mixing ratios and VOC reactivity. Therefore, upstream oil and gas sources are straightforward targets for VOC emissions controls (Abeleira et al. 2017). Advanced oil and gas practices and technologies could further reduce emissions from upstream oil and gas development. Given the variability in emissions observed across upstream oil and gas sources and sites, however, other studies point to mechanical inefficiencies of equipment at upstream oil and gas sites, rather than specific activities, as contributing to HDAP releases that impact local and regional air quality (Hildenbrand et al. 2016). These findings suggest that a combination of these two strategies – emission control technology and ongoing maintenance

PSE

and monitoring – would be the most effective approach to control and reduce HDAP emissions from upstream oil and gas sources.

Recommendation: Regulators should implement regulatory requirements, such as setback distance requirements for production sites, to protect sensitive populations (e.g., residences, hospitals, schools, playgrounds) from harmful exposures that can occur when vapor controls fail. Minimum surface setback distances should be long enough to adequately protect public health.

One way to mitigate exposures associated with oil and gas development, whether related to HDAP emissions or other exposures (e.g., noise and light pollution), is to increase the distance between a source (e.g., well pad) and a receptor (e.g., human populations). Many oil and gas states have implemented minimum surface setback distances between well pads and residences and other structures to mitigate impacts associated with oil and gas development. However, studies note that the low minimum surface setback distance in certain states and regions are not enough to mitigate exposures associated with upstream oil and gas development (Banan and Gernand, 2018; Olaguer et al. 2016). With respect to mitigating HDAP emissions from upstream oil and gas sites, Banan and Gernand (2018) simulated $PM_{2.5}$ emissions from a site in the Marcellus Shale in Pennsylvania under various scenarios and found that short–term air quality from drilling and hydraulic fracturing may exceed NAAQS for $PM_{2.5}$ and that the existing minimum surface setback distance of 500 feet (152 m) may not be sufficient to mitigate $PM_{2.5}$ in some cases. The authors propose that extending the setback distance to 2,415 ft (736 m) would help ensure compliance with the daily average concentration of $PM_{2.5}$ and that the number of wells on a single well pad is relevant to meeting annual average $PM_{2.5}$ exposure standard (i.e., NAAQS).

Recommendations to reduce specific HDAPs or subsets of HDAPs

Certain mitigation recommendations in the literature focus on specific HDAPs due to their toxic potential, contribution to overall emissions, or ability to serve as precursors for the formation of other air pollutants with health relevance. For example, Chen and Carter (2020) identify formaldehyde as a HDAP of concern and state that inhibiting the usage of formaldehyde – which can be used as a biocide or to reduce hydrogen sulfide during oil and gas production – could lead to significant reductions in the toxic potential of NMVOC emissions associated with liquid storage tanks. Chen and Carter (2020) also identify methanol, 2–propanol, and ethanol as targeted compounds for reducing emissions and associated occupational inhalation exposures related to storage tank operations.

PSE

Radon

Recommendation: Consider implementing emission control measures aimed at reducing overall radon emissions released from well sites. Recommended strategies for reducing radon include controlling production rates and reducing radium near the wellbore. Additional issues may arise if the latter strategy is implemented, however, as this hazardous waste stream would require proper disposal (Tian et al. 2017).

HDAPs associated with upstream oil and gas sources with radioactive potential are understudied in the peer–reviewed literature. Tian et al. (2017) modeled radon concentrations at the wellhead and noted simulated radon concentrations exceed safe levels and are likely to persist for numerous years.

Silica

Recommendation: Specific measures to reduce exposure to respirable silica should be considered, including (1) implementing sand belt enclosures and air curtains during mining operations; (2) improving sand transfer methods; and (3) repositioning workers away from sand dust when silica is being used at a well site (McCawley, 2015).

Exposure to crystalline (respirable) silica – a known respiratory irritant and human carcinogen – may occur at sites where silica (i.e., frac sand) is mined or at upstream development sites where silica is used as a proppant for well stimulation operations.

VOCs and NO$_X$

Recommendation: To reduce the formation of tropospheric ozone in oil and gas producing regions, consider implementing emission control technologies aimed at curbing ozone precursor emissions (e.g., VOCs and NO$_X$).

Perhaps the most common pollutant–specific mitigation recommendation in the literature is curbing ozone precursor emissions to reduce tropospheric ozone formation, particularly because many oil and gas development regions are in nonattainment for ozone under the Clean Air Act. Studies published since 2015 continue to indicate that alkane and NMVOC emissions from upstream oil and gas development contribute to ground–level ozone formation in oil and gas producing regions (Evans and Helmig 2017; Ghosh 2018; Lindaas et al. 2019; Pfister et al. 2019; Swarthout et al. 2015). VOC emissions from oil and gas development activities in certain regions (e.g., the Northern Front Range in Colorado) may contribute significantly enough to affect compliance with federal ozone standards (e.g., NAAQS) (Pfister et al. 2019; Swarthout et al. 2015) and contribute to overall frequency of high ozone days (Lindaas et al. 2019). NMVOCs and ammonia associated with oil and gas development may also contribute to formation of secondary organic aerosols and contribute to regional PM$_{2.5}$ emissions, as has been observed in examinations of vehicle emissions (von Stackelberg et al. 2013).

PSE

Recommendations to reduce HDAPs from specific upstream sources

Below we discuss emission mitigation recommendations specific to four upstream sources and activities: (1) wastewater handling and disposal; (2) flaring; (3) liquids unloading; and (4) storage tanks.

<u>**Wastewater handling and disposal**</u>

Recommendation: Future research should focus on the assessment of (1) the emissions associated with wastewater (e.g., flowback fluids, recovered fluids, produced water) handling and disposal, and (2) the evaporation of HDAPs from flowback wastewater and efficacy of associated emission mitigation measures.

Few studies investigate emissions associated with wastewater (e.g., flowback fluids, recovered fluids, produced water) handling and disposal. Bloomdahl et al. (2014) modeled air exposure to VOCs from flowback pits and did not identify human health risks that would necessitate additional strategies for flowback containment or protections for workers. However, the authors note that modeled scenarios did not mean that definitive conclusions could be drawn regarding risks to workers. On the other hand, Bean et al. (2018) found that wastewater evaporation can be a significant source of particulate matter, and that more research is needed to assess evaporation of HDAPs from flowback wastewater and the efficacy of emission mitigation measures.

<u>**Flares**</u>

Recommendation: Future research should focus on deriving updated emission factors for flares that reflect the variability in emission rates and gas composition observed during real life flaring activities. Additional efforts to identify reductions in emissions, including and beyond flaring, are still needed and should be considered.

Flaring has been identified as a significant source of benzene and NO_X emissions in the upstream oil and gas sector (Roest and Schade 2020; Schade and Roest 2018). Temperature at combustion significantly influences composition and rate of emissions, with low–temperature combustion flares contributing significantly to NO_X (Roest and Schade 2020). In a recent study evaluating flare activity in the Bakken Shale in North Dakota, Gvakharia et al. (2017) found that emissions from flaring activity varied widely, with the majority of methane, black carbon, and ethane emissions stemming from a small number of flares. In an examination of NO_2 in the Marcellus and Utica Shales, gas flaring was found to be a primary contributor to the increasing ambient NO_X concentrations observed in the region (Majid et al. 2017). More broadly, Prenni et al. (2016) evaluated air quality impacts for oil and gas activities in the Bakken Shale in North Dakota, and found that even if new state regulations aimed at reducing emissions from flaring met targets for 2020, up to 10% of produced gas will still be flared, exceeding the national average.

Storage tanks

Recommendation: Identify and implement vapor control measures aimed at reducing fugitive leaks from liquid storage tanks, especially from tank thief hatches. In addition to vapor control measures, implementation of regulatory setback distances from liquid storage tanks for sensitive populations (e.g., residences, hospitals, schools, playgrounds) should also be considered.

Brantley et al. (2015) found that fugitive leaks from liquid storage tanks (e.g., condensate, produced water) in Colorado were an important source of VOC and HAP emissions. On–site samples and corresponding remote measurements of condensate storage tanks identified thief hatches (also referred to as gauge hatches), which are used to access tank contents and provide pressure control, as the most frequently observed leak location, even with the implementation of vapor control technology (Brantley et al. 2015). These findings suggest that VOC and HAP emissions can still be emitted by condensate storage tanks, despite the presence of vapor control measures.

Results from Brantley et al. (2015) are consistent with findings in Eagle Ford Shale, Texas, in which liquid storage tanks were identified as a likely major contributor of the non–methane hydrocarbon (NMHC) enhancements observed in the region (Roest and Schade 2017). Specifically, Roest and Schade (2017) found storage tanks to contribute more than half of higher alkane emissions observed in the region, contributing 90% of n–butane, 83% of isobutane, 82% of propane, and 55% of ethane emissions. Brantley et al. (2015) found samples from produced water tanks, emissions of which were uncontrolled at the time of sampling, to have the largest fraction of NMVOC (91.8%) and HAP (18.5%) emissions of all the components tested.

Liquid unloadings

Recommendation: During liquid unloading, advanced plunger lift control algorithms have been used to reduce venting emissions (Allen, 2016) and should be considered.

4.2.   Review of midstream HDAP studies

We identified nine studies published from 2015–2020 related to health damaging air pollutants (HDAPs) associated with midstream oil and gas equipment and processes (Allen, 2016; Conley et al. 2016; Garcia-Gonzales et al. 2019; Goetz et al. 2015; Jacobson, 2019; Marrero et al. 2016; Oltmans et al. 2016; Russo and Carpenter, 2019). Three studies focused on emissions related to the 2015 Aliso Canyon gas storage facility blowout in Los Angeles, California (Conley et al. 2016; Garcia–Gonzales et al. 2019; Jacobson, 2019), and one was a critical review of the emissions associated with midstream compressors in New York (Russo and Carpenter, 2019).

PSE

4.2.1. Midstream HDAPs: detailed findings

Garcia-Gonzales et al. (2019) evaluated pollutant emissions released during the natural gas blowout event that occurred in 2015 at the Aliso Canyon gas storage facility in Los Angeles, California. The authors found uncontrolled leaks from midstream oil and gas sites, such as those observed at natural gas storage facilities, to be a potential source of harmful air pollutants. The authors recommended that facility-specific meteorological and air quality data-collection equipment be installed at natural gas storage facilities and that support of environmental surveillance after severe off-normal operation events be considered to ensure harmful exposures are properly monitored and mitigated.

Similarly, findings from Conley et al. (2016) suggest there is minimal variation over time in the benzene composition of leaking natural gas. The presence of benzene, a human carcinogen, during natural gas leaks is especially important when considering exposure risks to proximate populations downwind of blowout events. Similarly, hydrocarbon composition results taken from surface locations downwind from the leak are consistent with a leak of "pipeline–quality processed natural gas," with plume enhancements of ethane, propane, and butanes (i.e., natural gas liquids), pentanes and longer–chain hydrocarbons (i.e., condensates), and trace enhancements of benzene, toluene, ethylbenzene, and xylenes detected. These findings suggest that single–point failures of natural gas storage facilities such as Aliso Canyon can severely impede emission control strategies and present a potentially harmful exposure risk (Conley et al. 2016).

Russo and Carpenter (2019) evaluated emissions from compressor stations in New York. The magnitude of releases varied greatly among different compressor stations, some of which can be explained by differences in the equipment used on–site. The authors found the largest releases in emissions,[7] by far, to be nitrogen oxides (NO_x) – an ozone precursor – and carbon monoxide (CO), followed by VOCs, known carcinogens including benzene, formaldehyde, and 1,3–butadiene, coarse particulate matter (PM_{10}), sulfur dioxide, and ammonia. The detection of ammonia is especially important when considering the potential for secondary formation of particulate matter, which ammonia is known to contribute to. These findings, the authors conclude, "add to the evidence for urgency for the replacement of fossil fuels with renewable sources of energy in order to both protect human health and reduce the immediate and long–term threats arising from climate change."

We also identified one additional peer–reviewed study, published in 2021, that evaluates pollutant emissions from leaking natural gas distribution pipelines (Anderson et al. 2021).

---

[7] Pollutants listed represent the top six pollutants emitted by compressor stations; see Table 2 of Russo and Carpenter (2019) for the full list of pollutants and their associated magnitudes.

PSE

While out of the scope of the review period, there is a limited number of studies that focus on emissions from midstream processes. As a result, we include a discussion of its findings and key mitigation recommendations in Section 5 of this report.

4.2.2.   Midstream HDAPs: key research gaps

Several studies include key research gaps that pertain to HDAP releases from midstream sources. All nine studies published from 2015–2020 that evaluated emissions from midstream oil and gas sources considered both methane and HDAP atmospheric releases.

Research Gap: Current research gaps are largely driven by the fact that readily available, rapid–response methods are not made widely available for use. Research focusing on the air quality and human health impacts of major unanticipated air pollutant releases fail to rely on rapid–response monitoring methods. Such monitoring networks are important for preventing single–point failures at natural gas storage facilities from going undetected, which can severely impede emission control strategies.

Findings from Conley et al. (2016) demonstrate the ability for single–point failures from natural gas storage facilities, such as the 2015 Aliso Canyon blowout event, to severely impede emission control strategies, especially when readily available, rapid–response methods are not made widely available for use. The authors provide the following research recommendations to address this key data gap: Implement timely, airborne sampling protocols that provide accurate leak rate measurements with high spatial and temporal resolution and chemical composition data for specific fugitive emission sources. The study states,

"Such information can help to document human exposure, formulate optimal well–control intervention strategies, quantify the efficacy of deliberate control measures, and assess the "air quality impacts of major unanticipated chemical releases to the atmosphere" (Conley et al. 2016).

Research Gap: Currently available research fails to: (1) properly characterize exposure risks and the complex emissions associated with natural gas facilities; and (2) provide a comprehensive picture of all HDAPs released during a blowout event.

Garcia–Gonzales et al. (2019) also evaluated the 2015 Aliso Canyon blowout event and found currently available data on natural gas facilities to be severely lacking. In addition, the authors found federally established health–based thresholds for single pollutants were inadequate at providing aggregate risk estimates from concurrent or many potential exposures. Similarly, health benchmark levels do not consistently address the risks associated with low–level or chronic exposures, have not necessarily kept up with the latest evidence of health harms, and do not adequately protect vulnerable populations.

PSE

Research Gap: Publicly available emissions inventories for compressor stations severely underestimate the true emissions released during operation. The lack of primary data collection studies focused on long-term monitoring of compressors and associated process equipment make it difficult to determine their true emissions concentrations.

The final study identified in this section evaluated emissions associated with compressor stations along natural gas pipelines (transmission) in New York (Russo and Carpenter, 2019). While not a primary data collection study, due to the limited available literature assessing midstream oil and gas HDAP emissions, we include a discussion of results here. The authors relied on two federal datasets that have emissions data for compressor stations in New York: the U.S. EPA's National Emissions Inventory and its Greenhouse Gas Inventory. Of the 74 compressor stations operating in New York from 2008–2014, only 18 reported emissions to the U.S. EPA. This finding suggests that emissions from compressor stations are severely underestimated for the state, and more should be done to obtain and integrate relevant pollutant information from all compressor stations into federal emissions inventories to ensure that that communities are protected from potential harmful air pollutant exposures due to midstream HDAP releases. By doing so, we would gain insight into the true magnitude of emissions and associated risks from compressor stations along natural gas pipelines.

4.2.3.   Midstream HDAPs: recommendations

As mentioned previously, all midstream studies identified in Section 4.2 considered both methane and HDAP atmospheric releases; therefore, mitigation and policy recommendations highlighted in these studies often have co–benefits, achieving reductions in both methane and HDAP emissions from midstream sources. As such, we include a summary of key findings, conclusions, and mitigation recommendations, which are specific to methane and associated HDAP releases from the midstream sector, in Section 5.2.1. and 5.2.2 of this report.

4.3.   Review of downstream HDAP studies

In total, we identified 17 peer–reviewed studies focusing on HDAPs and downstream oil and gas development in North America published from 2015–2020. Of the 17 studies included, 12 focused on downstream oil and gas development. Ten of the studies focused solely on HDAPs and five studies included data on methane and HDAPs. Five studies either compared pollutant concentrations to established health standards or conducted a form of exposure or health analysis. One study identified petroleum refineries as the primary source of enhanced lanthanum and total lanthanoid concentrations (rare earth elements), in paired indoor–outdoor $PM_{2.5}$ ratios at a high school in the Houston Ship Channel region, Texas (Bozlaker et al. 2017). Fluidized–bed catalytic cracking (FCC) is an essential conversion process used in many refineries to produce additional gasoline. Emissions from refinery FCC units contain airborne lanthanoids, which are readily absorbed by humans, exposure to which can result in a variety of toxicological and respiratory symptoms (Bozlaker et al. 2017). Du and Turner (2015) similarly evaluated $PM_{2.5}$ lanthanoid elements in Roxana, Illinois, along the fence of a petroleum

refinery; however, they were unable to attribute the $PM_{2.5}$ to the emissions from petroleum refinery FCC units.

4.3.1. Downstream HDAPs: detailed findings

Four of the 14 studies either compared pollutant concentrations to established standards based on health risk, or completed a health analysis based on the exposures measured in the study (Mitchell et al. 2016; Sanchez et al. 2019; Singer et al. 2017; Stidworthy et al. 2016). One additional study, Nsanzineza et al. (2019), evaluated the impacts of emissions fee scenarios on mortality reduction. Studies were inconsistent regarding health risk associated with air pollution emissions from downstream sources. However, when considering at–home exposures from the use of natural gas in ovens and stovetops, evidence suggests that people can be exposed to short–term elevated concentrations of harmful pollutants, specifically NO_2 (Singer et al. 2017). Singer et al. (2017) demonstrated that cooking with natural gas can result in acute exposure to NO_2 concentrations exceeding the NAAQS standard of 100 ppb with 1–hr NO_2 levels exceeding the standard, however, that the exposure can also be minimized through the use of a range hood or other ventilation system. The radon exposure that comes from the use of natural gas from cooking has been shown to be minimal (although there can be increased levels of radon in natural gas in pipelines that service homes).

Sanchez et al. (2019) evaluated three years of air monitoring data near a petroleum refinery in Richmond, California, and found that monitored concentrations of VOCs, hydrogen sulfide, ammonia, $PM_{2.5}$, and black carbon remained below the reference exposure levels (RELs) and NAAQS standards for Richmond. There were a few occasions during the study period where benzene levels exceeded the 8–hr REL; however, analysis indicated that the elevated benzene levels were related to wildfires and unrelated to the refinery. Particulate matter concentrations were also generally below the NAAQS standards, except for a period in October 2017 during a wildfire episode in California. The results of this study suggest that the refinery does not contribute to elevated concentrations of air pollutants above threshold limits in the Richmond area.

Mitchell et al. (2016) evaluated lung cancer risk from radon in Marcellus Shale gas provided to homes in the Northeastern United States. for natural gas cooking and heating. The calculated mean lifetime excess risk of lung cancer from radon exposure while cooking at home with natural gas was 1.8×10^{-4} and the calculated mean lifetime excess risk of lung cancer from radon exposure from space heating was 1.1×10^{-4} to 3.9×10^{-3} (Mitchell et al. 2016). Although there is evidence of increased excess risk of lung cancer, the radon exposure that results from cooking or heating with natural gas does not have a significant impact on lung cancer cases compared to the lung cancer cases as a result of background radon exposure (Mitchell et al. 2016). Measures can be taken to reduce individual exposure, such as the use of ventilation (e.g., range hoods or microwave vents) while cooking, and using space heaters with vents (Mitchell et al. 2016). Stidworthy et al. (2016) also evaluated radon exposure from burning natural gas in power plants. Radon concentrations attributed to power plant emissions were below the U.S.

PSE

EPA action level. However, the radon concentrations of the natural gas that flowed into the power plant did have elevated concentrations of radon that went above the U.S. EPA action level. It is likely that concentrations of radon were diluted during combustion and therefore resulted in lower concentrations of radon being emitted suggesting that power plants are not a significant source of radon in downwind areas and not a substantial human health concern (Stidworthy et al. 2016).

Nsanzineza et al. (2019) modeled emissions of ozone from energy and power production and the impacts of ozone on air quality in Colorado, northern New Mexico, Utah, and Wyoming from 2011–2030. The study compared different scenarios of oil and gas production and shares of electricity from natural gas, coal, and renewable sources of energy as the baseline year, and then predicted the ozone emissions associated with each scenario. According to the models, ozone concentrations declined between 2011–2030, with a 3.5 ppb median decrease in maximum 8–hr daily ozone concentrations and a 7.1 ppb, 90th percentile decrease in maximum 8–hr daily ozone concentrations (Nsanzineza et al. 2019). These reductions in ozone concentrations correspond to prevention of 200 premature deaths per year (Nsanzineza et al. 2019). The models also showed that VOC emissions from energy and power production would be higher in 2030 than in 2011 for all scenarios considered in the study (as a result of more oil and gas production); however, reductions in NO_x and SO_2 were seen in some scenarios between 2011–2030 (Nsanzineza et al. 2019). The authors suggest that adopting GHG fees would reduce premature mortalities and could save almost an additional $200 million in costs.

4.3.2.   Downstream HDAPs: key research gaps

Several studies highlight key research gaps that pertain to HDAP releases from downstream sources. Below is a summary of key research gaps found in the 2015–2020 peer-reviewed literature.

Research Gap: The peer-reviewed literature fails to incorporate existing information of air pollutant emissions into analyses that evaluate gas releases.

The relationship between methane and HDAP emissions in the downstream oil and gas sector is understudied, as methane emissions are often not evaluated alongside one or more HDAPs. Of the 13 downstream studies evaluated in this review, only two studies evaluated methane and HDAPs simultaneously; however, neither of these studies explicitly conducted a correlation analysis between HDAPs and methane emissions. Nsanzineza et al. (2019) used two models to estimate emissions from various energy sectors and evaluated the change in methane and VOC emissions under three different energy and power production scenarios. However, although the study had modeled data on methane and VOCs, the changes in methane and VOC emissions were evaluated separately, without evaluating correlation or trends between changes in methane and VOC emissions.

PSE

Sun et al. (2020) modeled U.S. refinery emissions of greenhouse gases, including methane, and criteria air pollutants by source looking at both GHG and criteria air pollutant (CAP) emissions from refinery primary products and secondary products, providing an enhanced understanding of refinery emissions. Although the study simultaneously evaluated GHG and CAP emissions, similar to Nsanzineza et al. (2019) the CAP emissions were not evaluated in the context of methane emissions. This lack of incorporation of data on the co–emission of methane and HDAPs inhibits the understanding of the sources of these pollutants, limits the information available to develop strategies to mitigate and reduce the emissions of these chemicals, and may lead to inaccurate health assessments. Furthermore, it is important to look at how methane and HDAPs are co–emitted because there could be differences in the co–emissions by equipment, sector, maintenance level, and activity periods, etc., which can all influence the relationship between methane and HDAP emissions.

Research Gap: There is a scarcity of consistent source profiles for VOC emissions across studies.

A major challenge in evaluating HDAP emissions is lack of consistent source profiles for HDAPs. Sanchez et al. (2019) found that VOC source profiles vary in the literature between studies, making it difficult to accurately identify sources of emissions and reducing the ability to accurately quantify the emissions from oil and natural gas activities. Furthermore, the variability in source profiles for HDAPs makes it difficult to compare studies. The lack of source–specific emissions estimates also limits overall assessment of emissions in an area because estimates of source–specific emissions may vary between emissions estimate methods.

Research Gap: The peer-reviewed literature suggests that the accuracy of currently available emission factors is uncertain.

Large uncertainties surround HDAP emissions. In many cases, uncertainty in emissions estimates occur because the composition of the emissions from refineries (and other oil and gas activities) varies in the type and quantity of chemical emitted (Hoyt and Raun, 2015). Uncertainty in emissions estimates could be improved by taking primary measurements of emissions. Spatial and temporal variations, as well as meteorological factors such as wind speed and wind direction, also contribute significantly to uncertainty in HDAP emissions (Hoyt and Raun, 2015). Another source of uncertainty arises from scaling of emissions and non–reporting of emissions by facilities (Sun et al. 2019). Studies often do not consider differences in emissions from different downstream activities or technical equipment.

Research Gap: The peer-reviewed literature provides a limited assessment of variability in emissions among different downstream oil and natural gas activities, both across source sectors and across temporal factors (i.e., monitoring during the day vs. at night, adjusting for influence of insulation for indoor studies).

Limited assessment of variability in emissions among different downstream oil and natural gas activities, and at different endpoints, has contributed to a gap in the understanding of

emissions and exposure to HDAPs. Most emissions data is collected during the day and few studies evaluate emissions from activities that occur overnight, which may lead to over or underestimation of total HDAP emissions. While Hoyt and Raun (2015) noted previous studies demonstrated that measured emissions did not vary significantly overnight, this may not be the case for all activities. For example, Mitchell, Griffin, and Casman (2016) identified a limitation in evaluating HDAPs in indoor exposure studies: not including a factor in model calculations that accounts for the influence of tight construction in the buildings, which can influence air circulation and exposure to HDAPs.

Research Gap: There is a paucity of exposure and health assessments for HDAPs focusing on the downstream oil and gas sector in the peer-reviewed literature.

Only one study included in the HDAPs and downstream oil and gas sector conducted a health analysis analyzing cancer risk from exposure to oil and gas activity. Mitchell et al. (2016) evaluated the mean lifetime excess risk of lung cancer from radon exposure from space heating and from cooking at home with natural gas and compared those risks to excess cancer cases as a result of background radon exposure. Another study, Nsanzineza et al. (2019) evaluated how reductions in ozone concentrations from energy and power production corresponded to a prevention of 200 premature deaths per year. Three other studies compared emissions concentrations of HDAPs to national and state standards, and health–based guidelines for emissions, to determine if the emissions exceeded normal exposure levels. There were no studies included in the HDAP downstream oil and gas sector that evaluated acute or chronic noncancer health impacts of exposure to any HDAPs.

Research Gap: The peer-reviewed literature considers a limited number of HDAPs in their air monitoring and pollutant modeling efforts.

Many of the studies included in this review evaluated a limited number of HDAPs. Six studies evaluated only one HDAP in their research, two studies evaluated three HDAPs in their research, and five studies evaluated three or more HDAPs. Furthermore, many of the studies evaluated the limited chemical species (e.g., criteria air pollutants). Evaluating the emissions of HDAPs not only in the context of methane emissions, but also in the context of other HDAP emissions, is critical to understanding exposure to health–damaging chemicals and evaluating health risk. Exposure to multiple chemicals can often have compounded health effects compared to exposure to a single pollutant.

Research Gap: Review of the downstream HDAP literature highlighted a lack of research focused on HDAP emissions and HDAP–methane co–emissions; specifically, the peer-reviewed literature lacks a detailed evaluation of these emissions by component and mitigation strategy, which limits the ability to accurately monitor and predict emissions and evaluate the associated environmental and health impacts. There is also a need for health analyses and longer–term monitoring periods to better capture emissions from refineries and other downstream facilities.

PSE

Studies consistently report over or underestimation of emissions depending on the component studied and measurement method, due to the inability to account for emissions from all sources. Sanchez et al. (2019) identified a scarcity of continuous long–term monitoring of emissions near refineries, underscoring the need for longer–term monitoring periods to capture a better representation of emissions from refineries, as well as other facilities. There is even less information regarding health analyses from exposure to downstream HDAP emissions. For health analyses of HDAPs Burns et al. (2017) recommends evaluating HDAP exposures by specific task at refineries, because emissions exposures vary by activity.

4.3.3. Downstream HDAPs: recommendations

Below we discuss explicit recommendations included in the peer–reviewed literature from 2015–2020 specific to mitigating HDAP emissions from downstream oil and gas sources and reducing exposure to HDAPs associated with the downstream oil and gas sector.

Nsanzineza et al. (2019) focused on policy interventions and their impacts on emissions. Overall, the study supports the implementation of GHG fees to promote reliance on renewable energy and decrease HDAP and methane emissions from oil and natural gas production and electricity generation. As GHG fees can result in co-reductions in HDAP emissions, this recommendation and supporting findings are discussed in detail in Section 5.3.2 of the report.

Recommendation: Regulators interested in targeting emissions reductions from downstream oil and gas sectors should consider using mathematical programs to help identify the most appropriate control strategy for specific refineries (as well as cost), thereby helping refineries reach emission reduction goals. Appropriate emission reductions technology could have a significant impact on pollutant releases at the lowest economic cost to the facility.

Alnahdi et al. (2019) evaluated technology that could reduce pollutant emissions from oil refineries. Specifically, the study examined the use of mathematical programming to help identify the most effective pollution controls technology for NO_X, SO_X, and CO_2 emissions from refineries. The program considered emissions under multiple different scenarios and provided insight into the most appropriate control strategy for a specific refinery (as well as cost), thereby helping refineries reach emission reduction goals. The authors selected an industrial scale oil refinery in North Toronto, Canada, to apply this methodology. The analysis demonstrated that by implementing the best emissions control technology for the given geographical and meteorological conditions of that refinery, emissions of NO_X, SO_X, and CO_2 could be reduced by 60% for only a 10.7% increase in costs (Alnahdi et al. 2019).

Recommendation: Emission controls for downstream sources should be considered and include (1) the use of kitchen ventilation systems to reduce exposures from natural gas appliance use; and (2) the implementation of a high time resolution monitoring network at refineries to allow for more targeted monitoring and source apportionment.

Mullen et al. (2015) evaluated the impact of natural gas appliance use in 352 Californian homes and demonstrated that NO_x, NO_2 and highest 1–hr carbon monoxide (CO) concentrations occurred in homes that used natural gas for cooking (compared to electric cooking); however, these associations were reduced when scaled to an average–size home. The use of kitchen ventilation systems could lower concentrations of NO_x, NO_2, and CO (Mullen et al. 2016). Du and Turner (2015) evaluated $PM_{2.5}$ lanthanoid concentrations near a petroleum refinery and highlighted the need for a network of high time resolution monitors to allow for more complete monitoring of emissions from facilities and enable more targeted source apportionment.

5.0   Results: Methane and HDAP studies (2015–2020)

While the majority of studies identified in this review focus on methane or HDAP emissions only, there is a subset of studies that evaluate both methane and HDAP releases from the oil and gas sector. These studies are particularly important, as they provide useful insight into mitigation measures that could potentially co-reduce methane and HDAP emissions from the oil and gas sector, thus providing a potential co-benefit of reducing the climatic impact of methane while protecting the health of nearby populations from harmful pollutant releases. Section 5 evaluates the key findings and conclusions presented in these studies and identifies emission control strategies aimed at reducing both methane and HDAP emissions from the oil and gas sector.

We identified 29 studies published from 2015–2020 that evaluated both methane and at least one health-damaging air pollutant (HDAP). As discussed in Section 4, an HDAP refers to criteria air pollutants, hazardous air pollutants, and other non-methane volatile organic compounds (VOCs) released by the oil and gas sector. The majority of studies focused on air pollutant emissions from upstream oil and gas sources. Only four studies focused on air pollutant emissions from midstream processes, while four studies evaluated emissions from downstream processes. Two of these downstream studies also evaluate emissions from upstream sources. We also identified six additional peer-reviewed studies focused on emissions from the oil and gas sector, published in 2021, that were out of the scope of the review period. However, due to the relevancy of these findings, we include a discussion of the results and recommendations here.

5.1.   Review of upstream methane and HDAP studies

We identified 23 studies published from 2015–2020 that evaluate emissions of both methane and HDAPs from upstream oil and gas processes. Studies were conducted in the Marcellus Shale region in Pennsylvania and/or West Virginia (Goetz et al. 2015, 2017; McCawley, 2015; Yuan et al. 2015); Utah (Koss et al. 2015; Oltmans et al. 2016; Ahmadov et al. 2015); Texas (Allen, 2016; Marrero et al. 2016; Roest and Schade, 2017); Colorado (Brantley et al. 2015; Hecobian et al. 2019; Milford, 2015); California (Cui et al. 2015; Wunch et al. 2016); North Dakota (Gvakharia et al. 2017; Weyant et al. 2016); Canada (Baillie et al. 2019; Hurry et al. 2016; O'Connell et al. 2019); the United States (Johnson et al. 2018); the east coast of the United States (Plant et al. 2019); and multiple states in the southwestern United States (i.e., Colorado, northern New Mexico, Utah, and Wyoming) (Nsanzineza et al. 2019).

We also identified five additional peer-reviewed studies, published in 2021, that evaluate methane and HDAP emissions from upstream oil and gas processes (Buonocore et al. 2021; Holliman and Schade 2021; Johnston et al. 2021; Okorn et al. 2021; Orak et al. 2021). Studies were conducted in California, Texas, West Virginia, and the United States.

PSE

5.1.1. Upstream methane and HDAPs: findings and conclusions

Peer-reviewed literature published from 2015–2020

In the Marcellus Shale region of Pennsylvania, Goetz et al. (2015) used real-time measurements coupled with tracer release ratio methods to determine emission rates from various oil and gas sources. The authors found evidence of elevated levels of methane and ethane from oil- and gas-associated combustion sources, in addition to elevated levels of carbon monoxide and nitrogen oxides. Elevated levels of methanol, a hazardous air pollutant, were observed at a compressor station and near the well pad, while benzene and toluene were not detected (Goetz et al. 2015). Compressor stations and "transient sites" (e.g., drilling and well completions) were the largest emitters of methane, carbon dioxide, carbon monoxide, and nitrogen oxides, followed by producing well sites. This is consistent with findings from Goetz et al. (2017), which identified oil and gas wells as significant sources of methane, ethane, and carbon monoxide — but not major contributors of toluene and benzene — using two ground-based mobile measurement campaigns. The low levels of toluene and benzene may be due to the presence of dry-gas wells, which is composed of mainly methane, in the region, as opposed to wet-gas, which is composed of methane and other light alkanes (Goetz et al. 2017). The studies found ethane to methane enhancement ratios consistent with ratios similar to dry gas, supporting this hypothesis. Another Pennsylvania study, using measurements from two aircraft campaigns, found methane to benzene enhancement ratios to be consistent with emissions signatures associated with upstream oil and gas development (Yuan et al. 2015). The authors note that ~10% of facilities (e.g., gas processing facilities, compressor stations) accounted for ~40% of methane emissions observed in the monitored regions, highlighting the potential presence of super-emitting facilities that require further mitigation.

In Utah, methane emissions from a natural gas field were significantly correlated with levels of volatile organic compounds (VOCs), including ethane, propane, n-butane, i-pentane, n-pentane, hexane, benzene, heptane, toluene, octane, and xylenes (Oltmans et al. 2016). These findings were derived from measurements taken from an aircraft sampling campaign and from discrete air samples collected with the National Oceanic and Atmospheric Administration's (NOAA) Global Monitoring Division (GMD) glass flask packages (Oltmans et al. 2016). Emissions were traced to several upstream sources, including well sites, gathering pipelines, compressor stations, and two large processing plants. Consistency in the distribution of these non-methane VOCs with methane distributions suggests they are co-emitted with methane (Oltmans et al. 2016). A 2015 study in the Uintah Basin, Utah, found pollutant emission ratios derived from proton-transfer-reaction mass-spectrometry (PTR-MS) to be consistent with contributions of emissions from oil and gas producing wells (Koss et al. 2015). In addition, the methane emission rate, extrapolated from the emission rate for benzene, was consistent with an independent evaluation of methane emissions using aircraft measurements (top-down) from 2012.

Another Utah-based study evaluated emissions from oil and natural gas operations using a top-down (i.e., aircraft measurements) and bottom-up (i.e., emissions inventory) approach (Ahmadov et al. 2015). The authors found the bottom-up approach to underestimate methane and other VOCs (e.g., BTEX) and overestimate emissions of nitrogen oxides when compared to the top-down approach; although the authors note that the top-down approach still underestimates methane levels by approximately 40% (Ahmadov et al. 2015). Similarly, the top-down approach was better able to capture the observed high ozone concentrations in the Uinta Basin, which is not surprising, as ozone is not directly emitted and would therefore not be captured by the bottom-up inventory. The top-down approach was also better able to capture the temporal variability of ethane, toluene, xylene, and secondary species including ozone, peroxyacyl nitrates (PAN), and acetaldehyde, compared to the bottom-up approach. High emissions of non-methane VOCs compared to emissions of nitrogen oxides suggest oil and natural gas operations are a significant source of ozone in the region. These findings may be limited to Western states, however, as states in the West are likely VOC limited, whereas states in the east are likely nitrogen oxides (NO_x) limited (Ahmadov et al. 2015).

Marrero et al. (2016) collected whole air samples upwind and downwind from a number of upstream oil and gas sources in Texas. Concurrent with whole air sampling, the authors measured methane fluxes using the mobile flux plane (MFP) method (top-down), and derived emission factors using a spatially refined methane emissions inventory. Using these methods, the highest hexane and m- and p-xylene mixing ratios were observed downwind of well pads with compressors, where methane leak rates were highest, while the highest toluene and benzene mixing ratios were found near oil-producing wells (Marrero et al. (2016). Estimates of hexane, benzene, and toluene in Texas were consistent with estimates in Colorado and Utah, suggesting that there may be some similarity in emissions profiles from oil and natural gas development across geographic regions (Marrero et al. 2016).

Findings from a Texas-based study suggest that a small number of upstream oil and gas sources are responsible for a significant portion of methane and VOC emissions ("super-emitters") (Allen, 2016). While it is still uncertain why specific sites become super-emitters, the evidence suggests that differences in operational practices at well sites, as well as operational failures of high-emitting oil and gas components like pneumatic controllers and compressors, are potential factors (Allen, 2016).

A study conducted by Hecobian et al. (2019), using a tracer ratio method, found variations in measured emission rates of air toxics and VOCs at the various stages of production in the Denver-Julesburg and Piceance Basins in Colorado. Emission rates differed depending on the basin and phase of production, with flowback operations accounting for the highest levels of light and heavy alkane (e.g., n-hexane, n-heptane) emissions among all the sites sampled. Drilling and production activities produced elevated levels of light alkane emissions (e.g., ethane, propane, n-butane), but at much lower levels than during hydraulic fracturing and flowback operations. When the duration of operations is considered, however, drilling and production activities could still present a significant risk, because drilling and production

PSE

activities (including conventional methods) are continuous (e.g., 8 hours or more per day) and generally fixed in one location (i.e., longer exposure duration), whereas stimulation treatments and flowback operations occur over shorter intervals (e.g., 5 hours of operation per day) and move from location to location (Hecobian et al. 2019).

These findings are consistent with findings from (Milford, 2015), which identified diesel-powered drill rigs and natural-gas powered compressor stations as the largest contributors to emissions of nitrogen oxides in Colorado. In addition to nitrogen oxides, large reciprocating natural-gas powered compressors are significant sources of VOCs, carbon monoxide, particulate matter, carbon dioxide, and methane; and diesel fuel-powered drill rigs are significant sources of particulate matter, VOCs, and sulfur dioxide (Milford, 2015). The largest VOC emissions were attributed to flashing losses[1] from crude oil and condensate storage tanks, fugitive emissions from leaks in valves, fittings and other equipment, venting of hydrocarbons from completions and blowdowns, venting from glycol dehydration units, and natural gas-driven pneumatic devices (Milford, 2015).

McCawley (2015) evaluated releases from drill sites in West Virginia using tapered element oscillating microbalance (TEOM) 24-hr dust samples and found $PM_{2.5}$ and PM_{10} (particulate matter) concentrations to not exceed 24-hr national ambient air quality standards (NAAQS). Average concentrations of ammonia, nitrogen oxides, ozone, and sulfur dioxide "did not indicate a concern for ambient or occupational exposures," although the author did not offer direct comparison to standards for these pollutants. Nsanzieza et al. (2019) evaluated future emissions from both downstream electricity systems and upstream oil production in Colorado, northern New Mexico, Utah, and Wyoming using the MARKAL (MARKet ALlocation) least-cost planning model with the U.S. EPA's nine-region energy system database and the Comprehensive Air Quality Model with Extensions (CAMx). Findings from Nsanzineza et al. (2019) illustrate that the implementation of greenhouse gas (GHG) emission fees and regulations aimed to reduce GHG emissions would also reduce methane emissions from both upstream and downstream sources by 42%, nitrogen oxide emissions by 48%, and VOC emissions by 42% by 2030.

Findings from Roest and Schade (2017) in Texas confirmed methane and non-methane hydrocarbons (NMHCs) are indeed co-emitted from liquid storage tanks, with alkane mixing ratios increasing in the Eagle Ford Shale region in tandem with increasing oil and natural gas production rates. These findings were derived using a variety of methods, including the use of atmospheric enhancement of alkanes from the Texas Commission on Environmental Quality's (TCEQ) VOC monitors coupled with back trajectory and dispersion modeling, as well as ethane-to-methane ratios in gaseous emissions. The largest fraction of methane emissions identified

---

[1] Flashing losses can occur when storage tank liquids, such as condensate or oil, experience a decrease in pressure or an increase in temperature.

PSE

in Brantley et al. (2015) were found in tank samples collected from a dehydrator (64.4%), which is a device used to remove excess water vapor from natural gas.

Unconventional well development in the United States also involves intensive use of service trucks, horizontal drilling rigs, and hydraulic fracturing pumps, all of which are typically diesel fuel-powered (Johnson et al. 2018). Using a variety of bottom-up approaches (e.g., emission inventories, activity information), engines used during hydraulic fracturing activities were found to produce the largest amount of nitrogen oxides emissions, drilling rigs were found to produce large amounts of carbon monoxide emissions, and diesel-powered trucks produced the largest total hydrocarbon emissions of all phases evaluated (Johnson et al. 2018).

Findings from studies conducted in Canada are consistent with findings in the United States. A study conducted in the Weyburn oil field in Saskatchewan, Canada estimated methane-carbon dioxide and methane-hydrogen sulfide mixing ratios and found peak levels of methane and hydrogen sulfide (H_2S) to reach 2.2 ppm and 15 ppb, respectively (Hurry et al. 2016). The authors also evaluated local emissions from specific oil and gas source types operating in the region. At 1.3 km (~4,260 ft) downwind from a drill rig, the authors found concentrations of methane and H_2S to be 2 ppm and 1.3 ppb, respectively. Similarly, at 1.4 km (~4,590 ft) from a service rig, concentrations of methane and H_2S peaked at 2.3 ppm and 1.7 ppb, respectively, with mixing ratios strongly indicating a fossil fuel combustion source.

Another study evaluated emissions from multiple oil and gas-producing sites in Alberta, Canada (O'Connell et al. 2019). The authors found publicly available emissions inventories to significantly underestimate the true pollutant releases from upstream oil and gas sites active in the Alberta region, consistent with findings in the United States. The authors also found relatively low incidence of hydrocarbon plumes in the Peace River oil and gas region to be most likely due to federally mandated emission control requirements focused on reducing venting and flaring emissions in the region (O'Connell et al. 2019). Regulatory requirements mandated in the region include tank vapor recovery systems, flare requirements, and infrastructure improvements to new and existing gathering systems (e.g., additional compression for injection into the gathering system, capacity expansion). Even with these emission control requirements, however, the Peace River region was found to have the highest H_2S concentrations of all the regions sampled, with concentrations exceeding Canada's ambient air quality thresholds for H_2S (one-hour average should not exceed 10 ppbv; 24-hour average not to exceed 3 ppbv); this exceedance is most likely due to the sulfurous nature of the underlying deposits (O'Connell et al. 2019).

These findings are supported by another study in Saskatchewan, Canada, which estimated peak H_2S concentrations to be 132 ppb from all campaigns, and found the Weyburn-Midale formation to have a H_2S content seven times greater than that of the Bakken shale region, a formation that overlies regions in Canada and the United States, including Montana and North Dakota (Baillie et al. 2019). In regions where unconventional and conventional oil and gas production were co-located, differences in emissions intensities were most likely due to the

PSE

indirect mitigation of methane through the implementation of H₂S-specific management practices at conventional sites, highlighting the potential co-benefits that health protective mitigation measures can have on methane emissions from the oil and gas sector.

There is substantial evidence that methane emissions from many oil and gas sources are indeed co-emitted, and in some cases, significantly correlated with emissions of HDAPs and other non-methane VOCs. Fugitive (unintentional) releases of methane, and associated non-methane VOCs, can occur from component and equipment leaks, including from valves, screwed connections, flanges, open-ended lines, and pump seals (ExxonMobil, 2021; US EPA, 2016). Direct venting of emissions can also occur during well stimulation treatments, specifically during flowback operations and manual liquids unloadings. In some cases, the intended function of a component results in the intentional release of methane emissions, such as is the case with natural gas-powered pneumatic devices, which directly release or "bleed" gas (ExxonMobil, 2021). VOCs and HDAPs are often co-emitted with methane releases from pneumatic controllers and pumps (US EPA, 2016). Methane is the largest component of vapor releases from storage vessels, but these vapor releases may also include releases of n-hexane, alkanes (e.g., ethane, butane, propane) and HDAPs (e.g., benzene, toluene, ethylbenzene and xylenes) (US EPA, 2016).

Additional sources of methane and associated non-methane VOCs from upstream oil and gas development include releases from incomplete combustion (e.g., flaring), centrifugal and reciprocating compressors, and transmission pipeline blowdowns (ExxonMobil, 2021; US EPA, 2016). Incomplete combustion during flaring events was found to be a significant source of black carbon and methane emissions, as evidenced in the Bakken region of North Dakota (Gvakharia et al. 2017; Weyant et al. 2016). Combustion and incomplete combustion (e.g., flaring) of organic pollutants also produces secondary pollutants including nitrogen oxides, carbon monoxide, sulfur dioxide and particulates (US EPA, 2016). A summary of the main findings by general upstream source type is provided in Table 5.1.

Table 5.1. Summary of main findings from peer-reviewed literature published from 2015–2020 related to methane and HDAP emissions from upstream oil and gas activity by general source type.

Upstream source(s) evaluated	Main findings related to methane and HDAPs
Compressor stations near upstream oil and gas sites	- Elevated levels of methanol, a hazardous air pollutant, were observed at a compressor station (Goetz et al. 2015). - ~10% of facilities (e.g., gas processing facilities, compressor stations) accounted for ~40% of methane emissions observed, highlighting the potential presence of super-emitting facilities that require further mitigation (Yuan et al. 2015). - Compressor stations were found to be the largest emitters of methane, carbon dioxide, carbon monoxide, and nitrogen oxides (Goetz et al. 2015). - Methane emissions from several upstream sources, including compressor stations, were significantly correlated with levels of ethane, propane, n-butane, i-pentane, n-pentane, hexane, benzene, heptane, toluene, octane, and xylenes (Oltmans et al. 2016). - Highest hexane and m- and p-xylene mixing ratios were observed downwind of well pads with compressors, where methane leak rates were highest (Marrero et al. 2016). - Natural-gas powered compressor stations are one of the largest contributors to emissions of nitrogen oxides in Colorado (Milford, 2015). - In addition to nitrogen oxides, large reciprocating natural-gas powered compressors are significant sources of VOCs, carbon monoxide, particulate matter, carbon dioxide, and methane (Milford, 2015).
Hydraulic fracturing and flowback operations	- Flowback operations were found to account for the highest levels of light and heavy alkanes (e.g., n-hexane, n-heptane) among all the sites sampled (Hecobian et al. 2019). - Drilling and production activities produced elevated levels of light alkanes (e.g., ethane, propane, n-butane), but at much lower levels than during hydraulic fracturing and flowback operations (Hecobian et al. 2019). - Diesel-powered engines used during hydraulic fracturing activities were found to produce the largest amount of nitrogen oxides emissions (Johnson et al. 2018).

PSE

Upstream source(s) evaluated	Main findings related to methane and HDAPs
Drilling and well completions (e.g., "transient" sites)	- "Transient sites" (e.g., drilling and well completions) were found to be the largest emitters of methane, carbon dioxide, carbon monoxide, and nitrogen oxides, followed by producing well sites (Goetz et al. 2015). - Drilling and production activities produced elevated levels of light alkanes (e.g., ethane, propane, n-butane), but at much lower levels than during hydraulic fracturing and flowback operations (Hecobian et al. 2019). - Diesel fuel-powered drill rigs are significant sources of particulate matter, VOCs, and sulfur dioxide (Milford, 2015). - Drill rigs are one of the largest contributors to emissions of nitrogen oxides in Colorado (Milford, 2015). - Drilling rigs were found to produce large amounts of carbon monoxide emissions (Johnson et al. 2018). - $PM_{2.5}$ and PM_{10} concentrations at drill sites did not exceed 24-hr NAAQS. Average concentrations of ammonia, nitrogen oxides, ozone, sulfur dioxide "did not indicate a concern for ambient or occupational exposures", though the author did not offer direct comparison to standards for these pollutants (McCawley, 2015). - At 1.3 km (~4,260 ft) downwind from a drill rig, Hurry et al. (2016) found concentrations of methane and H_2S to be 2 ppm and 1.3 ppb, respectively. - At 1.4 km (~4,590 ft) from a service rig, concentrations of methane and H2S peaked at 2.3 ppm and 1.7 ppb, respectively, with mixing ratios strongly indicating a fossil fuel combustion source (Hurry et al. 2016)
Oil and gas production	- "Dry-gas" wells were identified as significant sources of methane, ethane, and carbon monoxide, but not major contributors of toluene and benzene (Goetz et al. 2017). - Pollutant emission ratios were found to be consistent with contributions of emissions from oil and gas producing wells (Koss et al. 2015). - Drilling and production activities produced elevated levels of light alkanes (e.g., ethane, propane, n-butane), but at much lower levels than during hydraulic fracturing and flowback operations (Hecobian et al. 2019). - The highest toluene and benzene mixing ratios were found near oil-producing wells (Marrero et al. 2016). - The largest VOC emissions were attributed to "flashing losses from crude oil and condensate storage tanks, fugitive emissions from leaks in valves, fittings and other equipment, venting of hydrocarbons from completions and blowdowns, venting from glycol dehydration units and natural gas-driven pneumatic devices" (Milford, 2015). - In regions where unconventional and conventional oil and gas production were co-located, differences in emissions intensities were due to the indirect mitigation of methane through the implementation of H2S-specific management practices at conventional sites, highlighting the potential co-benefits that health protective mitigation measures can have on methane emissions from the oil and gas sector (Baillie et al. 2019).

PSE

Upstream source(s) evaluated	Main findings related to methane and HDAPs
Other sources of combustion (e.g., flaring, diesel-powered trucks)	- Elevated levels of methane and ethane from oil and gas-associated combustion sources were observed, in addition to elevated levels of carbon monoxide and nitrogen oxides (Goetz et al. 2015). - Diesel-powered trucks produced the largest total hydrocarbon emissions of all phases evaluated (Johnson et al. 2018). - Incomplete combustion during flaring events was found to be a significant source of black carbon and methane emissions (Gvakharia et al. 2017; Weyant et al. 2016). - Combustion and incomplete combustion (e.g., flaring) of organic pollutants also produces secondary pollutants including nitrogen oxides, carbon monoxide, sulfur dioxide and particulates (US EPA, 2016).
Liquid storage tanks (e.g., condensate, produced water)	- Fugitive leaks from liquid storage tanks (e.g., condensate, produced water) are an important source of VOC and hazardous air pollutant (HAP) emissions (Brantley et al. 2015). - Fugitive VOC emissions from liquid storage tanks are highly variable (Brantley et al. 2015). - Thief hatches on condensate tanks are the most frequently reported source of VOC leaks, even on tanks with emission control technology implemented (Brantley et al. 2015). - The largest fraction of methane emissions was found in samples taken at the dehydrator (Brantley et al. 2015). - Samples from produced water tanks, emissions of which were uncontrolled at the time of sampling, were found to have the largest fraction of non-methane VOCs and HAP emissions of all the components tested in the study (Brantley et al. 2015). - Liquid storage tanks are a likely major contributor of the non-methane hydrocarbon (NMHC) enhancements observed in the region (Roest and Schade, 2017). - Methane and NMHCs are indeed co-emitted from liquid storage tanks (Roest and Schade, 2017). - Pollutants co-emitted with methane include n-butane, isobutane, propane, and ethane emissions (Roest and Schade, 2017).

PSE

Peer-reviewed literature published in 2021

We identified five additional peer-reviewed studies, published in 2021, that evaluate methane and HDAP emissions from upstream oil and gas processes.

Two studies were conducted near urban oil and gas production sites in Los Angeles, California (Johnston et al. 2021; Okorn et al. 2021). Oil and gas production facilities, such as AllenCo in Los Angeles, have periods of active production as well as idle periods, emissions of which greatly differ depending on the phase. Johnston et al. (2021) relied upon ambient air monitoring, 5-minute trigger samples, and 24-hr passive canister samples collected by the South Coast Air Quality Management District (SCAQMD) to determine the methane and HDAP concentrations adjacent to an oil and gas production site. Average concentrations of methane, non-methane hydrocarbons (NMHCs), benzene, toluene, ethylbenzene, xylenes (BTEX), styrene, n-hexane, n-pentane, ethane, and propane decreased once production activities idled (Johnston et al. 2021). Specifically, the authors observed a 28%, 32%, and 69% decrease in toluene, benzene, and n-hexane concentrations, respectively, after production at the site idled. Findings from this study suggest that natural gas drilling during the active phase contributes 23.7% of the total VOCs measured, while the idle period only contributes 0.6% (Johnston et al. 2021). While concentrations at the fence line of the facility (i.e., facility boundary) were below state acute reference exposure levels, they were higher than background concentrations taken by the California Air Resources Board (Annual Toxics Summaries by Monitoring Site, 2013) and SCAQMD (Final Multiple Air Toxics Exposure Study (MATES) IV, 2015) for the area, suggesting a local emissions source.

From 2016 to 2019, Okorn et al. (2021) deployed low-cost air sensors that measure methane, NMHCs, carbon dioxide, and carbon monoxide in three Los Angeles communities located near oil and natural gas facilities, with active operations occurring at sites 1 and 3 and no production occurring at site 2 (well activity ceased in 2013). All three facilities are located within 3 km (~1.86 miles) of each other and draw from the Las Cienegas oil field (Okorn et al. 2021).[2] Results from this study demonstrate that methane levels varied based on proximity to an oil and natural gas facility (Okorn et al. 2021). Specifically, monitoring results show that methane levels are higher within 500 m (1,640 ft) of the three oil and natural gas facilities — and near a natural gas pipeline — compared to concentrations farther away. The authors suggest this trend is likely a result of proximity to emission sources (Okorn et al. 2021). Significant methane concentrations were also found at Site 2, where wells have been idle since 2013, indicating that fugitive emissions

---

[2] At each site, anywhere from four to 11 devices were installed within 500 m (1,640 ft) of the facility. Two to 11 devices were deployed outside this 500 m radius: at a distance of 800 m to 8 km (~2,624 ft to 26,247 ft) for Site 1; 4 km away (13,123 ft) for Site 2; and 800 m to 1 km (~2,624 ft to 3,280 ft) for Site 3. The devices deployed outside the 500 m radius were used to estimate emissions from major roadways and to act as controls.

PSE

of methane may still be released by oil and gas well sites long after active operations have stopped.

Unlike methane, which shows a clear and significant association with proximity to upstream oil and gas activity, NMHCs results were less straightforward. NMHCs concentrations within the 500 m radius were similar to concentrations found outside this radius (control sites and near major freeways), with modest differences seen at Sites 1 and 2. NMHCs levels were significantly associated with proximity to freeways for Sites 1 and 3, suggesting that traffic is an important source of NMHCs in these communities. However, NMHCs monitoring results show that short-term, episodic emissions spikes tended to be higher at locations near an oil and natural gas facility compared to variances seen outside of the 500 m radius, suggesting these events may be associated with specific oil and gas activities conducted on-site (Okorn et al. 2021). These findings are consistent with Johnston et al. (2021), which found emissions to increase or decrease depending on the phase of production and whether production was active or idle.

Orak et al. (2021) evaluated emissions from an oil and gas production site located in the Marcellus Shale region, West Virginia. Fence line monitoring 900 m (~2,950 ft) away from activity revealed an increase in NO_x and NO during the fracturing phase and a significant increase in ethane and methane concentrations during the flowback phase (Orak et al. 2021). Oil and gas production over several phases was found to contribute to "engine emissions" (e.g., combustion and incomplete combustion), with peak contributions observed during the drill out phase. Peak contributions to "engine emissions" were also observed during production, most likely due to the presence of maintenance vehicles and other short-lived sources of combustion on-site. Similarly, emissions from horizontal drilling, including the end of the horizontal drilling phase, significantly contribute to emissions from "natural gas" (Orak et al. 2021).

Findings from a top-down approach used in Holliman and Schade (2021) suggest that emissions from oil and gas operations in the Eagle Ford Shale region, Texas, exceed their permitted allowance. The authors compared measured emission estimates from the Texas Commission on Environmental Quality's (TCEQ) air quality monitoring stations to two major source types: (1) oil and gas activity in the region (quantified using point source inventories, non-permitted (e.g., "upset") emissions reports, and air permits issued by the TCEQ); and, (2) traffic-related emissions (assessed in a previous study) (Holliman and Schade, 2021). Results from this assessment indicated that the permit-based emissions estimates accounted for just 86% of the propane, butane, pentane, and hexane (C_3-C_6 hydrocarbons) emissions (median) estimated at the monitor. The authors conclude,

"Since the measurement-based emissions encompass a smaller section of the shale than the calculated maximum permitted emissions, this strongly suggests that the actual emissions from oil and gas operations in this part of the Eagle Ford exceeded their permitted allowance" (Holliman and Schade, 2021).

PSE

The authors note this discrepancy may be due to emissions from abandoned wells, as well as from excessive venting of raw gas from unlit flares.

Buonocore et al. (2021) evaluated the human health impacts from air pollution emissions released during oil and gas production activities in the United States. In 2016, health impacts from air contaminants released by the oil and gas sector contributed to approximately 7,500 excess deaths, 2,200 new cases of childhood asthma, and 410,000 new cases of asthma exacerbation (Buonocore et al. 2021). Of these, 2,100 excess deaths were due to $PM_{2.5}$, 2,600 were due to ozone, and 2,800 were due to nitrogen dioxide (NO_2). This additional pollution from the oil and gas sector also resulted in multiple exceedances of daily $PM_{2.5}$ (29 instances of 12×12 km grid cells, ~7.5×7.5-mile grid) and annual ozone (634 exceedances) National Ambient Air Quality Standards (NAAQS).

Buonocore et al. (2021) also evaluated the potential health and air quality benefits of various methane emissions reduction policies, if implemented in 2028. Of the five scenarios assessed, the largest air quality and human health benefits were found in the scenario in which stronger federal and state policies were aimed at reducing methane emissions from the oil and gas sector (when compared to the business-as-usual scenario), followed by the scenario in which stronger state policies were implemented. The "weakened federal rules" scenario was found to result in increased emissions and thus, an increased health burden; the health and air quality impacts under the federal base scenario, however, were found to be higher, highlighting the importance of state-level oil and gas methane policies (Buonocore et al. 2021). The authors note that the air quality and health co-benefits from stronger oil- and gas-related state and federal methane reduction policies are likely to vary based on geography, as is the case with policies aimed at reducing combustion-related carbon dioxide emissions. For these reasons, the authors recommend "the health co-pollutant costs of methane... be separately assessed alongside impacts of methane itself" (Buonocore et al. 2021). Findings from this study have important policy implications, as is emphasized by the authors. The authors conclude,

"Our results here provide evidence of variability in the contributions of different elements of O&G production. Unlike fossil fuel combustion, where GHGs and air pollutants are emitted from the same process, our results indicate that different elements of O&G production have differing contributions to GHG emissions and air pollution. Therefore, unlike fossil fuel combustion where many CO_2 reduction policies are likely to have proportionate reductions in air pollution and proportionate health benefits, policies and strategies which reduce methane emissions may not necessarily have proportionate health benefits."

This is evidenced by leak detection and repair (LDAR) programs aimed at reducing methane emissions from the oil and gas industry through increased monitoring and timely mitigation strategies (Buonocore et al. 2021). These LDAR programs often result in very large reductions in methane emissions — along with modest human health benefits — as a result of reductions in non-methane VOC emissions and associated formation of secondary compounds. Therefore, the implementation of other control measures should be considered in tandem with programs

PSE

aimed at detecting and mitigating methane emissions from leaking oil and gas infrastructure. For example, the implementation of control measures aimed at controlling NO_x emissions from equipment, such as flares, compressor stations, and other high-emitting sources, may have significantly higher health benefits when compared to the overall health benefits from methane-focused LDAR programs (Buonocore et al. 2021).

5.1.2. Upstream methane and HDAPs: recommendations

There is substantial evidence that methane emissions from upstream oil and gas sources are indeed co-emitted, and in some cases, significantly correlated with emissions of health damaging air pollutants (HDAPs). Below we identify emission control and policy recommendations aimed at reducing both methane and HDAP emissions from the upstream oil and gas sector.

Super-emitters

Recommendation: Efforts to identify super-emitting facilities are necessary to achieve further emission reductions and can be facilitated through the use of new technologies such as infrared cameras to detect emission plumes, or vehicle-mounted methane sensors to rapidly detect super-emitting sites and mitigate their emissions.

A small number of upstream oil and gas sites and equipment are responsible for a large (and disproportionate) fraction of methane and HDAP emissions. While uncertainties exist, differences in operational practices at well sites and operational failures of high-emitting oil and gas components are potential factors.

Recommendation: Emission inventories should consider the utilization of separate emission factors and activity/equipment counts for super-emitters. As emission inventories are updated to properly account for super-emitters, studies that rely on top-down approaches should be considered in tandem with bottom-up approaches, as the use of both are better able to capture emissions from these source types.

Emission inventories that rely on single emission factors for source categories in the oil and gas supply chain are inadequate for capturing the true emissions from super-emitting facilities.

Recommendation: Emission control measures aimed at reducing methane, black carbon, and the secondary formation of nitrogen oxides, carbon monoxide, sulfur dioxide, and particulates, such as more accurate monitoring of steam- and/or air-assisted flares, should be implemented in an effort to ensure that a subset of flares do not become "super-emitting."

A small number of high-emitting flares used during production are responsible for a large fraction of flaring emissions, due to incomplete combustion, which can occur in large-capacity flares with low-flow rates that fail to achieve desired combustion efficiencies. Super-emitting

PSE

flares can also be created if too much air or steam is added to reduce the formation of smoke (i.e., over-assisted). Incomplete combustion during flaring events is a significant source of black carbon and methane emissions. Combustion and incomplete combustion (e.g., flaring) of organic pollutants also produces secondary pollutants, including nitrogen oxides, carbon monoxide, sulfur dioxide, and particulates.

Compressor stations and pneumatic controllers

Recommendation: Methane and HDAP emissions from pneumatic controllers and compressors can be further reduced and/or eliminated by: (1) equipment repair or replacement; (2) electrification of natural gas-powered compressors used during production (existing and new) at compressor stations that use natural gas-powered compressor engines; and (3) implementing additional emission control measures, such as the installation of NO_x emission controls, to reduce emissions from compressor station process equipment.

To ensure proper enforcement, these emission control measures should be required as a part of the permit certification process. Permitting requirements should include: (1) restrictions on siting near occupied buildings and other sensitive receptors; (2) the use of low-NO_x or electrified compressor engines; and (3) emission rate limitations for NO_x, carbon monoxide, and VOCs for both compressor engines and all associated process equipment. Prior to certification, permits for compressors should also specify monitoring and reporting requirements to ensure compliance with these emission control measures. The New Mexico Environment Department has implemented a multitude of permitting requirements that include these types of emission control measures, providing a precedent for other oil and gas-producing states.

Compressor stations that use natural gas-powered compressor engines run continuously to transport gas from well sites to pipelines. Due to their continuous operation and the use of process equipment such as glycol dehydrators and in-line heaters, compressor stations are a large source of methane and HDAP emissions including carbon monoxide, nitrogen oxides, VOCs, particulate matter, benzene, toluene, xylenes, and several types of alkanes. Compressor stations with electric-powered compressor engines are still a major source of pollutant emissions due to the use of process equipment such as glycol dehydrators, separators, in-line heaters, liquids tanks, etc. Therefore, reductions in both compressor-engines and associated process equipment are necessary to adequately reduce emissions from compressor stations.

High emissions from pneumatic controllers and compressors throughout the United States have also been attributed to devices not operating as designed, suggesting that equipment replacement and repair could be one potential avenue for reducing emissions from these sources. Findings suggest there are multiple avenues to further mitigate emissions of VOCs and HDAPs from compressor stations. These mitigation measures would also help in co-reducing methane emissions from these sources.

Liquid storage tanks

Recommendation: Agencies with jurisdiction should implement additional control and safety measures, such as a regulatory setback distance requirement from sensitive populations, to provide a margin of safety for when vapor control technologies on storage tanks fail or become ineffective over time. Agencies should also implement additional retrofit and replacement requirements for thief hatches and dehydrators on condensate tanks and for produced water tanks, both new and existing.

Liquid storage tanks (e.g., condensate, produced water) are an important source of methane and HDAPs, and VOCs and HDAPs can be emitted by condensate tanks even in the presence of vapor control measures. Thief hatches, also referred to as gauge hatches, are a frequent location of leaks from condensate tanks, even with vapor control measures in place. Dehydrators are another large source of methane emissions from liquid storage tanks. Finally, produced water tanks are a significant source of VOC and HDAP emissions when left uncontrolled.

Hydraulic fracturing and flowback operations

Recommendation: Technological advances, such as the use of closed loop systems to capture and transport flowback fluids offsite, have recently been used and are an effective method for further reducing emissions from hydraulic fracturing and flowback operations. Diesel-powered engines used during hydraulic fracturing activities should be equipped with diesel oxidation catalysts, which have been used by industry as an effective method for reducing carbon monoxide from dual fuel retrofitted engines. Engine optimization and sophisticated system integration has also been shown to be a viable alternative, in addition to the use of heavily integrated high-pressure direct injection "HPDI" engines, which use dual-fuel combustion and offer reduced methane emissions compared to retrofit kits.

While methane and HDAP emissions are released during all stages of oil and gas production, specific phases, activities, and equipment account for a larger portion of emissions than others. For example, hydraulic fracturing and flowback operations, as well as drilling and well completions, are associated with higher emissions when compared to emissions from well production. There are multiple avenues to further mitigate emissions of VOCs and HDAPs from these source types, which would also co-reduce methane releases in the process.

Flowback operations account for high levels of light and heavy alkanes (e.g., n-hexane, n-heptane). Diesel-powered engines used during hydraulic fracturing activities produce large amounts of nitrogen oxides emissions. This is due to the high load operation and amount of total energy consumed, in addition to less-stringent off-road emission standards. Current methods to control emissions from hydraulic fracturing engines include retrofitting existing diesel engines with dual fuel engines. While this option reduces some air pollutant emissions, it can also result in an increase of pollutants such as carbon monoxide and non-methane

PSE

hydrocarbons. Therefore, other control measures should be implemented in place of dual-fuel retrofits to adequately reduce pollutant emissions.

Drilling and well completions

Recommendation: Replacement of diesel-powered drill rigs with natural gas-powered engines and three-way catalysts would significantly reduce emissions during this phase; however, these reductions could be negated if the three-way catalysts do not operate correctly. Therefore, in addition to replacing diesel-powered drilling rigs with natural gas, operators should also implement catalyst maintenance procedures to maximize emissions benefits. Additional control measures, such as the use of Tier 2 diesel engines or higher (i.e., Tier 3 or 4), would also significantly reduce emissions, although not to the extent that natural gas engines would provide.

Drilling and well completion activities are a large source of methane, carbon dioxide, carbon monoxide, and nitrogen oxides. This is partly due to the use of diesel-powered drill rigs, which are a significant source of particulate matter, VOCs, nitrogen oxides, carbon monoxide and sulfur dioxide.

Oil and gas production

Recommendation: Regulators aiming to reduce emissions during oil and gas production should consider implementing additional emission control requirements, such as plunger lift systems, fugitive methane leak detection and repair (LDAR) programs, and replacement of existing high-bleed pneumatic valves. Enhanced LDAR remote monitoring systems for production and control equipment could aid in both system maintenance and regulatory compliance. Several states, including California, Colorado, and Wyoming, have recently adopted requirements for LDAR at well pads and have achieved sufficient emissions reductions through these LDAR programs.

Oil- and gas-producing wells and associated process equipment are associated with high emissions of methane and HDAPs including benzene, alkanes, carbon monoxide, and toluene. The largest VOC emissions were attributed to flashing losses from crude oil and condensate storage tanks; fugitive emissions from leaks in valves, fittings, and other equipment; venting of hydrocarbons from completions and blowdowns; venting from glycol dehydration units; and natural gas-driven pneumatic devices. Oil-producing wells had the highest toluene and benzene mixing ratios, whereas dry-gas wells were major contributors of methane, ethane, and carbon monoxide.

Recommendation: Control measures, such as the use of advanced plunger lift control algorithms, should be implemented in tandem with continued monitoring and optimization over the life cycle of the well.

PSE

Mature wells with low reservoir pressure and high rates of liquids production are more likely to have high, geographically concentrated emissions during liquid unloadings. Control measures, such as the use of advanced plunger lift control algorithms, have been used to reduce venting during liquid unloading activities; however, changes in reservoir characteristics over time will affect the type of solution deployed.

5.2. Review of midstream methane and HDAP studies

We identified nine studies published from 2015–2020 that evaluated methane and HDAP emissions from midstream oil and gas development, equipment, and processes. Three of these studies evaluated methane and associated co-pollutant emissions released during the 2015 natural gas blowout event at the Aliso Canyon gas storage facility in Los Angeles, California. This event was the largest anthropogenic release of methane from a single point source in the United States and brought much attention to underground gas storage and our reliance on an aging infrastructure.

We also identified one additional peer-reviewed study, published in 2021, that evaluates methane and HDAP emissions from leaking natural gas distribution pipelines (Anderson et al. 2021). While out of the scope of the review time period, due to the limited number of studies that focus on emissions from midstream processes, we include a discussion of its findings and key recommendations here.

5.2.1. Midstream methane and HDAPs: findings and conclusions

Results from (Conley et al. 2016) show leak rates for methane of up to 60 metric tons (MT) per hour and for ethane of up to 4.5 MT/hr from the Aliso Canyon blowout event. The estimated leak rate for methane from this one event is comparable to total methane emission rates from entire oil- and gas-producing regions in the United States (e.g., Barnett Shale, 76 MT/hr; Fayetteville Shale, 39 MT/hr) (Conley et al. 2016). Methane to benzene enhancement ratios estimated in Conley et al. (2016) suggest minimal variation over time occurs in the benzene composition of leaking natural gas. This minimal variation in benzene over time is notable considering the very high levels of methane and ethane detected in the San Fernando Valley, a densely populated community located only a few kilometers south of the Aliso Canyon leak, demonstrating that downwind transport of elevated pollution plumes from unintentional natural gas releases can be substantial.

Findings from Conley et al. (2016) are consistent with results from Jacobson (2019), which found the Aliso Canyon 2015 methane leak significantly increased mixing ratios of additional by-products, including carbon monoxide, formaldehyde, acetaldehyde, peroxyacetyl nitrate, and ozone.

Another study on the Aliso Canyon leak, Garcia-Gonzales et al. (2019), deployed air monitors in the nearby Porter Ranch community at various sensitive receptor locations. Results from the

PSE

24-hr canister samples found strong correlations among methane and several other non-methane VOCs, including n-hexane, toluene, styrene, and benzene. These findings corresponded with results from the 5-minute "trigger" canister samples taken downwind from the blowout event, which found a significant (p-value < 0.0001) relationship among peak methane releases from the site and the following hazardous air pollutants (HAPs): n-hexane, benzene, 2,2,4-trimethypentane, m,p-xylene, ethylbenzene, toluene, and o-xylene (Garcia-Gonzales et al. 2019). The authors note that while average values were below acute, health-based benchmarks, HAP concentrations from individual 5-min canister samples exceeded the 8-h and chronic refence exposure levels (RELs) set by the Office of Environmental Health Hazard Assessment (Garcia-Gonzales et al. 2019). Therefore, if these results are adjusted to fit within a chronic exposure time frame, HAPs co-emitted with methane may have posed a substantial health risk to the adjacent Porter Ranch community and other neighborhoods near oil and natural gas storage facilities.

In addition, Garcia-Gonzales et al. (2019) evaluated criteria air pollutants associated with the methane leak and found particulate concentrations were most strongly correlated with methane releases during the last few days of the leak event (before leak control was implemented), suggesting a potential co-emission may have occurred during the final "well kill attempt" (Garcia-Gonzales et al. 2019).

Finally, using information from the U.S. EPA's National Emissions Inventory and Greenhouse Gas Inventory (bottom-up), (Russo and Carpenter, 2019) evaluated emissions from compressor stations in New York and found methane was co-emitted with various HDAPs, with significant emissions of carbon dioxide, methane, NO_x, carbon monoxide, formaldehyde, and particulates found. In addition, substantial concentrations of 38 different carcinogens were detected, the sum of which could present a potential health risk to nearby populations.

Anderson et al. (2021) evaluated methane and HDAP emissions in Philadelphia, Pennsylvania. Findings from this study suggest the majority of urban methane emissions (which are due to continuous leaks from aging natural gas infrastructure) follow a diurnal pattern. This finding is interesting given that leaking natural gas infrastructure in urban environments has little temporal variability and often results in "near constant emissions throughout the day" (Anderson et al. 2021). While the driving factors are unclear, mixing layer dynamics in combination with emissions is one potential explanation for the diurnal methane profile observed. More research is needed to understand the diurnal pattern.

Anderson et al. (2021) also reported evidence of errors and underestimations when comparing their emissions estimates with existing emissions inventories. The authors found evidence of errors within the National Emissions Inventory (NEI), specifically for nitrogen oxides (NO_x) and carbon monoxide fuel-based emissions factors. Similarly, comparison of measured methane/carbon dioxide ratios to U.S. EPA emissions inventories demonstrated that existing emissions inventories underestimate methane emissions by nearly a factor of four (Anderson

PSE

et al. 2021). Based on these findings, the authors recommended additional long-term monitoring networks in urban environments.

Below we summarize the main findings and conclusions from the midstream methane and HDAP studies published from 2015–2020. These findings are categorized into two main source types: (1) natural gas storage facilities; and (2) compressor stations along natural gas pipelines. Results from each study are summarized in Table 5.2.

Table 5.2. Summary of main findings related to methane and HDAPs from midstream sources, listed by study.

Midstream source(s) evaluated	Main findings related to methane and HDAPs	Study
Natural gas storage facilities (unintentional release event at Aliso Canyon, Los Angeles, California)	- Uncontrolled leaks from midstream oil and gas sites, such as those observed at natural gas storage facilities, could be a potential source of harmful air pollutants. - Strong correlations among methane and several other non-methane VOCs, including n-hexane, toluene, styrene, and benzene from 24-hr canister samples. - A significant relationship among peak methane releases from the site (5-min "trigger" canister samples) and the following HAPs: n-hexane, benzene, 2,2,4-trimethypentane, m/p-xylene, ethylbenzene, toluene, and o-xylene. - Particulate concentrations were most strongly correlated with elevated methane levels during the last few days of the leak event.	Garcia-Gonzales et al. 2019
	- Leak rates of methane from this single unintentional release event were comparable to total methane emission rates from entire oil- and gas-producing regions in the U.S. (e.g., Barnett Shale, 76 MT/hr; Fayetteville Shale, 39 MT/hr). - Methane to benzene enhancement ratios were consistent across samples, suggesting that minimal variation over time in the benzene composition of leaking natural gas likely occurred. - Hydrocarbon composition results taken from surface locations downwind from the leak were consistent with a leak of "pipeline-quality processed natural gas." - Detectable plume enhancements of ethane, propane, and butanes (i.e., natural gas liquids), pentanes and longer-chain hydrocarbons (i.e., condensates), and trace enhancements of benzene, toluene, ethylbenzene, and xylenes downwind from the leak.	Conley et al. 2016
	- Leak rates of methane from this single unintentional release event to be associated with the following by-products, suggesting a co-release: carbon monoxide, formaldehyde, acetaldehyde, peroxyacetyl nitrate, and ozone. - These findings are specific to California, which is VOC-limited, and are chemistry dependent; therefore, these findings may not be reflected in other geographic regions.	Jacobson, 2019

PSE

Midstream source(s) evaluated	Main findings related to methane and HDAPs	Study
Compressor stations along natural gas pipelines	- Emissions from compressor stations were severely underestimated. - Emissions from site to site were highly variable, most likely due to differences in equipment deployed on-site. - The largest releases in emissions, by far, were nitrogen oxides and carbon monoxide, followed by VOCs, known carcinogens including benzene, formaldehyde, and 1,3-butadiene, and coarse particulate matter (PM_{10}). - 36 additional carcinogens were released by compressor stations. - High levels of carbon dioxide and methane were co-emitted with health-relevant pollutants.	Russo and Carpenter, 2019

5.2.2. Midstream methane and HDAPs: recommendations

A small number of studies focus on HDAPs and HDAP-methane co-emissions by component and mitigation strategy from midstream oil and gas sources. Even so, the handful of studies that do evaluate co-emitting oil and gas sources provide several recommendations of feasible emission control options and policy improvements. Below we identify emission control and policy recommendations aimed at reducing both methane and HDAP emissions from the midstream oil and gas sector.

Natural gas storage facilities

Recommendation: Facility-specific meteorological and continuous air quality data-collection equipment should be installed at natural gas storage facilities. Support of environmental surveillance after severe off-normal operation events (e.g., accidental pollutant releases) should be considered to ensure harmful exposures are properly monitored and promptly mitigated.

Single-point failures of natural gas storage facilities, such as the event that occurred at Aliso Canyon, California, in 2015, can severely impede emission control strategies and present a potentially harmful exposure risk, especially to proximate populations. HDAPs are co-emitted with unintentional methane releases, and there are additional emission control strategies available to further reduce these emissions.

More comprehensive and detailed emissions data related to gas storage facilities would allow for a more accurate determination of the cumulative air pollutant emissions and associated exposure risks for sensitive populations. Continuous monitoring would also be better able to capture emission trends over time. This finding is supported by Alden et al. (2020), which found methane emissions at gas storage facilities can change rapidly through time and vary according to operating phase. The authors conclude that "continuous monitoring captures

PSE

large emission events and lognormal distribution of emissions, both of which are missed by intermittent aircraft sampling" (Alden et al. 2020). Support of environmental surveillance after severe off-normal operation events (e.g., accidental pollutant releases) is especially important when considering exposure risks to proximate populations downwind of blowout events, as findings suggest minimal variation occurs over time in the benzene (a known human carcinogen) composition of leaking natural gas.

<u>Compressor stations</u>

Methane emissions from compressor stations are co-emitted with HDAPs. Significant emissions of carbon dioxide, methane, NO_X, carbon dioxide, and coarse particulate matter (PM_{10}) in addition to 39 carcinogens were found, including formaldehyde, benzene, and 1,3-butadiene.

The findings reported in Russo and Carpenter (2019) "add to the evidence for urgency for the replacement of fossil fuels with renewable sources of energy in order to both protect human health and reduce the immediate and long-term threats arising from climate change." This would encompass near-term transition measures in order to achieve long-term reductions.

5.3. Review of downstream methane and HDAP studies

Five studies evaluated the emissions of both HDAPs and methane from downstream oil and gas development activities and equipment (Cui et al. 2015; Lopez-Coto et al. 2020; Nsanzineza et al. 2019; Oltmans et al. 2016; Sun et al. 2019). Two of these studies focused on emissions from both upstream and downstream oil and gas sources (Cui et al. 2015; Nsanzineza et al. 2019).

Using the MARKAL (MARKet ALlocation) least-cost planning model with the U.S. EPA's nine-region energy system database and the Comprehensive Air Quality Model with Extensions (CAMx), Nsanzieza et al (2019) evaluated emissions from both downstream electricity systems and upstream oil production in Colorado, northern New Mexico, Utah, and Wyoming. Findings from Nsanzineza et al. (2019) illustrate that the implementation of greenhouse gas (GHG) emission fees, and regulations aimed to reduce GHG emissions, would also reduce methane emissions from both upstream and downstream sources by 42%, emissions of nitrogen oxides by 48%, and VOC emissions by 42% by 2030. A combination of power plant retirements, fuel switching, and nitrogen oxides control regulations would reduce summertime emissions of nitrogen oxides from power plants by 50%. Similarly, under the GHG fees scenario, annual power plant emissions of nitrogen oxides would decrease by 40% in 2030. The authors also note that by reducing GHG emissions there would also be a reduction in mortality of approximately 200 deaths annually, because emission reductions of nitrogen oxides and VOCs would reduce overall ozone concentrations in these regions (Nsanzineza et al. 2019).

Lopez-Coto et al. (2020) evaluated power plant and traffic emissions in the Washington, D.C. – Baltimore, Maryland metropolitan area using a combination of top-down (i.e., two airborne

PSE

sampling campaigns) and bottom-up approaches (i.e., greenhouse gas inventories). Using hourly data provided in the Continuous Emissions Monitoring System (CEMS) for power plants, the authors find power plant emissions of methane, carbon dioxide, and carbon monoxide to vary greatly throughout the day, up to a factor of two, with larger variations observed between days due to occasional shutdowns. Large variations in hourly operating conditions of power plants and traffic in the region, combined with variations in sampling time and space of the flight campaigns, both contributed to the variation observed, with 97% of this variability explained when the sampling time and location of major sources that exhibit large hourly variability, such as power plants, are considered.

Cui et al. (2015) also evaluated methane and HDAP emissions from upstream (e.g., oil and gas production) and downstream (e.g., transmission and distribution systems) oil and gas sources in the Southern California Air Basin. The authors created two inventories of methane and carbon monoxide emissions – a "prior" inventory, which relies on data from the 2005 National Emissions Inventory (NEI) only, emissions of which are very small for the oil and gas sector; and a "posterior" inventory, which relies on data from the 2011 NEI and considers top-down observations from large industrial sources, including oil and gas, collected from six flights within the Air Basin. The ratios between methane and carbon monoxide emissions, which show the slope of correlation between the atmospheric mixing ratios, were consistently found to be in better agreement with top-down observations in the posterior inventory than the prior inventory (Cui et al. 2015). Ratios from the prior inventory were found to be underestimated and exemplified a more homogenous spatial distribution across the Basin, which is most likely due to the lack of emissions estimates from the oil and gas sector in the prior model. Methane-carbon monoxide ratios in the posterior inventory were consistent with top-down observations and demonstrated a heterogenous spatial distribution, with larger ratios found near specific industrial sites, such as oil and gas sources. Specifically, emissions from the posterior inventory were found to be 1.8 times higher than the prior inventory.

Sun et al. (2019) estimated GHG and criteria air pollutant emissions from refinery activities in the United States and provided emissions for both primary and secondary refinery products using data from the Greenhouse Gas Reporting Program and National Emissions Inventory data from 2014 (bottom-up). Results of the study indicate that combustion (e.g., heat and steam generation, which result from the combustion of fuel to supply energy) is the major source of carbon dioxide, carbon monoxide, nitrogen dioxide, nitrogen oxides, and particulate matter (PM_{10}, $PM_{2.5}$) emissions, while the primary source of methane and VOCs were attributed to facility-wide emissions (e.g., emissions from the cooling water supply system; wastewater treatment plant; and flare, fugitive, tanker, and other auxiliary processes). More specifically, flare and fugitive emissions were responsible for a large portion of methane emissions, while tanks and other fugitive sources were the predominant source of VOC emissions. Nationally, most combustion emissions from refineries are due to combined gas combustion followed by catalyst coke, accounting for 91–100% of emissions of the pollutants studied.

5.3.1. Downstream methane and HDAPs: findings and conclusions

Below we summarize the main findings and conclusions from the downstream methane and HDAP studies. These findings are categorized into two source types: energy generation (i.e., power plants) and refineries. Results from each study are summarized in Table 5.3.

Table 5.3. Summary of main findings related to methane and HDAPs from downstream sources, listed by study.

Downstream source(s) evaluated	Main findings related to methane and HDAPs	Study
Energy generation (e.g., power plants)	- Focused on both downstream electricity systems emissions and upstream oil production emissions. Looked at scenarios under different schemes of oil and gas production and energy production from renewable and nonrenewable sources. - Summer nitrogen oxides, sulfur dioxide, carbon dioxide emissions from electricity generation from oil and natural gas (e.g., natural gas- and coal-powered power plants) have a negative percent change from 2030 baseline under all three scenarios (cheap gas, costly gas, and greenhouse gas (GHG) fees). - The greatest percent decrease was seen under the GHG fees scenario. - The GHG fee scenario showed a decrease in generation of electricity from natural gas as it shifted towards production from renewables. - The cheap gas scenario showed an increase in electricity production from natural gas and a decrease in production from coal. - Reductions in emissions from power plants are greater on an annual scale compared to the summer season. - Overall reductions in ozone formation were a result of reduced nitrogen oxides emissions from power plants, as well as oil and natural gas production and transportation areas. - The reduced emissions resulting from less reliance on coal for electricity production could be offset by emissions from increased production of electricity from oil and gas. - Implementing GHG fees could promote the use of renewable energy sources and as a result reduce emissions from both power plants and oil and gas production.	Nsanzineza et al. 2019
	- Emissions of methane, carbon dioxide, and carbon monoxide from power plants vary greatly throughout the day, up to a factor of two, with larger variations observed between days due to occasional shutdowns. - 97% of the variability observed can be explained when the sampling time and location of major sources that exhibit large hourly variability, such as power plants, are considered.	Lopez-Coto et al. 2020

PSE

Downstream source(s) evaluated	Main findings related to methane and HDAPs	Study
Energy generation (e.g., power plants)	- Due to the quick deployment and large spatial coverage provided by a mobile platform, flight campaigns are useful for emissions estimation; however, they are limited in the temporal coverage that they can provide. - Irregular sampling in time and space can both contribute to the variability of emissions estimates as well as impact the emissions estimates themselves.	Lopez-Coto et al. 2020 (*continued*)
Refineries	- Established baseline data for GHG and criteria air pollutant emissions from refinery products that can be used as a reference for future studies evaluating refinery emissions. - Nationally, most combustion emissions from refineries are due to oil and gas combustion followed by catalyst coke, accounting for almost all emissions of the pollutants studied. - Combined gas (natural gas and refinery gas) was the top source for criteria air pollutant emissions. - Some emissions factors inconsistencies were seen across the Petroleum Administration for Defense Districts (PADDs) for carbon dioxide. - Liquid petroleum gas emission factors were different (low) from previously established emissions factors, distillate carbon dioxide emissions factors were higher, and residual GHG and criteria air pollutant emissions factors were similar to those in the AP-42 and GREET models.	Sun et al. 2019
Distribution systems	- Evaluated methane and HDAP emissions from upstream (e.g., oil and gas production) and downstream (e.g., transmission and distribution systems) oil and gas sources in the Southern California Air Basin by creating two inventories – prior and posterior. - Ratios of methane to carbon monoxide, which show the slope of correlation between the atmospheric mixing ratios, were consistently found to be in better agreement with top-down observations in the posterior inventory than the prior inventory. - Ratios from the prior inventory were found to be underestimated and exemplified a more homogenous spatial distribution across the Basin - most likely due to the lack of emissions estimates from the oil and gas sector in the prior model. - Methane-carbon monoxide ratios in the posterior inventory were consistent with top-down observations and demonstrated a heterogenous spatial distribution, with larger ratios found near specific industrial sites, such as oil and gas sources. - Emissions from the posterior inventory were found to be 1.8 times higher than the prior inventory.	Cui et al. 2015

PSE

5.3.2. Downstream methane and HDAPs: recommendations

A small number of studies focus on HDAPs and HDAP-methane co-emissions by component and mitigation strategy from downstream sources. As a result, the ability to accurately monitor and predict emissions from downstream sources and evaluate the associated environmental and health impacts is severely limited. Studies consistently report the over- or under-estimation of emissions depending on the component studied and measurement method used (e.g., top-down vs. bottom-up). The observed discrepancy between methods is most likely due to the inability to account for emissions from all sources (e.g., super-emitters, unidentified leaks), something that future researchers should consider. Even so, the four downstream studies discussed here provide useful insight into the emissions profiles from power plants and refineries in the United States and provide emission control strategies and other mitigation measures that would reduce both methane and HDAP emissions. Below is a summary of these recommendations.

<u>Power plants</u>

Recommendation: A combination of power plant retirements, fuel switching, and nitrogen oxides control regulations would significantly reduce emissions of nitrogen oxides from power plants, thereby reducing the formation of tropospheric ozone. The implementation of fees on greenhouse gas emissions could also reduce methane and numerous HDAPs emissions by encouraging the move towards renewables. The implementation of similar fees for HDAPs should also be considered.

Natural gas-powered power plants are a predominant point source of methane, nitrogen oxides, sulfur dioxide, and VOCs, and contribute to ozone formation. Nsanzineza et al. (2019) supports the implementation of greenhouse gas fees to promote reliance on renewable energy and decrease HDAP and methane emissions from the oil and natural gas sector. The fees applied in this study were $55[3] per metric ton of carbon dioxide and $1400 per metric ton of methane, based on the Interagency Working Group's (2013) social cost of carbon for a 3% discount rate (Nsanzineza et al. 2019). Therefore, future regulations should consider incorporating fees for methane and carbon dioxide to encourage operators to look for alternative, clean methods for power generation. While not discussed in Nsanzineza et al. (2019), the implementation of similar fees for HDAPs should also be considered. Cost estimates for criteria air pollutants, for example, are available and could help to reduce harmful emissions released by natural gas-powered power plants (CACES, 2021).

---

[3] Value represents fee pricing in 2011 dollars.

PSE

Recommendation: Future airborne measurement campaigns focused on sources with large hourly variability, such as power plants, include multiple flights over a region at different times of the day, week, month, and year (i.e., summer vs. winter conditions) to account for temporal variability in emissions and source activities, and if possible, multiple aircraft should be deployed together in an effort to maximize the spatial coverage of the campaign.

Power plants exhibit large variability in emissions throughout the day and week due to differences in operating conditions and activities. Due to the quick deployment and large spatial coverage provided by a mobile platform, flight campaigns are useful for emissions estimation; however, flight campaigns are limited in the temporal coverage that they can provide. Irregular sampling in time and space can both contribute to the variability of emissions estimates as well as impact the emissions estimates themselves.

Combustion activities at refineries

Recommendation: Current reporting requirements are insufficient at accurately capturing emissions from combustion of combined gas (natural gas and still gas). Additional mandates for emissions reporting for refinery operators, such as requiring the reporting of complete air emission data based on consistent operation, emission data, and methodology (i.e., consistent year, region, facility coverage, etc.), should be considered to ensure that reported emissions and fuel-consumption are reflective of real-life operations.

Combustion activities account for the majority of methane and HDAP emissions released from refineries. Nationally, combustion-related emissions from refineries are attributed to the use of refinery still gas and natural gas (i.e., combined gas), followed by refinery catalyst coke — together accounting for 91–100% of emissions (Sun et al. 2019). These fuels are also the major fuel types used in U.S. refineries, together accounting for 98% of the combustion energy produced on-site. Refineries can either use continuous emissions monitoring or specific emission calculation methods (as approved by the U.S. EPA) when reporting emissions. This variation in reporting method may result in "variable accuracy of emissions from various facilities, units, or combustion fuels" (Sun et al. 2019). Miscategorization of fuel types have also resulted in major discrepancies among refinery fuel consumption and associated criteria air pollutant and greenhouse gas emissions. As stated by Sun et al. (2019),

"Many individual facilities do not report refinery still gas consumption or report no natural gas consumption, both of which are unlikely. Furthermore, carbon dioxide releases from fuel combustion should be consistent across all combustion technologies based on the carbon content of the fuel. However, carbon dioxide emissions estimates from reported natural gas combustion are significantly lower than the normal range, while emissions from reported refinery gas consumption are significantly higher than the normal range, indicating some reporting errors through miscategorizating fuel types."

These issues, coupled with variations in refinery operations and associated variations in emissions (i.e., by year, region, facility, emission control technology, etc.) justify the reporting of complete air emission data based on consistent operation, emission data, and methodology (i.e., consistent year, region, facility coverage, etc.).

Recommendation: Further development of combustion control technologies should be considered. Examples include: (1) the adoption of new front-end fuel combustion technology, which reduces emissions of nitrogen oxides and carbon monoxide; (2) the implementation of continuous monitoring using more sensitive instruments to capture and adequately mitigate pollutant releases and leaks when they happen; and (3) the implementation of responsive system controls for specific refinery units (e.g., combustion, flare control, cool tower, sulfur recovery unit), which could promote more efficient fuel combustion and reduce process leak, thus reducing pollutant releases.

In the past couple of decades, refineries in the United States have reduced their air pollutant emissions through the implementation of emission control measures, such as selective catalytic reduction and ultralow nitrogen oxides burner technologies. However, refineries are still a large point source of methane and HDAP emissions. Combustion activities (e.g., heat and steam generation) at refineries are a major source of greenhouse gases (carbon dioxide, nitrogen dioxide, methane) and criteria air pollutants (VOCs, carbon monoxide, nitrogen oxides, PM_{10}, $PM_{2.5}$). Therefore, additional efforts should be made to further reduce their emissions.

Distribution systems

Recommendation: Top-down measurement approaches, such as airborne measurements, could be one method to monitor emissions and associated sources within the Southern California Air Basin.

Findings from Cui et al. (2015) demonstrate that bottom-up inventories provided by states such as California underestimate emissions from larger sources, such as urban oil and gas distribution systems.

PSE

References

Abeleira, A., Pollack, I. B., Sive, B., Zhou, Y., Fischer, E. V., & Farmer, D. K. (2017). Source characterization of volatile organic compounds in the Colorado Northern Front Range Metropolitan Area during spring and summer 2015: SOURCES OF VOCS IN THE COLORADO NFRMA. Journal of Geophysical Research: Atmospheres, 122(6), 3595–3613. https://doi.org/10.1002/2016JD026227.

Ahmadov, R., McKeen, S., Trainer, M., Banta, R., Brewer, A., Brown, S., Edwards, P. M., de Gouw, J. A., Frost, G. J., Gilman, J., Helmig, D., Johnson, B., Karion, A., Koss, A., Langford, A., Lerner, B., Olson, J., Oltmans, S., Peischl, J., … Zamora, R. (2015). Understanding high wintertime ozone pollution events in an oil- and natural gas-producing region of the western US. Atmospheric Chemistry and Physics, 15(1), 411–429. https://doi.org/10.5194/acp-15-411-2015.

Alberta Energy Regulator. (2021). Submission Checklist for Alternative FEMP Proposals. https://www.aer.ca/regulating-development/rules-and-directives/directives/submission-checklist-for-alternative-femp-proposals.

Alden, C., Wright, R., Coburn, S., Caputi, D., Wendland, G., Rybchuk, A., Conley, S., Faloona, I., & Rieker, G. (2020). Temporal Variability of Emissions Revealed by Continuous, Long-Term Monitoring of an Underground Natural Gas Storage Facility. Environmental Science & Technology, 54(22), 14589–14597. https://doi.org/10.1021/acs.est.0c03175.

Allen, D. T. (2016a). Emissions from oil and gas operations in the United States and their air quality implications. Journal of the Air & Waste Management Association, 66(6), 549–575. https://doi.org/10.1080/10962247.2016.1171263.

Allen, D. T. (2016b). Attributing Atmospheric Methane to Anthropogenic Emission Sources. Accounts of Chemical Research, 49(7), 1344–1350. https://doi.org/10.1021/acs.accounts.6b00081.

Allen, D. T., Cardoso-Saldaña, F. J., & Kimura, Y. (2017). Variability in Spatially and Temporally Resolved Emissions and Hydrocarbon Source Fingerprints for Oil and Gas Sources in Shale Gas Production Regions. Environmental Science & Technology, 51(20), 12016–12026. https://doi.org/10.1021/acs.est.7b02202.

Allen, D. T., Pacsi, A. P., Sullivan, D. W., Zavala-Araiza, D., Harrison, M., Keen, K., Fraser, M. P., Daniel Hill, A., Sawyer, R. F., & Seinfeld, J. H. (2015a). Methane Emissions from Process Equipment at Natural Gas Production Sites in the United States: Pneumatic Controllers. Environmental Science & Technology, 49(1), 633–640. https://doi.org/10.1021/es5040156.

Allen, D. T., Smith, D., Torres, V. M., & Saldaña, F. C. (2016). Carbon dioxide, methane and black carbon emissions from upstream oil and gas flaring in the United States. Current Opinion in Chemical Engineering, 13, 119–123. https://doi.org/10.1016/j.coche.2016.08.014.

Allen, D. T., Sullivan, D. W., Zavala-Araiza, D., Pacsi, A. P., Harrison, M., Keen, K., Fraser, M. P., Daniel Hill, A., Lamb, B. K., Sawyer, R. F., & Seinfeld, J. H. (2015b). Methane Emissions from Process Equipment at Natural Gas Production Sites in the United States: Liquid Unloadings. Environmental Science & Technology, 49(1), 641–648. https://doi.org/10.1021/es504016r.

Allen, D. T., Torres, V. M., Thomas, J., Sullivan, D. W., Harrison, M., Hendler, A., Herndon, S. C., Kolb, C. E., Fraser, M. P., Hill, A. D., Lamb, B. K., Miskimins, J., Sawyer, R. F., & Seinfeld, J. H. (2013). Measurements of methane emissions at natural gas production sites in the United States. Proceedings of the National Academy of Sciences, 110(44), 17768–17773. https://doi.org/10.1073/pnas.1304880110.

PSE

Allshouse, W. B., McKenzie, L. M., Barton, K., Brindley, S., & Adgate, J. L. (2019). Community Noise and Air Pollution Exposure During the Development of a Multi-Well Oil and Gas Pad. Environmental Science & Technology, 53(12), 7126–7135. https://doi.org/10.1021/acs.est.9b00052.

Alnahdi, A., Elkamel, A., Shaik, M. A., Al-Sobhi, S. A., & Erenay, F. S. (2019). Optimal Production Planning and Pollution Control in Petroleum Refineries Using Mathematical Programming and Dispersion Models. Sustainability, 11(14), 3771. https://doi.org/10.3390/su11143771.

Alvarez, R. A., Pacala, S. W., Winebrake, J. J., Chameides, W. L., & Hamburg, S. P. (2012). Greater focus needed on methane leakage from natural gas infrastructure. Proceedings of the National Academy of Sciences, 109(17), 6435–6440. https://doi.org/10.1073/pnas.1202407109.

Alvarez, R. A., Zavala-Araiza, D., Lyon, D. R., Allen, D. T., Barkley, Z. R., Brandt, A. R., Davis, K. J., Herndon, S. C., Jacob, D. J., Karion, A., Kort, E. A., Lamb, B. K., Lauvaux, T., Maasakkers, J. D., Marchese, A. J., Omara, M., Pacala, S. W., Peischl, J., Robinson, A. L., … Hamburg, S. P. (2018). Assessment of methane emissions from the U.S. oil and gas supply chain. Science, 361(6398), 186–188. https://doi.org/10.1126/science.aar7204.

Amoatey, P., Omidvarborna, H., Baawain, M. S., & Al-Mamun, A. (2019). Emissions and exposure assessments of SOX, NOX, PM10/2.5 and trace metals from oil industries: A review study (2000–2018). Process Safety and Environmental Protection, 123, 215–228. https://doi.org/10.1016/j.psep.2019.01.014

Anderson, D. C., Lindsay, A., DeCarlo, P. F., & Wood, E. C. (2021). Urban Emissions of Nitrogen Oxides, Carbon Monoxide, and Methane Determined from Ground-Based Measurements in Philadelphia. Environmental Science & Technology, 55(8), 4532–4541. https://doi.org/10.1021/acs.est.1c00294.

Arata, C., Rahn, T., & Dubey, M. K. (2016). Methane Isotope Instrument Validation and Source Identification at Four Corners, New Mexico, United States. The Journal of Physical Chemistry A, 120(9), 1488–1494. https://doi.org/10.1021/acs.jpca.5b12737.

Atherton, E., Risk, D., Fougère, C., Lavoie, M., Marshall, A., Werring, J., Williams, J. P., & Minions, C. (2017). Mobile measurement of methane emissions from natural gas developments in northeastern British Columbia, Canada. Atmospheric Chemistry and Physics, 17(20), 12405–12420. https://doi.org/10.5194/acp-17-12405-2017.

Baillie, J., Risk, D., Atherton, E., O'Connell, E., Fougère, C., Bourlon, E., & MacKay, K. (2019). Methane emissions from conventional and unconventional oil and gas production sites in southeastern Saskatchewan, Canada. Environmental Research Communications, 1(1), 011003. https://doi.org/10.1088/2515-7620/ab01f2.

Balcombe, P., Brandon, N. P., & Hawkes, A. D. (2018). Characterising the distribution of methane and carbon dioxide emissions from the natural gas supply chain. Journal of Cleaner Production, 172, 2019–2032. https://doi.org/10.1016/j.jclepro.2017.11.223.

Banan, Z., & Gernand, J. M. (2018). Evaluation of gas well setback policy in the Marcellus Shale region of Pennsylvania in relation to emissions of fine particulate matter. Journal of the Air & Waste Management Association, 68(9), 988–1000. https://doi.org/10.1080/10962247.2018.1462866.

Baray, S., Darlington, A., Gordon, M., Hayden, K. L., Leithead, A., Li, S.-M., Liu, P. S. K., Mittermeier, R. L., Moussa, S. G., O'Brien, J., Staebler, R., Wolde, M., Worthy, D., & McLaren, R. (2018). Quantification of methane sources in the Athabasca Oil Sands Region of Alberta by aircraft mass balance. Atmospheric Chemistry and Physics, 18, 7361–7378. https://doi.org/10.5194/acp-18-7361-2018.

PSE

Barkley, Z. R., Davis, K. J., Feng, S., Balashov, N., Fried, A., DiGangi, J., Choi, Y., & Halliday, H. S. (2019a). Forward Modeling and Optimization of Methane Emissions in the South Central United States Using Aircraft Transects Across Frontal Boundaries. Geophysical Research Letters, 46(22), 13564–13573. https://doi.org/10.1029/2019GL084495.

Barkley, Z. R., Lauvaux, T., Davis, K. J., Deng, A., Fried, A., Weibring, P., Richter, D., Walega, J. G., DiGangi, J., Ehrman, S. H., Ren, X., & Dickerson, R. R. (2019b). Estimating Methane Emissions From Underground Coal and Natural Gas Production in Southwestern Pennsylvania. Geophysical Research Letters, 46(8), 4531–4540. https://doi.org/10.1029/2019GL082131.

Barkley, Z. R., Lauvaux, T., Davis, K. J., Deng, A., Miles, N. L., Richardson, S. J., Cao, Y., Sweeney, C., Karion, A., Smith, M., Kort, E. A., Schwietzke, S., Murphy, T., Cervone, G., Martins, D., & Maasakkers, J. D. (2017). Quantifying methane emissions from natural gas production in north-eastern Pennsylvania. Atmospheric Chemistry and Physics, 17(22), 13941–13966. https://doi.org/10.5194/acp-17-13941-2017.

Bean, J. K., Bhandari, S., Bilotto, A., & Hildebrandt Ruiz, L. (2018). Formation of Particulate Matter from the Oxidation of Evaporated Hydraulic Fracturing Wastewater. Environmental Science & Technology, 52(8), 4960–4968. https://doi.org/10.1021/acs.est.7b06009.

Bell, C. S., Vaughn, T. L., Zimmerle, D., Herndon, S. C., Yacovitch, T. I., Heath, G. A., Pétron, G., Edie, R., Field, R. A., Murphy, S. M., Robertson, A. M., & Soltis, J. (2017). Comparison of methane emission estimates from multiple measurement techniques at natural gas production pads. Elem Sci Anth, 5(0), 79. https://doi.org/10.1525/elementa.266.

Bloomdahl, R., Abualfaraj, N., Olson, M., & Gurian, P. L. (2014). Assessing worker exposure to inhaled volatile organic compounds from Marcellus Shale flowback pits. Journal of Natural Gas Science and Engineering, 21, 348–356. https://doi.org/10.1016/j.jngse.2014.08.018.

Bozlaker, A., Peccia, J., & Chellam, S. (2017). Indoor/Outdoor Relationships and Anthropogenic Elemental Signatures in Airborne PM2.5 at a High School: Impacts of Petroleum Refining Emissions on Lanthanoid Enrichment. Environmental Science & Technology, 51(9), 4851–4859. https://doi.org/10.1021/acs.est.6b06252.

Bradshaw, J. L. (2018). Measuring Leak Rates from Abandoned Natural Gas Wells in Western Pennsylvania. 2(1), 5.

Brandt, A. R., Heath, G. A., & Cooley, D. (2016a). Methane Leaks from Natural Gas Systems Follow Extreme Distributions. Environmental Science & Technology, 50(22), 12512–12520. https://doi.org/10.1021/acs.est.6b04303.

Brandt, A. R., Heath, G. A., Kort, E. A., O'Sullivan, F., Pétron, G., Jordaan, S. M., Tans, P., Wilcox, J., Gopstein, A. M., Arent, D., Wofsy, S., Brown, N. J., Bradley, R., Stucky, G. D., Eardley, D., & Harriss, R. (2014). Methane Leaks from North American Natural Gas Systems. Science, 343(6172), 733–735. https://doi.org/10.1126/science.1247045.

Brandt, A. R., Yeskoo, T., McNally, M. S., Vafi, K., Yeh, S., Cai, H., & Wang, M. Q. (2016b). Energy Intensity and Greenhouse Gas Emissions from Tight Oil Production in the Bakken Formation. Energy & Fuels, 30(11), 9613–9621. https://doi.org/10.1021/acs.energyfuels.6b01907.

Brantley, H. L., Thoma, E. D., & Eisele, A. P. (2015). Assessment of volatile organic compound and hazardous air pollutant emissions from oil and natural gas well pads using mobile remote and on-site direct measurements. Journal of the Air & Waste Management Association, 65(9), 1072–1082. https://doi.org/10.1080/10962247.2015.1056888.

PSE

Bruhwiler, L. M., Basu, S., Bergamaschi, P., Bousquet, P., Dlugokencky, E., Houweling, S., Ishizawa, M., Kim, H.-S., Locatelli, R., Maksyutov, S., Montzka, S., Pandey, S., Patra, P. K., Petron, G., Saunois, M., Sweeney, C., Schwietzke, S., Tans, P., & Weatherhead, E. C. (2017). U.S. CH4 emissions from oil and gas production: Have recent large increases been detected?: U.S. Emissions From Oil and Gas Production. Journal of Geophysical Research: Atmospheres, 122(7), 4070–4083. https://doi.org/10.1002/2016JD026157.

Buchwitz, M., Schneising, O., Reuter, M., Heymann, J., Krautwurst, S., Bovensmann, H., Burrows, J. P., Boesch, H., Parker, R. J., Somkuti, P., Detmers, R. G., Hasekamp, O. P., Aben, I., Butz, A., Frankenberg, C., & Turner, A. J. (2017). Satellite-derived methane hotspot emission estimates using a fast data-driven method. Atmospheric Chemistry and Physics, 17(9), 5751–5774. https://doi.org/10.5194/acp-17-5751-2017.

Buonocore, J., Reka, S., Yang, D., Change, C., Roy, A., Thompson, T., Lyon, D., McVay, R., Michanowicz, D., & Arunachalam, S. (2021). Air quality and health benefits of methane policies in the United States Oil & Gas Sector. https://doi.org/10.1038/s43247-021-00252-1.

Burns, A., Shin, J. (Mi), Unice, K. M., Gaffney, S. H., Kreider, M. L., Gelatt, R. H., & Panko, J. M. (2017). Combined analysis of job and task benzene air exposures among workers at four US refinery operations. Toxicology and Industrial Health, 33(3), 193–210. https://doi.org/10.1177/0748233715619072.

CACES (Center for Air, Climate, and Energy Solutions). (2021). The Center for Air, Climate, and Energy Solutions Economic Costs. CACES. https://www.caces.us.

Cambaliza, M. O. L., Shepson, P. B., Bogner, J., Caulton, D. R., Stirm, B., Sweeney, C., Montzka, S. A., Gurney, K. R., Spokas, K., Salmon, O. E., Lavoie, T. N., Hendricks, A., Mays, K., Turnbull, J., Miller, B. R., Lauvaux, T., Davis, K., Karion, A., Moser, B., … Richardson, S. (2015). Quantification and source apportionment of the methane emission flux from the city of Indianapolis. Elem Sci Anth, 3(0), 000037. https://doi.org/10.12952/journal.elementa.000037.

Cardoso-Saldaña, F. J., Kimura, Y., Stanley, P., McGaughey, G., Herndon, S. C., Roscioli, J. R., Yacovitch, T. I., & Allen, D. T. (2019). Use of Light Alkane Fingerprints in Attributing Emissions from Oil and Gas Production. Environmental Science & Technology, 53(9), 5483–5492. https://doi.org/10.1021/acs.est.8b05828.

Caulton, D. R., Li, Q., Bou-Zeid, E., Fitts, J. P., Golston, L. M., Pan, D., Lu, J., Lane, H. M., Buchholz, B., Guo, X., McSpiritt, J., Wendt, L., & Zondlo, M. A. (2018). Quantifying uncertainties from mobile-laboratory-derived emissions of well pads using inverse Gaussian methods. Atmospheric Chemistry and Physics, 18(20), 15145–15168. https://doi.org/10.5194/acp-18-15145-2018.

Caulton, D. R., Lu, J. M., Lane, H. M., Buchholz, B., Fitts, J. P., Golston, L. M., Guo, X., Li, Q., McSpiritt, J., Pan, D., Wendt, L., Bou-Zeid, E., & Zondlo, M. A. (2019). Importance of Superemitter Natural Gas Well Pads in the Marcellus Shale. Environmental Science & Technology, 53(9), 4747–4754. https://doi.org/10.1021/acs.est.8b06965.

Caulton, D. R., Shepson, P. B., Cambaliza, M. O. L., McCabe, D., Baum, E., & Stirm, B. H. (2014a). Methane Destruction Efficiency of Natural Gas Flares Associated with Shale Formation Wells. Environmental Science & Technology, 48(16), 9548–9554. https://doi.org/10.1021/es500511w.

PSE

Caulton, D. R., Shepson, P. B., Santoro, R. L., Sparks, J. P., Howarth, R. W., Ingraffea, A. R., Cambaliza, M. O. L., Sweeney, C., Karion, A., Davis, K. J., Stirm, B. H., Montzka, S. A., & Miller, B. R. (2014b). Toward a better understanding and quantification of methane emissions from shale gas development. Proceedings of the National Academy of Sciences, 111(17), 6237–6242. https://doi.org/10.1073/pnas.1316546111.

Chamberlain, S. D., Ingraffea, A. R., & Sparks, J. P. (2016). Sourcing methane and carbon dioxide emissions from a small city: Influence of natural gas leakage and combustion. Environmental Pollution, 218, 102–110. https://doi.org/10.1016/j.envpol.2016.08.036.

Chan, E., Worthy, D. E., Chan, D., Ishizawa, M., Moran, M. D., Delcloo, A., & Vogel, F. (2020). Eight-year estimates of methane emissions from oil and gas operations in Western Canada are nearly twice those reported in inventories. Environmental Science & Technology, 54(23), 14899–14909.

Chen, H., & Carter, K. E. (2020). Hazardous substances as the dominant non-methane volatile organic compounds with potential emissions from liquid storage tanks during well fracturing: A modeling approach. Journal of Environmental Management, 268, 110715. https://doi.org/10.1016/j.jenvman.2020.110715.

Conley, S., Faloona, I., Mehrotra, S., Suard, M., Lenschow, D. H., Sweeney, C., Herndon, S., Schwietzke, S., Pétron, G., & Pifer, J. (2017). Application of Gauss's theorem to quantify localized surface emissions from airborne measurements of wind and trace gases. Atmospheric Measurement Techniques, 10(9), 3345–3358.

Conley, S., Franco, G., Faloona, I., Blake, D. R., Peischl, J., & Ryerson, T. B. (2016). Methane emissions from the 2015 Aliso Canyon blowout in Los Angeles, CA. Science, 351(6279), 1317–1320. https://doi.org/10.1126/science.aaf2348.

Cui, Y. Y., Brioude, J., McKeen, S. A., Angevine, W. M., Kim, S.-W., Frost, G. J., Ahmadov, R., Peischl, J., Bousserez, N., Liu, Z., Ryerson, T. B., Wofsy, S. C., Santoni, G. W., Kort, E. A., Fischer, M. L., & Trainer, M. (2015). Top-down estimate of methane emissions in California using a mesoscale inverse modeling technique: The South Coast Air Basin: Inverse Estimate of Methane Emissions. Journal of Geophysical Research: Atmospheres, 120(13), 6698–6711. https://doi.org/10.1002/2014JD023002.

Cui, Y. Y., Henze, D. K., Brioude, J., Angevine, W. M., Liu, Z., Bousserez, N., Guerrette, J., McKeen, S. A., Peischl, J., Yuan, B., Ryerson, T., Frost, G., & Trainer, M. (2019a). Inversion Estimates of Lognormally Distributed Methane Emission Rates From the Haynesville-Bossier Oil and Gas Production Region Using Airborne Measurements. Journal of Geophysical Research: Atmospheres, 124(6), 3520–3531. https://doi.org/10.1029/2018JD029489.

Cui, Y. Y., Vijayan, A., Falk, M., Hsu, Y.-K., Yin, D., Chen, X. M., Zhao, Z., Avise, J., Chen, Y., Verhulst, K., Duren, R., Yadav, V., Miller, C., Weiss, R., Keeling, R., Kim, J., Iraci, L. T., Tanaka, T., Johnson, M. S., … Croes, B. (2019b). A Multiplatform Inversion Estimation of Statewide and Regional Methane Emissions in California during 2014–2016. Environmental Science & Technology, 53(16), 9636–9645. https://doi.org/10.1021/acs.est.9b01769.

Cusworth, D. H., Duren, R. M., Thorpe, A. K., Olson-Duvall, W., Heckler, J., Chapman, J. W., Eastwood, M. L., Helmlinger, M. C., Green, R. O., Asner, G. P., & others. (2021a). Intermittency of Large Methane Emitters in the Permian Basin. Environmental Science & Technology Letters, 8(7), 567–573. https://doi.org/10.1021/acs.estlett.1c00173.

Cusworth, D. H., Duren, R. M., Thorpe, A. K., Pandey, S., Maasakkers, J. D., Aben, I., Jervis, D., Varon, D. J., Jacob, D. J., Randles, C. A., & others. (2021b). Multisatellite imaging of a gas well blowout enables quantification of total methane emissions. Geophysical Research Letters, 48(2), e2020GL090864.

Cusworth, D. H., Duren, R. M., Yadav, V., Thorpe, A. K., Verhulst, K., Sander, S., Hopkins, F., Rafiq, T., & Miller, C. E. (2020). Synthesis of methane observations across scales: Strategies for deploying a multitiered observing network. Geophysical Research Letters, 47(7), e2020GL087869.

Cusworth, D. H., Jacob, D. J., Varon, D. J., Chan Miller, C., Liu, X., Chance, K., Thorpe, A. K., Duren, R. M., Miller, C. E., Thompson, D. R., & others. (2019). Potential of next-generation imaging spectrometers to detect and quantify methane point sources from space. Atmospheric Measurement Techniques, 12(10), 5655–5668.

Davis, K. J., Deng, A., Lauvaux, T., Miles, N. L., Richardson, S. J., Sarmiento, D. P., Gurney, K. R., Hardesty, R. M., Bonin, T. A., Brewer, W. A., Lamb, B. K., Shepson, P. B., Harvey, R. M., Cambaliza, M. O., Sweeney, C., Turnbull, J. C., Whetstone, J., & Karion, A. (2017). The Indianapolis Flux Experiment (INFLUX): A test-bed for developing urban greenhouse gas emission measurements. Elem Sci Anth, 5(0), 21. https://doi.org/10.1525/elementa.188.

Deziel, N. C., Brokovich, E., Grotto, I., Clark, C. J., Barnett-Itzhaki, Z., Broday, D., & Agay-Shay, K. (2020). Unconventional oil and gas development and health outcomes: A scoping review of the epidemiological research. Environmental Research, 182, 109124. https://doi.org/10.1016/j.envres.2020.109124.

Du, L., & Turner, J. (2015). Using PM2.5 lanthanoid elements and nonparametric wind regression to track petroleum refinery FCC emissions. Science of The Total Environment, 529, 65–71. https://doi.org/10.1016/j.scitotenv.2015.05.034.

Duren, R. M., Thorpe, A. K., Foster, K. T., Rafiq, T., Hopkins, F. M., Yadav, V., Bue, B. D., Thompson, D. R., Conley, S., Colombi, N. K., Frankenberg, C., McCubbin, I. B., Eastwood, M. L., Falk, M., Herner, J. D., Croes, B. E., Green, R. O., & Miller, C. E. (2019). California's methane super-emitters. Nature, 575(7781), 180–184. https://doi.org/10.1038/s41586-019-1720-3.

Elliott, E. G., Trinh, P., Ma, X., Leaderer, B. P., Ward, M. H., & Deziel, N. C. (2017). Unconventional oil and gas development and risk of childhood leukemia: Assessing the evidence. Science of The Total Environment, 576, 138–147. https://doi.org/10.1016/j.scitotenv.2016.10.072.

Englander, J. G., Brandt, A. R., Conley, S., Lyon, D. R., & Jackson, R. B. (2018). Aerial Interyear Comparison and Quantification of Methane Emissions Persistence in the Bakken Formation of North Dakota, USA. Environmental Science & Technology, 52(15), 8947–8953. https://doi.org/10.1021/acs.est.8b01665.

Evans, J. M., & Helmig, D. (2017). Investigation of the influence of transport from oil and natural gas regions on elevated ozone levels in the northern Colorado front range. Journal of the Air & Waste Management Association, 67(2), 196–211. https://doi.org/10.1080/10962247.2016.1226989.

ExxonMobil. (2021). Mitigating Methane Emissions from the Oil and Gas Industry (p. 4). https://corporate.exxonmobil.com/~/media/Global/Files/newsroom/publications-and-reports/Mitigating-Methane-Emissions-from-the-Oil-and-Gas-Industry-Model-Regulatory-Framework.pdf.

Fann, N., Baker, K. R., Chan, E. A. W., Eyth, A., Macpherson, A., Miller, E., & Snyder, J. (2018). Assessing Human Health PM2.5 and Ozone Impacts from U.S. Oil and Natural Gas Sector Emissions in 2025. Environmental Science & Technology, 52(15), 8095–8103. https://doi.org/10.1021/acs.est.8b02050.

Field, R. A., Soltis, J. J., Pérez-Ballesta, P., Grandesso, E., & Montague, D. C. (2015). Distributions of air pollutants associated with oil and natural gas development measured in the Upper Green River Basin of Wyoming. Elementa: Science of the Anthropocene, 3, 000074. https://doi.org/10.12952/journal.elementa.000074.

PSE

Fischer, M. L., Chan, W. R., Delp, W., Jeong, S., Rapp, V., & Zhu, Z. (2018). An Estimate of Natural Gas Methane Emissions from California Homes. Environmental Science & Technology, 52(17), 10205–10213. https://doi.org/10.1021/acs.est.8b03217.

Fischer, M. L., Jeong, S., Faloona, I. C., & Mehrotra, S. (2017). A Survey of Methane Emissions from the California Natural Gas System (CEC-500-2017-033). Lawrence Berkeley National Laboratory. https://eta.lbl.gov/publications/survey-methane-emissions-california.

Floerchinger, C., McKain, K., Bonin, T., Peischl, J., Biraud, S. C., Miller, C., Ryerson, T. B., Wofsy, S. C., & Sweeney, C. (2019). Methane emissions from oil and gas production on the North Slope of Alaska. Atmospheric Environment, 218, 116985. https://doi.org/10.1016/j.atmosenv.2019.116985.

Fox, T. A., Barchyn, T. E., Risk, D., Ravikumar, A. P., & Hugenholtz, C. H. (2019). A review of close-range and screening technologies for mitigating fugitive methane emissions in upstream oil and gas. Environmental Research Letters, 14(5), 053002.

Franco, B., Mahieu, E., Emmons, L. K., Tzompa-Sosa, Z. A., Fischer, E. V., Sudo, K., Bovy, B., Conway, S., Griffin, D., Hannigan, J. W., Strong, K., & Walker, K. A. (2016). Evaluating ethane and methane emissions associated with the development of oil and natural gas extraction in North America. Environmental Research Letters, 11(4), 044010. https://doi.org/10.1088/1748-9326/11/4/044010.

Frankenberg, C., Thorpe, A. K., Thompson, D. R., Hulley, G., Kort, E. A., Vance, N., Borchardt, J., Krings, T., Gerilowski, K., Sweeney, C., Conley, S., Bue, B. D., Aubrey, A. D., Hook, S., & Green, R. O. (2016). Airborne methane remote measurements reveal heavy-tail flux distribution in Four Corners region. Proceedings of the National Academy of Sciences, 113(35), 9734–9739. https://doi.org/10.1073/pnas.1605617113.

Fries, A. E., Schifman, L. A., Shuster, W. D., & Townsend-Small, A. (2018). Street-level emissions of methane and nitrous oxide from the wastewater collection system in Cincinnati, Ohio. Environmental Pollution, 236, 247–256. https://doi.org/10.1016/j.envpol.2018.01.076.

Gallagher, M. E., Down, A., Ackley, R. C., Zhao, K., Phillips, N., & Jackson, R. B. (2015). Natural Gas Pipeline Replacement Programs Reduce Methane Leaks and Improve Consumer Safety. Environmental Science & Technology Letters, 2(10), 286–291. https://doi.org/10.1021/acs.estlett.5b00213.

Garcia-Gonzales, D. A., Popoola, O., Bright, V. B., Paulson, S. E., Wang, Y., Jones, R. L., & Jerrett, M. (2019). Associations among particulate matter, hazardous air pollutants and methane emissions from the Aliso Canyon natural gas storage facility during the 2015 blowout. Environment International, 132, 104855. https://doi.org/10.1016/j.envint.2019.05.049.

Garcia-Gonzales, D. A., Shonkoff, S. B. C., Hays, J., & Jerrett, M. (2019). Hazardous Air Pollutants Associated with Upstream Oil and Natural Gas Development: A Critical Synthesis of Current Peer-Reviewed Literature. Annual Review of Public Health, 40(1), 283–304. https://doi.org/10.1146/annurev-publhealth-040218-043715.

Ghosh, B. (2018). Impact of Changes in Oil and Gas Production Activities on Air Quality in Northeastern Oklahoma: Ambient Air Studies in 2015–2017. Environmental Science & Technology, 52(5), 3285–3294. https://doi.org/10.1021/acs.est.7b05726.

Goetz, J. D., Avery, A., Werden, B., Floerchinger, C., Fortner, E. C., Wormhoudt, J., Massoli, P., Herndon, S. C., Kolb, C. E., Knighton, W. B., Peischl, J., Warneke, C., De Gouw, J. A., Shaw, S. L., & DeCarlo, P. F. (2017). Analysis of local-scale background concentrations of methane and other gas-phase species in the Marcellus Shale. Elem Sci Anth, 5(0), 1. https://doi.org/10.1525/elementa.182.

PSE

Goetz, J. D., Floerchinger, C., Fortner, E. C., Wormhoudt, J., Massoli, P., Knighton, W. B., Herndon, S. C., Kolb, C. E., Knipping, E., Shaw, S. L., & DeCarlo, P. F. (2015). Atmospheric Emission Characterization of Marcellus Shale Natural Gas Development Sites. Environmental Science & Technology, 49(11), 7012–7020. https://doi.org/10.1021/acs.est.5b00452.

Gourdji, S. M., Yadav, V., Karion, A., Mueller, K. L., Conley, S., Ryerson, T., Nehrkorn, T., & Kort, E. A. (2018). Reducing errors in aircraft atmospheric inversion estimates of point-source emissions: The Aliso Canyon natural gas leak as a natural tracer experiment. Environmental Research Letters, 13(4), 045003. https://doi.org/10.1088/1748-9326/aab049.

Guanter, L., Irakulis-Loitxate, I., Gorroño, J., Sánchez-García, E., Cusworth, D., Cogliati, S., & Colombo, R. (2021). Mapping methane point emissions with the PRISMA spaceborne imaging spectrometer.

Gvakharia, A., Kort, E. A., Brandt, A., Peischl, J., Ryerson, T. B., Schwarz, J. P., Smith, M. L., & Sweeney, C. (2017). Methane, Black Carbon, and Ethane Emissions from Natural Gas Flares in the Bakken Shale, North Dakota. Environmental Science & Technology, 51(9), 5317–5325. https://doi.org/10.1021/acs.est.6b05183.

Hajny, K. D., Salmon, O. E., Rudek, J., Lyon, D. R., Stuff, A. A., Stirm, B. H., Kaeser, R., Floerchinger, C. R., Conley, S., Smith, M. L., & Shepson, P. B. (2019). Observations of Methane Emissions from Natural Gas-Fired Power Plants. Environmental Science & Technology, 53(15), 8976–8984. https://doi.org/10.1021/acs.est.9b01875.

Harriss, R., Alvarez, R. A., Lyon, D., Zavala-Araiza, D., Nelson, D., & Hamburg, S. P. (2015). Using Multi-Scale Measurements to Improve Methane Emission Estimates from Oil and Gas Operations in the Barnett Shale Region, Texas. Environmental Science & Technology, 49(13), 7524–7526. https://doi.org/10.1021/acs.est.5b02305.

Heath, G., Warner, E., Steinberg, D., & Brandt, A. (2015). Estimating US Methane Emissions from the Natural Gas Supply Chain. Approaches, Uncertainties, Current Estimates, and Future Studies. National Renewable Energy Lab. (NREL), Golden, CO (United States).

Hecobian, A., Clements, A. L., Shonkwiler, K. B., Zhou, Y., MacDonald, L. P., Hilliard, N., Wells, B. L., Bibeau, B., Ham, J. M., Pierce, J. R., & Collett, J. L. (2019). Air Toxics and Other Volatile Organic Compound Emissions from Unconventional Oil and Gas Development. Environmental Science & Technology Letters, 6(12), 720–726. https://doi.org/10.1021/acs.estlett.9b00591.

Heimburger, A. M. F., Harvey, R. M., Shepson, P. B., Stirm, B. H., Gore, C., Turnbull, J., Cambaliza, M. O. L., Salmon, O. E., Kerlo, A.-E. M., Lavoie, T. N., Davis, K. J., Lauvaux, T., Karion, A., Sweeney, C., Brewer, W. A., Hardesty, R. M., & Gurney, K. R. (2017). Assessing the optimized precision of the aircraft mass balance method for measurement of urban greenhouse gas emission rates through averaging. Elem Sci Anth, 5(0), 26. https://doi.org/10.1525/elementa.134.

Helmig, D., Rossabi, S., Hueber, J., Tans, P., Montzka, S. A., Masarie, K., Thoning, K., Plass-Duelmer, C., Claude, A., Carpenter, L. J., Lewis, A. C., Punjabi, S., Reimann, S., Vollmer, M. K., Steinbrecher, R., Hannigan, J. W., Emmons, L. K., Mahieu, E., Franco, B., … Pozzer, A. (2016). Reversal of global atmospheric ethane and propane trends largely due to US oil and natural gas production. Nature Geoscience, 9(7), 490–495. https://doi.org/10.1038/ngeo2721.

Hendrick, M. F., Ackley, R., Sanaie-Movahed, B., Tang, X., & Phillips, N. G. (2016). Fugitive methane emissions from leak-prone natural gas distribution infrastructure in urban environments. Environmental Pollution, 213, 710–716. https://doi.org/10.1016/j.envpol.2016.01.094.

PSE

Hildenbrand, Z. L., Mach, P. M., McBride, E. M., Dorreyatim, M. N., Taylor, J. T., Carlton, D. D., Meik, J. M., Fontenot, B. E., Wright, K. C., Schug, K. A., & Verbeck, G. F. (2016). Point source attribution of ambient contamination events near unconventional oil and gas development. Science of The Total Environment, 573, 382–388. https://doi.org/10.1016/j.scitotenv.2016.08.118.

Holder, C., Hader, J., Avanasi, R., Hong, T., Carr, E., Mendez, B., Wignall, J., Glen, G., Guelden, B., & Wei, Y. (2019). Evaluating potential human health risks from modeled inhalation exposures to volatile organic compounds emitted from oil and gas operations. Journal of the Air & Waste Management Association, 69(12), 1503–1524. https://doi.org/10.1080/10962247.2019.1680459.

Holliman, J., & Schade, G. W. (2021). Comparing Permitted Emissions to Atmospheric Observations of Hydrocarbons in the Eagle Ford Shale Suggests Permit Violations. Energies, 14(3), 780. https://doi.org/10.3390/en14030780.

Hong, B., & Howarth, R. W. (2016). Greenhouse gas emissions from domestic hot water: Heat pumps compared to most commonly used systems. Energy Science & Engineering, 4(2), 123–133. https://doi.org/10.1002/ese3.112

Hopkins, F. M., Ehleringer, J. R., Bush, S. E., Duren, R. M., Miller, C. E., Lai, C.-T., Hsu, Y.-K., Carranza, V., & Randerson, J. T. (2016). Mitigation of methane emissions in cities: How new measurements and partnerships can contribute to emissions reduction strategies: URBAN METHANE MITIGATION. Earth's Future, 4(9), 408–425. https://doi.org/10.1002/2016EF000381

Howarth, R. W. (2015). Methane emissions and climatic warming risk from hydraulic fracturing and shale gas development: Implications for policy. Energy and Emission Control Technologies, 3, 45–54.

Howarth, R. W., Santoro, R., & Ingraffea, A. (2011). Methane and the greenhouse-gas footprint of natural gas from shale formations: A letter. Climatic Change, 106(4), 679–690. https://doi.org/10.1007/s10584-011-0061-5

Hoyt, D., & Raun, L. H. (2015). Measured and estimated benzene and volatile organic carbon (VOC) emissions at a major U.S. refinery/chemical plant: Comparison and prioritization. Journal of the Air & Waste Management Association, 65(8), 1020–1031. https://doi.org/10.1080/10962247.2015.1058304.

Huang, Y., Kort, E. A., Gourdji, S., Karion, A., Mueller, K., & Ware, J. (2019). Seasonally Resolved Excess Urban Methane Emissions from the Baltimore/Washington, DC Metropolitan Region. Environmental Science & Technology, 53(19), 11285–11293. https://doi.org/10.1021/acs.est.9b02782.

Hurry, J., Risk, D., Lavoie, M., Brooks, B.-G., Phillips, C. L., & Göckede, M. (2016). Atmospheric monitoring and detection of fugitive emissions for Enhanced Oil Recovery. International Journal of Greenhouse Gas Control, 45, 1–8. https://doi.org/10.1016/j.ijggc.2015.11.031.

Ingraffea, A. R., Wawrzynek, P. A., Santoro, R., & Wells, M. (2020). Reported Methane Emissions from Active Oil and Gas Wells in Pennsylvania, 2014–2018. Environmental Science & Technology, 54(9), 5783–5789. https://doi.org/10.1021/acs.est.0c00863.

IPCC (Intergovernmental Panel on Climate Change). (2019). 2018: Global Warming of 1.5°C. An IPCC Special Report on the impacts of global warming of 1.5°C above pre-industrial levels and related global greenhouse gas emission pathways, in the context of strengthening the global response to the threat of climate change, sustainable development, and efforts to eradicate poverty. https://www.ipcc.ch/sr15/download/.

Irakulis-Loitxate, I., Guanter, L., Liu, Y.-N., Varon, D. J., Maasakkers, J. D., Zhang, Y., Chulakadabba, A., Wofsy, S. C., Thorpe, A. K., Duren, R. M., & others. (2021). Satellite-based survey of extreme methane emissions in the Permian basin. Science Advances, 7(27), eabf4507.

PSE

Islam, S. M. N., Jackson, P. L., & Aherne, J. (2016). Ambient nitrogen dioxide and sulfur dioxide concentrations over a region of natural gas production, Northeastern British Columbia, Canada. Atmospheric Environment, 143, 139–151. https://doi.org/10.1016/j.atmosenv.2016.08.017.

Jackson, R. B., Down, A., Phillips, N. G., Ackley, R. C., Cook, C. W., Plata, D. L., & Zhao, K. (2014). Natural Gas Pipeline Leaks Across Washington, DC. Environmental Science & Technology, 48(3), 2051–2058. https://doi.org/10.1021/es404474x.

Jacob, D. J., Turner, A. J., Maasakkers, J. D., Sheng, J., Sun, K., Liu, X., Chance, K., Aben, I., McKeever, J., & Frankenberg, C. (2016). Satellite observations of atmospheric methane and their value for quantifying methane emissions. Atmospheric Chemistry and Physics, 16(22), 14371–14396. https://doi.org/10.5194/acp-16-14371-2016.

Jacobson, M. Z. (2019). Short-Term Impacts of the Aliso Canyon Natural Gas Blowout on Weather, Climate, Air Quality, and Health in California and Los Angeles. Environmental Science & Technology, 53(10), 6081–6093. https://doi.org/10.1021/acs.est.9b01495.

Jakober, C. A., Mara, S. L., Hsu, Y.-K., & Herner, J. D. (2015). Mobile measurements of climate forcing agents: Application to methane emissions from landfill and natural gas compression. Journal of the Air & Waste Management Association, 65(4), 404–412. https://doi.org/10.1080/10962247.2014.996269.

Janardanan, R., Maksyutov, S., Tsuruta, A., Wang, F., Tiwari, Y. K., Valsala, V., Ito, A., Yoshida, Y., Kaiser, J. W., Janssens-Maenhout, G., Arshinov, M., Sasakawa, M., Tohjima, Y., Worthy, D. E. J., Dlugokencky, E. J., Ramonet, M., Arduini, J., Lavric, J. V., Piacentino, S., … Matsunaga, T. (2020). Country-Scale Analysis of Methane Emissions with a High-Resolution Inverse Model Using GOSAT and Surface Observations. Remote Sensing, 12(3), 375. https://doi.org/10.3390/rs12030375.

Jeong, S., Newman, S., Zhang, J., Andrews, A. E., Bianco, L., Bagley, J., Cui, X., Graven, H., Kim, J., Salameh, P., LaFranchi, B. W., Priest, C., Campos-Pineda, M., Novakovskaia, E., Sloop, C. D., Michelsen, H. A., Bambha, R. P., Weiss, R. F., Keeling, R., & Fischer, M. L. (2016). Estimating methane emissions in California's urban and rural regions using multitower observations. Journal of Geophysical Research: Atmospheres, 121(21), 13,031-13,049. https://doi.org/10.1002/2016JD025404.

Johnson, D., Heltzel, R., Nix, A., Darzi, M., & Oliver, D. (2018). Estimated Emissions from the Prime-Movers of Unconventional Natural Gas Well Development Using Recently Collected In-Use Data in the United States. Environmental Science & Technology, 52(9), 5499–5508. https://doi.org/10.1021/acs.est.7b06694.

Johnson, M. R., Tyner, D. R., & Szekeres, A. J. (2021). Blinded evaluation of airborne methane source detection using Bridger Photonics LiDAR. Remote Sensing of Environment, 259, 112418.

Johnson, M. R., Tyner, D. R., Conley, S., Schwietzke, S., & Zavala-Araiza, D. (2017). Comparisons of Airborne Measurements and Inventory Estimates of Methane Emissions in the Alberta Upstream Oil and Gas Sector. Environmental Science & Technology, 51(21), 13008–13017. https://doi.org/10.1021/acs.est.7b03525.

Johnston, J. E., Okorn, K., Van Horne, Y. O., & Jimenez, A. (2021). Changes in neighborhood air quality after idling of an urban oil production site. Environmental Science: Processes & Impacts, 23(7), 967–980. https://doi.org/10.1039/D1EM00048A.

Kang, M., Christian, S., Celia, M. A., Mauzerall, D. L., Bill, M., Miller, A. R., Chen, Y., Conrad, M. E., Darrah, T. H., & Jackson, R. B. (2016). Identification and characterization of high methane-emitting abandoned oil and gas wells. Proceedings of the National Academy of Sciences, 113(48), 13636–13641. https://doi.org/10.1073/pnas.1605913113.

PSE

Kang, M., Kanno, C. M., Reid, M. C., Zhang, X., Mauzerall, D. L., Celia, M. A., Chen, Y., & Onstott, T. C. (2014). Direct measurements of methane emissions from abandoned oil and gas wells in Pennsylvania. Proceedings of the National Academy of Sciences, 111(51), 18173–18177. https://doi.org/10.1073/pnas.1408315111.

Kang, M., Mauzerall, D. L., Ma, D. Z., & Celia, M. A. (2019). Reducing methane emissions from abandoned oil and gas wells: Strategies and costs. Energy Policy, 132, 594–601. https://doi.org/10.1016/j.enpol.2019.05.045.

Karion, A., Sweeney, C., Kort, E. A., Shepson, P. B., Brewer, A., Cambaliza, M., Conley, S. A., Davis, K., Deng, A., Hardesty, M., Herndon, S. C., Lauvaux, T., Lavoie, T., Lyon, D., Newberger, T., Pétron, G., Rella, C., Smith, M., Wolter, S., … Tans, P. (2015). Aircraft-Based Estimate of Total Methane Emissions from the Barnett Shale Region. Environmental Science & Technology, 49(13), 8124–8131. https://doi.org/10.1021/acs.est.5b00217.

Karion, A., Sweeney, C., Pétron, G., Frost, G., Michael Hardesty, R., Kofler, J., Miller, B. R., Newberger, T., Wolter, S., Banta, R., Brewer, A., Dlugokencky, E., Lang, P., Montzka, S. A., Schnell, R., Tans, P., Trainer, M., Zamora, R., & Conley, S. (2013). Methane emissions estimate from airborne measurements over a western United States natural gas field: CH4 EMISSIONS OVER A NATURAL GAS FIELD. Geophysical Research Letters, 40(16), 4393–4397. https://doi.org/10.1002/grl.50811

Kemp, C. E., & Ravikumar, A. P. (2021). New Technologies Can Cost Effectively Reduce Oil and Gas Methane Emissions, but Policies Will Require Careful Design to Establish Mitigation Equivalence. Environmental Science & Technology.

Khalaj, F., & Sattler, M. (2019). Modeling of VOCs and criteria pollutants from multiple natural gas well pads in close proximity, for different terrain conditions: A Barnett Shale case study. Atmospheric Pollution Research, 10(4), 1239–1249. https://doi.org/10.1016/j.apr.2019.02.007

Kille, N., Chiu, R., Frey, M., Hase, F., Sha, M. K., Blumenstock, T., Hannigan, J. W., Orphal, J., Bon, D., & Volkamer, R. (2019). Separation of Methane Emissions From Agricultural and Natural Gas Sources in the Colorado Front Range. Geophysical Research Letters, 46(7), 3990–3998. https://doi.org/10.1029/2019GL082132.

Klemun, M. M., & Trancik, J. E. (2019). Timelines for mitigating the methane impacts of using natural gas for carbon dioxide abatement. Environmental Research Letters, 14(12), 124069. https://doi.org/10.1088/1748-9326/ab2577.

Koch, K., Thomas, S., Arana, E., Roest, G., & Schade, G. W. (2016). On Methane Leaks from Pipelines in Bryan and College Station, Texas, USA. The Open Atmospheric Science Journal, 10(1). https://doi.org/10.2174/1874282301610010056.

Koss, A. R., de Gouw, J., Warneke, C., Gilman, J. B., Lerner, B. M., Graus, M., Yuan, B., Edwards, P., Brown, S. S., Wild, R., Roberts, J. M., Bates, T. S., & Quinn, P. K. (2015). Photochemical aging of volatile organic compounds associated with oil and natural gas extraction in the Uintah Basin, UT, during a wintertime ozone formation event. Atmospheric Chemistry and Physics Discussions, 15(5), 6403–6444. https://doi.org/10.5194/acpd-15-6403-2015.

Lamb, B. K., Cambaliza, M. O. L., Davis, K. J., Edburg, S. L., Ferrara, T. W., Floerchinger, C., Heimburger, A. M. F., Herndon, S., Lauvaux, T., Lavoie, T., Lyon, D. R., Miles, N., Prasad, K. R., Richardson, S., Roscioli, J. R., Salmon, O. E., Shepson, P. B., Stirm, B. H., & Whetstone, J. (2016). Direct and Indirect Measurements and Modeling of Methane Emissions in Indianapolis, Indiana. Environmental Science & Technology, 50(16), 8910–8917. https://doi.org/10.1021/acs.est.6b01198.

Lamb, B. K., Edburg, S. L., Ferrara, T. W., Howard, T., Harrison, M. R., Kolb, C. E., Townsend-Small, A., Dyck, W., Possolo, A., & Whetstone, J. R. (2015). Direct Measurements Show Decreasing Methane Emissions from Natural Gas Local Distribution Systems in the United States. Environmental Science & Technology, 49(8), 5161–5169. https://doi.org/10.1021/es505116p.

PSE

Lan, X., Talbot, R., Laine, P., & Torres, A. (2015). Characterizing Fugitive Methane Emissions in the Barnett Shale Area Using a Mobile Laboratory. Environmental Science & Technology, 49(13), 8139–8146. https://doi.org/10.1021/es5063055.

Lan, X., Tans, P., Sweeney, C., Andrews, A., Dlugokencky, E., Schwietzke, S., Kofler, J., McKain, K., Thoning, K., Crotwell, M., Montzka, S., Miller, B. R., & Biraud, S. C. (2019). Long-Term Measurements Show Little Evidence for Large Increases in Total U.S. Methane Emissions Over the Past Decade. Geophysical Research Letters, 46(9), 4991–4999. https://doi.org/10.1029/2018GL081731.

Lauvaux, T., Giron, C., Mazzolini, M., d'Aspremont, A., Duren, R., Cusworth, D., Shindell, D., & Ciais, P. (2021). Global Assessment of Oil and Gas Methane Ultra-Emitters. ArXiv Preprint ArXiv:2105.06387.

Lavoie, T. N., Shepson, P. B., Cambaliza, M. O. L., Stirm, B. H., Conley, S., Mehrotra, S., Faloona, I. C., & Lyon, D. (2017a). Spatiotemporal Variability of Methane Emissions at Oil and Natural Gas Operations in the Eagle Ford Basin. Environmental Science & Technology, 51(14), 8001–8009. https://doi.org/10.1021/acs.est.7b00814.

Lavoie, T. N., Shepson, P. B., Cambaliza, M. O. L., Stirm, B. H., Karion, A., Sweeney, C., Yacovitch, T. I., Herndon, S. C., Lan, X., & Lyon, D. (2015). Aircraft-Based Measurements of Point Source Methane Emissions in the Barnett Shale Basin. Environmental Science & Technology, 49(13), 7904–7913. https://doi.org/10.1021/acs.est.5b00410.

Lavoie, T. N., Shepson, P. B., Gore, C. A., Stirm, B. H., Kaeser, R., Wulle, B., Lyon, D., & Rudek, J. (2017b). Assessing the Methane Emissions from Natural Gas-Fired Power Plants and Oil Refineries. Environmental Science & Technology, 51(6), 3373–3381. https://doi.org/10.1021/acs.est.6b05531.

Lebel, E. D., Lu, H. S., Speizer, S. A., Finnegan, C. J., & Jackson, R. B. (2020). Quantifying Methane Emissions from Natural Gas Water Heaters. Environmental Science & Technology, 54(9), 5737–5745. https://doi.org/10.1021/acs.est.9b07189.

Lim, G. Q., Matin, M., & John, K. (2019). Spatial and temporal characteristics of ambient atmospheric hydrocarbons in an active shale gas region in North Texas. Science of The Total Environment, 656, 347–363. https://doi.org/10.1016/j.scitotenv.2018.11.313.

Lindaas, J., Farmer, D. K., Pollack, I. B., Abeleira, A., Flocke, F., & Fischer, E. V. (2019). Acyl Peroxy Nitrates Link Oil and Natural Gas Emissions to High Ozone Abundances in the Colorado Front Range During Summer 2015. Journal of Geophysical Research: Atmospheres, 124(4), 2336–2350. https://doi.org/10.1029/2018JD028825.

Long, C. M., Briggs, N. L., & Bamgbose, I. A. (2019). Synthesis and health-based evaluation of ambient air monitoring data for the Marcellus Shale region. Journal of the Air & Waste Management Association, 69(5), 527–547. https://doi.org/10.1080/10962247.2019.1572551.

Long, C. M., Briggs, N. L., Cochran, B. A., & Mims, D. M. (2021). Health-based evaluation of ambient air measurements of PM2.5 and volatile organic compounds near a Marcellus Shale unconventional natural gas well pad site and a school campus. Journal of Exposure Science & Environmental Epidemiology, 31(4), 614–627. https://doi.org/10.1038/s41370-021-00298-5.

Long, J. C. S., Birkholzer, J. T., Mace, A. J., Brady, S. E., Arthur, J. D., Duren, R. M., Edson, K., Jackson, R. B., Jerrett, M. L. B., Meshkati, N., Perfect, S. A., Thorn, T., Traina, S. J., Wara, M. W., Elder, C. M., Greenblatt, J. B., & Oldenburg, C. M. (2018). Long-Term Viability of Underground Natural Gas Storage in California: An Independent Review of Scientific and Technical Information. California Council on Science and Technology. https://ccst.us/wp-content/uploads/Full-Technical-Report-v2_max.pdf.

PSE

Lopez-Coto, I., Ren, X., Salmon, O. E., Karion, A., Shepson, P. B., Dickerson, R. R., Stein, A., Prasad, K., & Whetstone, J. R. (2020). Wintertime CO2, CH4, and CO Emissions Estimation for the Washington, DC–Baltimore Metropolitan Area Using an Inverse Modeling Technique. Environmental Science & Technology, 54(5), 2606–2614. https://doi.org/10.1021/acs.est.9b06619.

Luck, B., Zimmerle, D., Vaughn, T., Lauderdale, T., Keen, K., Harrison, M., Marchese, A., Williams, L., & Allen, D. (2019). Multiday Measurements of Pneumatic Controller Emissions Reveal the Frequency of Abnormal Emissions Behavior at Natural Gas Gathering Stations. Environmental Science & Technology Letters, 6(6), 348–352. https://doi.org/10.1021/acs.estlett.9b00158.

Lyman, S. N., Tran, T., Mansfield, M. L., & Ravikumar, A. P. (2019). Aerial and ground-based optical gas imaging survey of Uinta Basin oil and gas wells. Elem Sci Anth, 7(1), 43. https://doi.org/10.1525/elementa.381.

Lyon, D. R., Alvarez, R. A., Zavala-Araiza, D., Brandt, A. R., Jackson, R. B., & Hamburg, S. P. (2016). Aerial Surveys of Elevated Hydrocarbon Emissions from Oil and Gas Production Sites. Environmental Science & Technology, 50(9), 4877–4886. https://doi.org/10.1021/acs.est.6b00705.

Lyon, D. R., Hmiel, B., Gautam, R., Omara, M., Roberts, K. A., Barkley, Z. R., Davis, K. J., Miles, N. L., Monteiro, V. C., Richardson, S. J., Conley, S., Smith, M. L., Jacob, D. J., Shen, L., Varon, D. J., Deng, A., Rudelis, X., Sharma, N., Story, K. T., … Hamburg, S. P. (2021). Concurrent variation in oil and gas methane emissions and oil price during the COVID-19 pandemic. Atmospheric Chemistry and Physics, 21(9), 6605–6626. https://doi.org/10.5194/acp-21-6605-2021.

Lyon, D. R., Zavala-Araiza, D., Alvarez, R. A., Harriss, R., Palacios, V., Lan, X., Talbot, R., Lavoie, T., Shepson, P., Yacovitch, T. I., Herndon, S. C., Marchese, A. J., Zimmerle, D., Robinson, A. L., & Hamburg, S. P. (2015). Constructing a Spatially Resolved Methane Emission Inventory for the Barnett Shale Region. Environmental Science & Technology, 49(13), 8147–8157. https://doi.org/10.1021/es506359c.

Maasakkers, J. D., Jacob, D. J., Sulprizio, M. P., Scarpelli, T. R., Nesser, H., Sheng, J.-X., Zhang, Y., Hersher, M., Bloom, A. A., Bowman, K. W., Worden, J. R., Janssens-Maenhout, G., & Parker, R. J. (2019). Global distribution of methane emissions, emission trends, and OH concentrations and trends inferred from an inversion of GOSAT satellite data for 2010–2015. Atmospheric Chemistry and Physics, 19(11), 7859–7881. https://doi.org/10.5194/acp-19-7859-2019.

Maasakkers, J. D., Jacob, D. J., Sulprizio, M. P., Turner, A. J., Weitz, M., Wirth, T., Hight, C., DeFigueiredo, M., Desai, M., Schmeltz, R., Hockstad, L., Bloom, A. A., Bowman, K. W., Jeong, S., & Fischer, M. L. (2016). Gridded National Inventory of U.S. Methane Emissions. Environmental Science & Technology, 50(23), 13123–13133. https://doi.org/10.1021/acs.est.6b02878.

MacKay, K., Lavoie, M., Bourlon, E., Atherton, E., O'Connell, E., Baillie, J., Fougère, C., & Risk, D. (2021). Methane emissions from upstream oil and gas production in Canada are underestimated. Scientific Reports, 11(1), 1–8.

MacKay, K., Risk, D., Atherton, E., Fougére, C., Bourlon, E., O'Connell, E., & Baillie, J. (2019). Fugitive and vented methane emissions surveying on the Weyburn CO2-EOR field in southeastern Saskatchewan, Canada. International Journal of Greenhouse Gas Control, 88, 118–123. https://doi.org/10.1016/j.ijggc.2019.05.032.

Majid, A., Val Martin, M., Lamsal, L. N., & Duncan, B. N. (2017). A decade of changes in nitrogen oxides over regions of oil and natural gas activity in the United States. Elem Sci Anth, 5(0), 76. https://doi.org/10.1525/elementa.259.

Marchese, A. J., Vaughn, T. L., Zimmerle, D. J., Martinez, D. M., Williams, L. L., Robinson, A. L., Mitchell, A. L., Subramanian, R., Tkacik, D. S., Roscioli, J. R., & Herndon, S. C. (2015). Methane Emissions from United States Natural Gas Gathering and Processing. Environmental Science & Technology, 49(17), 10718–10727. https://doi.org/10.1021/acs.est.5b02275.

Marrero, J. E., Townsend-Small, A., Lyon, D. R., Tsai, T. R., Meinardi, S., & Blake, D. R. (2016). Estimating Emissions of Toxic Hydrocarbons from Natural Gas Production Sites in the Barnett Shale Region of Northern Texas. Environmental Science & Technology, 50(19), 10756–10764. https://doi.org/10.1021/acs.est.6b02827.

Maskrey, J. R., Insley, A. L., Hynds, E. S., & Panko, J. M. (2016). Air monitoring of volatile organic compounds at relevant receptors during hydraulic fracturing operations in Washington County, Pennsylvania. Environmental Monitoring and Assessment, 188(7), 410. https://doi.org/10.1007/s10661-016-5410-4.

McCawley, M. (2015). Air Contaminants Associated with Potential Respiratory Effects from Unconventional Resource Development Activities. Seminars in Respiratory and Critical Care Medicine, 36(03), 379–387. https://doi.org/10.1055/s-0035-1549453.

McDuffie, E. E., Edwards, P. M., Gilman, J. B., Lerner, B. M., Dubé, W. P., Trainer, M., Wolfe, D. E., Angevine, W. M., deGouw, J., Williams, E. J., Tevlin, A. G., Murphy, J. G., Fischer, E. V., McKeen, S., Ryerson, T. B., Peischl, J., Holloway, J. S., Aikin, K., Langford, A. O., … Brown, S. S. (2016). Influence of oil and gas emissions on summertime ozone in the Colorado Northern Front Range: COLORADO O&G AND SUMMERTIME OZONE. Journal of Geophysical Research: Atmospheres, 121(14), 8712–8729. https://doi.org/10.1002/2016JD025265.

McKain, K., Down, A., Raciti, S. M., Budney, J., Hutyra, L. R., Floerchinger, C., Herndon, S. C., Nehrkorn, T., Zahniser, M. S., Jackson, R. B., Phillips, N., & Wofsy, S. C. (2015). Methane emissions from natural gas infrastructure and use in the urban region of Boston, Massachusetts. Proceedings of the National Academy of Sciences, 112(7), 1941–1946. https://doi.org/10.1073/pnas.1416261112.

McKenzie, L. M., Blair, B., Hughes, J., Allshouse, W. B., Blake, N. J., Helmig, D., Milmoe, P., Halliday, H., Blake, D. R., & Adgate, J. L. (2018). Ambient Nonmethane Hydrocarbon Levels Along Colorado's Northern Front Range: Acute and Chronic Health Risks. Environmental Science & Technology, 52(8), 4514–4525. https://doi.org/10.1021/acs.est.7b05983.

McMullin, T., Bamber, A., Bon, D., Vigil, D., & Van Dyke, M. (2018). Exposures and Health Risks from Volatile Organic Compounds in Communities Located near Oil and Gas Exploration and Production Activities in Colorado (U.S.A.). International Journal of Environmental Research and Public Health, 15(7), 1500. https://doi.org/10.3390/ijerph15071500.

Merrin, Z., & Francisco, P. W. (2019). Unburned Methane Emissions from Residential Natural Gas Appliances. Environmental Science & Technology, 53(9), 5473–5482. https://doi.org/10.1021/acs.est.8b05323.

Michanowicz, D. R., Buonocore, J. J., Rowland, S. T., Konschnik, K. E., Goho, S. A., & Bernstein, A. S. (2017). A national assessment of underground natural gas storage: Identifying wells with designs likely vulnerable to a single-point-of-failure. Environmental Research Letters, 12(6), 064004. https://doi.org/10.1088/1748-9326/aa7030.

Milford, J. (2015). Out in Front: State and Federal Regulation of Air Pollution Emissions from Oil and Gas Production Activities in the Western United States. Natural Resources, 55(1). https://digitalrepository.unm.edu/nrj/vol55/iss1/3.

PSE

Mitchell, A. L., Griffin, W. M., & Casman, E. A. (2016). Lung Cancer Risk from Radon in Marcellus Shale Gas in Northeast U.S. Homes: Radon-Related Lung Cancer Risk from Marcellus Shale Natural Gas. Risk Analysis, 36(11), 2105–2119. https://doi.org/10.1111/risa.12570.

Mitchell, A. L., Tkacik, D. S., Roscioli, J. R., Herndon, S. C., Yacovitch, T. I., Martinez, D. M., Vaughn, T. L., Williams, L. L., Sullivan, M. R., Floerchinger, C., Omara, M., Subramanian, R., Zimmerle, D., Marchese, A. J., & Robinson, A. L. (2015). Measurements of Methane Emissions from Natural Gas Gathering Facilities and Processing Plants: Measurement Results. Environmental Science & Technology, 49(5), 3219–3227. https://doi.org/10.1021/es5052809.

Moher, D., Shamseer, L., Clarke, M., Ghersi, D., Liberati, A., Petticrew, M., Shekelle, P., & Stewart, L. A. (2015). Preferred reporting items for systematic review and meta-analysis protocols (PRISMA-P) 2015 statement. Systematic Reviews, 4(1), 1–9.

Mullen, N. A., Li, J., Russell, M. L., Spears, M., Less, B. D., & Singer, B. C. (2016). Results of the California Healthy Homes Indoor Air Quality Study of 2011-2013: Impact of natural gas appliances on air pollutant concentrations. Indoor Air, 26(2), 231–245. https://doi.org/10.1111/ina.12190.

Nathan, B. J., Golston, L. M., O'Brien, A. S., Ross, K., Harrison, W. A., Tao, L., Lary, D. J., Johnson, D. R., Covington, A. N., Clark, N. N., & Zondlo, M. A. (2015). Near-Field Characterization of Methane Emission Variability from a Compressor Station Using a Model Aircraft. Environmental Science & Technology, 49(13), 7896–7903. https://doi.org/10.1021/acs.est.5b00705.

National Academies of Sciences, Engineering, and Medicine. (2018). Improving Characterization of Anthropogenic Methane Emissions in the United States. The National Academies Press. https://www.nap.edu/catalog/24987/improving-characterization-of-anthropogenic-methane-emissions-in-the-united-states.

Nsanzineza, R., Capps, S. L., & Milford, J. B. (2019). Modeling Emissions and Ozone Air Quality Impacts of Future Scenarios for Energy and Power Production in the Rocky Mountain States. Environmental Science & Technology, 53(13), 7893–7902. https://doi.org/10.1021/acs.est.9b00356.

Nye, M., Knuckles, T., Yan, B., Ross, J., Orem, W., Varonka, M., Thurston, G., Dzomba, A., & McCawley, M. (2020). Use of Tracer Elements for Estimating Community Exposure to Marcellus Shale Development Operations. International Journal of Environmental Research and Public Health, 17(6), 1837. https://doi.org/10.3390/ijerph17061837.

O'Connell, E., Risk, D., Atherton, E., Bourlon, E., Fougère, C., Baillie, J., Lowry, D., & Johnson, J. (2019). Methane emissions from contrasting production regions within Alberta, Canada: Implications under incoming federal methane regulations. Elem Sci Anth, 7(1), 3. https://doi.org/10.1525/elementa.341.

Okorn, K., Jimenez, A., Collier-Oxandale, A., Johnston, J., & Hannigan, M. (2021). Characterizing methane and total non-methane hydrocarbon levels in Los Angeles communities with oil and gas facilities using air quality monitors. Science of The Total Environment, 777, 146194. https://doi.org/10.1016/j.scitotenv.2021.146194.

Olaguer, E. P., Erickson, M., Wijesinghe, A., Neish, B., Williams, J., & Colvin, J. (2016). Updated methods for assessing the impacts of nearby gas drilling and production on neighborhood air quality and human health. Journal of the Air & Waste Management Association, 66(2), 173–183. https://doi.org/10.1080/10962247.2015.1083914.

Oltmans, S. J., Karion, A., Schnell, R. C., Pétron, G., Helmig, D., Montzka, S. A., Wolter, S., Neff, D., Miller, B. R., Hueber, J., Conley, S., Johnson, B. J., & Sweeney, C. (2016). O3, CH4, CO2, CO, NO2 and NMHC aircraft measurements in the Uinta Basin oil and gas region under low and high ozone conditions in winter 2012 and 2013. Elementa: Science of the Anthropocene, 4, 000132. https://doi.org/10.12952/journal.elementa.000132.

PSE

Omara, M., Sullivan, M. R., Li, X., Subramanian, R., Robinson, A. L., & Presto, A. A. (2016). Methane Emissions from Conventional and Unconventional Natural Gas Production Sites in the Marcellus Shale Basin. Environmental Science & Technology, 50(4), 2099–2107. https://doi.org/10.1021/acs.est.5b05503.

Omara, M., Zimmerman, N., Sullivan, M. R., Li, X., Ellis, A., Cesa, R., Subramanian, R., Presto, A. A., & Robinson, A. L. (2018). Methane Emissions from Natural Gas Production Sites in the United States: Data Synthesis and National Estimate. Environmental Science & Technology, 52(21), 12915–12925. https://doi.org/10.1021/acs.est.8b03535.

Orak, N. H., Reeder, M., & Pekney, N. J. (2021). Identifying and quantifying source contributions of air quality contaminants during unconventional shale gas extraction. Atmospheric Chemistry and Physics, 21(6), 4729–4739. https://doi.org/10.5194/acp-21-4729-2021.

Pacsi, A. P., Ferrara, T., Schwan, K., Tupper, P., Lev-On, M., Smith, R., & Ritter, K. (2019). Equipment leak detection and quantification at 67 oil and gas sites in the Western United States. Elem Sci Anth, 7(1), 29. https://doi.org/10.1525/elementa.368.

Pandey, S., Gautam, R., Houweling, S., van der Gon, H. D., Sadavarte, P., Borsdorff, T., Hasekamp, O., Landgraf, J., Tol, P., van Kempen, T., Hoogeveen, R., van Hees, R., Hamburg, S. P., Maasakkers, J. D., & Aben, I. (2019). Satellite observations reveal extreme methane leakage from a natural gas well blowout. Proceedings of the National Academy of Sciences, 116(52), 26376–26381. https://doi.org/10.1073/pnas.1908712116.

Paulik, L. B., Donald, C. E., Smith, B. W., Tidwell, L. G., Hobbie, K. A., Kincl, L., Haynes, E. N., & Anderson, K. A. (2016). Emissions of Polycyclic Aromatic Hydrocarbons from Natural Gas Extraction into Air. Environmental Science & Technology, 50(14), 7921–7929. https://doi.org/10.1021/acs.est.6b02762.

Paulik, L. B., Hobbie, K. A., Rohlman, D., Smith, B. W., Scott, R. P., Kincl, L., Haynes, E. N., & Anderson, K. A. (2018). Environmental and individual PAH exposures near rural natural gas extraction. Environmental Pollution, 241, 397–405. https://doi.org/10.1016/j.envpol.2018.05.010.

Payne and Ackley. (2015). A proposed rapid method for measuring area methane emissions: An exploratory application in Manhattan, New York, USA.

Peischl, J., Eilerman, S. J., Neuman, J. A., Aikin, K. C., Gouw, J. de, Gilman, J. B., Herndon, S. C., Nadkarni, R., Trainer, M., Warneke, C., & Ryerson, T. B. (2018). Quantifying Methane and Ethane Emissions to the Atmosphere From Central and Western U.S. Oil and Natural Gas Production Regions. Journal of Geophysical Research: Atmospheres, 123(14), 7725–7740. https://doi.org/10.1029/2018JD028622.

Peischl, J., Karion, A., Sweeney, C., Kort, E. A., Smith, M. L., Brandt, A. R., Yeskoo, T., Aikin, K. C., Conley, S. A., Gvakharia, A., Trainer, M., Wolter, S., & Ryerson, T. B. (2016). Quantifying atmospheric methane emissions from oil and natural gas production in the Bakken shale region of North Dakota: CH4 Emissions From the Bakken. Journal of Geophysical Research: Atmospheres, 121(10), 6101–6111. https://doi.org/10.1002/2015JD024631.

Peischl, J., Ryerson, T. B., Aikin, K. C., de Gouw, J. A., Gilman, J. B., Holloway, J. S., Lerner, B. M., Nadkarni, R., Neuman, J. A., Nowak, J. B., Trainer, M., Warneke, C., & Parrish, D. D. (2015). Quantifying atmospheric methane emissions from the Haynesville, Fayetteville, and northeastern Marcellus shale gas production regions: CH4 emissions from shale gas production. Journal of Geophysical Research: Atmospheres, 120(5), 2119–2139. https://doi.org/10.1002/2014JD022697.

PSE

Peischl, J., Ryerson, T. B., Brioude, J., Aikin, K. C., Andrews, A. E., Atlas, E., Blake, D., Daube, B. C., de Gouw, J. A., Dlugokencky, E., Frost, G. J., Gentner, D. R., Gilman, J. B., Goldstein, A. H., Harley, R. A., Holloway, J. S., Kofler, J., Kuster, W. C., Lang, P. M., … Parrish, D. D. (2013). Quantifying sources of methane using light alkanes in the Los Angeles basin, California: SOURCES OF METHANE IN L.A. Journal of Geophysical Research: Atmospheres, 118(10), 4974–4990. https://doi.org/10.1002/jgrd.50413.

Pekney, N. J., Diehl, J. R., Ruehl, D., Sams, J., Veloski, G., Patel, A., Schmidt, C., & Card, T. (2018). Measurement of methane emissions from abandoned oil and gas wells in Hillman State Park, Pennsylvania. Carbon Management, 9(2), 165–175. https://doi.org/10.1080/17583004.2018.1443642.

Pétron, G., Frost, G., Miller, B. R., Hirsch, A. I., Montzka, S. A., Karion, A., Trainer, M., Sweeney, C., Andrews, A. E., Miller, L., Kofler, J., Bar-Ilan, A., Dlugokencky, E. J., Patrick, L., Moore, C. T., Ryerson, T. B., Siso, C., Kolodzey, W., Lang, P. M., … Tans, P. (2012). Hydrocarbon emissions characterization in the Colorado Front Range: A pilot study. Journal of Geophysical Research: Atmospheres, 117(D4). https://doi.org/10.1029/2011JD016360.

Pfister, G., Wang, C., Barth, M., Flocke, F., Vizuete, W., & Walters, S. (2019). Chemical Characteristics and Ozone Production in the Northern Colorado Front Range. Journal of Geophysical Research: Atmospheres, 124(23), 13397–13419. https://doi.org/10.1029/2019JD030544.

Phillips, N. G., Ackley, R., Crosson, E. R., Down, A., Hutyra, L. R., Brondfield, M., Karr, J. D., Zhao, K., & Jackson, R. B. (2013). Mapping urban pipeline leaks: Methane leaks across Boston. Environmental Pollution, 173, 1–4. https://doi.org/10.1016/j.envpol.2012.11.003.

Plant, G., Kort, E. A., Floerchinger, C., Gvakharia, A., Vimont, I., & Sweeney, C. (2019). Large Fugitive Methane Emissions From Urban Centers Along the U.S. East Coast. Geophysical Research Letters, 46(14), 8500–8507. https://doi.org/10.1029/2019GL082635.

Prenni, A. J., Day, D. E., Evanoski-Cole, A. R., Sive, B. C., Hecobian, A., Zhou, Y., Gebhart, K. A., Hand, J. L., Sullivan, A. P., Li, Y., Schurman, M. I., Desyaterik, Y., Malm, W. C., Collett Jr., J. L., & Schichtel, B. A. (2016). Oil and gas impacts on air quality in federal lands in the Bakken region: An overview of the Bakken Air Quality Study and first results. Atmospheric Chemistry and Physics, 16(3), 1401–1416. https://doi.org/10.5194/acp-16-1401-2016.

PSE Healthy Energy (Physicians, Scientists and Engineers for Healthy Energy). (2021). Repository for Oil and Gas Energy Research (ROGER). https://www.psehealthyenergy.org/our-work/shale-gas-research-library/.

Ravikumar, A. P., & Brandt, A. R. (2017). Designing better methane mitigation policies: The challenge of distributed small sources in the natural gas sector. Environmental Research Letters, 12(4), 044023. https://doi.org/10.1088/1748-9326/aa6791.

Ravikumar, A. P., Roda-Stuart, D., Liu, R., Bradley, A., Bergerson, J., Nie, Y., Zhang, S., Bi, X., & Brandt, A. R. (2020). Repeated leak detection and repair surveys reduce methane emissions over scale of years. Environmental Research Letters, 15(3), 034029. https://doi.org/10.1088/1748-9326/ab6ae1.

Ravikumar, A. P., Sreedhara, S., Wang, J., Englander, J., Roda-Stuart, D., Bell, C., Zimmerle, D., Lyon, D., Mogstad, I., Ratner, B., & others. (2019). Single-blind inter-comparison of methane detection technologies–results from the Stanford/EDF Mobile Monitoring Challenge. Elementa: Science of the Anthropocene, 7.

Rella, C. W., Tsai, T. R., Botkin, C. G., Crosson, E. R., & Steele, D. (2015). Measuring Emissions from Oil and Natural Gas Well Pads Using the Mobile Flux Plane Technique. Environmental Science & Technology, 49(7), 4742–4748. https://doi.org/10.1021/acs.est.5b00099.

PSE

Ren, X., Hall, D. L., Vinciguerra, T., Benish, S. E., Stratton, P. R., Ahn, D., Hansford, J. R., Cohen, M. D., Sahu, S., He, H., Grimes, C., Fuentes, J. D., Shepson, P. B., Salawitch, R. J., Ehrman, S. H., & Dickerson, R. R. (2019). Methane Emissions from the Marcellus Shale in Southwestern Pennsylvania and Northern West Virginia Based on Airborne Measurements. Journal of Geophysical Research: Atmospheres, 124(3), 1862–1878. https://doi.org/10.1029/2018JD029690.

Ren, X., Salmon, O. E., Hansford, J. R., Ahn, D., Hall, D., Benish, S. E., Stratton, P. R., He, H., Sahu, S., Grimes, C., Heimburger, A. M. F., Martin, C. R., Cohen, M. D., Stunder, B., Salawitch, R. J., Ehrman, S. H., Shepson, P. B., & Dickerson, R. R. (2018). Methane Emissions From the Baltimore-Washington Area Based on Airborne Observations: Comparison to Emissions Inventories. Journal of Geophysical Research: Atmospheres, 123(16), 8869–8882. https://doi.org/10.1029/2018JD028851.

Rich, A. L., & Orimoloye, H. T. (2016). Elevated Atmospheric Levels of Benzene and Benzene-Related Compounds from Unconventional Shale Extraction and Processing: Human Health Concern for Residential Communities. Environmental Health Insights, 10, EHI.S33314. https://doi.org/10.4137/EHI.S33314.

Richards, J., & Brozell, T. (2015). Assessment of Community Exposure to Ambient Respirable Crystalline Silica near Frac Sand Processing Facilities. Atmosphere, 6(8), 960–982. https://doi.org/10.3390/atmos6080960.

Riddick, S. N., Mauzerall, D. L., Celia, M. A., Kang, M., & Bandilla, K. (2020). Variability observed over time in methane emissions from abandoned oil and gas wells. International Journal of Greenhouse Gas Control, 100, 103116. https://doi.org/10.1016/j.ijggc.2020.103116.

Riddick, S. N., Mauzerall, D. L., Celia, M. A., Kang, M., Bressler, K., Chu, C., & Gum, C. D. (2019). Measuring methane emissions from abandoned and active oil and gas wells in West Virginia. Science of The Total Environment, 651, 1849–1856. https://doi.org/10.1016/j.scitotenv.2018.10.082.

Robertson, A. M., Edie, R., Field, R. A., Lyon, D., McVay, R., Omara, M., Zavala-Araiza, D., & Murphy, S. M. (2020). New Mexico Permian Basin measured well pad methane emissions are a factor of 5–9 times higher than US EPA estimates. Environmental Science & Technology, 54(21), 13926–13934.

Robertson, A. M., Edie, R., Snare, D., Soltis, J., Field, R. A., Burkhart, M. D., Bell, C. S., Zimmerle, D., & Murphy, S. M. (2017). Variation in Methane Emission Rates from Well Pads in Four Oil and Gas Basins with Contrasting Production Volumes and Compositions. Environmental Science & Technology, 51(15), 8832–8840. https://doi.org/10.1021/acs.est.7b00571.

Roest, G. S., & Schade, G. W. (2020). Air quality measurements in the western Eagle Ford Shale. Elem Sci Anth, 8(1), 18. https://doi.org/10.1525/elementa.414.

Roest, G., & Schade, G. (2017). Quantifying alkane emissions in the Eagle Ford Shale using boundary layer enhancement. Atmospheric Chemistry and Physics, 17(18), 11163–11176. https://doi.org/10.5194/acp-17-11163-2017.

Roohani, Y. H., Roy, A. A., Heo, J., Robinson, A. L., & Adams, P. J. (2017). Impact of natural gas development in the Marcellus and Utica shales on regional ozone and fine particulate matter levels. Atmospheric Environment, 155, 11–20. https://doi.org/10.1016/j.atmosenv.2017.01.001.

Roscioli, J. R., Yacovitch, T. I., Floerchinger, C., Mitchell, A. L., Tkacik, D. S., Subramanian, R., Martinez, D. M., Vaughn, T. L., Williams, L., Zimmerle, D., Robinson, A. L., Herndon, S. C., & Marchese, A. J. (2015). Measurements of methane emissions from natural gas gathering facilities and processing plants: Measurement methods. Atmospheric Measurement Techniques, 8(5), 2017–2035. https://doi.org/10.5194/amt-8-2017-2015.

PSE

Russell, S. J., Vines, C. D., Bohrer, G., Johnson, D. R., Villa, J. A., Heltzel, R., Rey-Sanchez, C., & Matthes, J. H. (2020). Quantifying CH4 concentration spikes above baseline and attributing CH4 sources to hydraulic fracturing activities by continuous monitoring at an off-site tower. Atmospheric Environment, 228, 117452. https://doi.org/10.1016/j.atmosenv.2020.117452.

Russo, P. N., & Carpenter, D. O. (2019). Air Emissions from Natural Gas Facilities in New York State. International Journal of Environmental Research and Public Health, 16(9), 1591. https://doi.org/10.3390/ijerph16091591.

Rutherford, J. S., Sherwin, E. D., Ravikumar, A. P., Heath, G. A., Englander, J., Cooley, D., Lyon, D., Omara, M., Langfitt, Q., & Brandt, A. R. (2021). Closing the methane gap in US oil and natural gas production emissions inventories. Nature Communications, 12(1), 1–12.

Sablan, O. M., Schade, G. W., & Holliman, J. (2020). Passively Sampled Ambient Hydrocarbon Abundances in a Texas Oil Patch. Atmosphere, 11(3), 241. https://doi.org/10.3390/atmos11030241.

Saint-Vincent, P. M. B., & Pekney, N. J. (2019). Beyond-the-Meter: Unaccounted Sources of Methane Emissions in the Natural Gas Distribution Sector. Environmental Science & Technology, acs.est.9b04657. https://doi.org/10.1021/acs.est.9b04657.

Sanchez, N. P., Saffari, A., Barczyk, S., Coleman, B. K., Naufal, Z., Rabideau, C., & Pacsi, A. P. (2019). Results of Three Years of Ambient Air Monitoring Near a Petroleum Refinery in Richmond, California, USA. Atmosphere, 10(7), 385. https://doi.org/10.3390/atmos10070385.

Sanchez, N. P., Zheng, C., Ye, W., Czader, B., Cohan, D. S., Tittel, F. K., & Griffin, R. J. (2018). Exploratory study of atmospheric methane enhancements derived from natural gas use in the Houston urban area. Atmospheric Environment, 176, 261–273. https://doi.org/10.1016/j.atmosenv.2018.01.001.

Sanchez, N., & Mays, D. C. (2015). Effect of methane leakage on the greenhouse gas footprint of electricity generation. Climatic Change, 133(2), 169–178. https://doi.org/10.1007/s10584-015-1471-6.

Scarpelli, T. R., Jacob, D. J., Villasana, C. A. O., Hernández, I. F. R., Moreno, P. R. C., Alfaro, E. A. C., García, M. Á. G., & Zavala-Araiza, D. (2020). A gridded inventory of anthropogenic methane emissions from Mexico based on Mexico's national inventory of greenhouse gases and compounds. Environmental Research Letters, 15(10), 105015.

Schade, G. W., & Roest, G. (2016). Analysis of non-methane hydrocarbon data from a monitoring station affected by oil and gas development in the Eagle Ford shale, Texas. Elementa: Science of the Anthropocene, 4, 000096. https://doi.org/10.12952/journal.elementa.000096.

Schade, G. W., & Roest, G. (2018). Source apportionment of non-methane hydrocarbons, NOx and H2S data from a central monitoring station in the Eagle Ford shale, Texas. Elem Sci Anth, 6(1), 35. https://doi.org/10.1525/elementa.289.

Schneising, O., Burrows, J. P., Dickerson, R. R., Buchwitz, M., Reuter, M., & Bovensmann, H. (2014). Remote sensing of fugitive methane emissions from oil and gas production in North American tight geologic formations: Remote sensing of fugitive methane emissions from oil and gas production. Earth's Future, 2(10), 548–558. https://doi.org/10.1002/2014EF000265.

Schollaert, C., Ackley, R. C., DeSantis, A., Polka, E., & Scammell, M. K. (2020). Natural gas leaks and tree death: A first-look case-control study of urban trees in Chelsea, MA USA. Environmental Pollution, 263, 114464. https://doi.org/10.1016/j.envpol.2020.114464.

PSE

Schwietzke, S., Harrison, M., Lauderdale, T., Branson, K., Conley, S., George, F. C., Jordan, D., Jersey, G. R., Zhang, C., Mairs, H. L., Pétron, G., & Schnell, R. C. (2019). Aerially guided leak detection and repair: A pilot field study for evaluating the potential of methane emission detection and cost-effectiveness. Journal of the Air & Waste Management Association, 69(1), 71–88. https://doi.org/10.1080/10962247.2018.1515123.

Schwietzke, S., Pétron, G., Conley, S., Pickering, C., Mielke-Maday, I., Dlugokencky, E. J., Tans, P. P., Vaughn, T., Bell, C., Zimmerle, D., Wolter, S., King, C. W., White, A. B., Coleman, T., Bianco, L., & Schnell, R. C. (2017). Improved Mechanistic Understanding of Natural Gas Methane Emissions from Spatially Resolved Aircraft Measurements. Environmental Science & Technology, 51(12), 7286–7294. https://doi.org/10.1021/acs.est.7b01810.

Schwietzke, S., Sherwood, O. A., Bruhwiler, L. M. P., Miller, J. B., Etiope, G., Dlugokencky, E. J., Michel, S. E., Arling, V. A., Vaughn, B. H., White, J. W. C., & Tans, P. P. (2016). Upward revision of global fossil fuel methane emissions based on isotope database. Nature, 538(7623), 88–91. https://doi.org/10.1038/nature19797.

Shen, L., Zavala-Araiza, D., Gautam, R., Omara, M., Scarpelli, T., Sheng, J., Sulprizio, M. P., Zhuang, J., Zhang, Y., Qu, Z., & others. (2021). Unravelling a large methane emission discrepancy in Mexico using satellite observations. Remote Sensing of Environment, 260, 112461.

Sheng, J.-X., Jacob, D. J., Maasakkers, J. D., Sulprizio, M. P., Zavala-Araiza, D., & Hamburg, S. P. (2017). A high-resolution (0.1° × 0.1°) inventory of methane emissions from Canadian and Mexican oil and gas systems. Atmospheric Environment, 158, 211–215. https://doi.org/10.1016/j.atmosenv.2017.02.036.

Sheng, J.-X., Jacob, D. J., Turner, A. J., Maasakkers, J. D., Benmergui, J., Bloom, A. A., Arndt, C., Gautam, R., Zavala-Araiza, D., Boesch, H., & Parker, R. J. (2018). 2010–2016 methane trends over Canada, the United States, and Mexico observed by the GOSAT satellite: Contributions from different source sectors. Atmospheric Chemistry and Physics, 18, 12257–12267. https://doi.org/10.5194/acp-18-12257-2018.

Sherwin, E. D., Chen, Y., Ravikumar, A. P., & Brandt, A. R. (2021). Single-blind test of airplane-based hyperspectral methane detection via controlled releases. Elementa: Science of the Anthropocene, 9(1).

Singer, B. C., Pass, R. Z., Delp, W. W., Lorenzetti, D. M., & Maddalena, R. L. (2017). Pollutant concentrations and emission rates from natural gas cooking burners without and with range hood exhaust in nine California homes. Building and Environment, 122, 215–229. https://doi.org/10.1016/j.buildenv.2017.06.021.

Smith, M. L., Gvakharia, A., Kort, E. A., Sweeney, C., Conley, S. A., Faloona, I., Newberger, T., Schnell, R., Schwietzke, S., & Wolter, S. (2017). Airborne Quantification of Methane Emissions over the Four Corners Region. Environmental Science & Technology, 51(10), 5832–5837. https://doi.org/10.1021/acs.est.6b06107.

Smith, M. L., Kort, E. A., Karion, A., Sweeney, C., Herndon, S. C., & Yacovitch, T. I. (2015). Airborne Ethane Observations in the Barnett Shale: Quantification of Ethane Flux and Attribution of Methane Emissions. Environmental Science & Technology, 49(13), 8158–8166. https://doi.org/10.1021/acs.est.5b00219.

Stidworthy, A. G., Davis, K. J., & Leavey, J. (2016). Radon emissions from natural gas power plants at The Pennsylvania State University. Journal of the Air & Waste Management Association, 66(11), 1141–1150. https://doi.org/10.1080/10962247.2016.1202157.

PSE

Subramanian, R., Williams, L. L., Vaughn, T. L., Zimmerle, D., Roscioli, J. R., Herndon, S. C., Yacovitch, T. I., Floerchinger, C., Tkacik, D. S., Mitchell, A. L., Sullivan, M. R., Dallmann, T. R., & Robinson, A. L. (2015). Methane Emissions from Natural Gas Compressor Stations in the Transmission and Storage Sector: Measurements and Comparisons with the EPA Greenhouse Gas Reporting Program Protocol. Environmental Science & Technology, 49(5), 3252–3261. https://doi.org/10.1021/es5060258.

Sun, C., Zhang, Y., Alessi, D. S., & Martin, J. W. (2019). Nontarget profiling of organic compounds in a temporal series of hydraulic fracturing flowback and produced waters. Environment International, 131, 104944. https://doi.org/10.1016/j.envint.2019.104944.

Sun, P., Young, B., Elgowainy, A., Lu, Z., Wang, M., Morelli, B., & Hawkins, T. (2019). Criteria Air Pollutant and Greenhouse Gases Emissions from U.S. Refineries Allocated to Refinery Products. Environmental Science & Technology, 53(11), 6556–6569. https://doi.org/10.1021/acs.est.8b05870.

Swarthout, R. F., Russo, R. S., Zhou, Y., Miller, B. M., Mitchell, B., Horsman, E., Lipsky, E., McCabe, D. C., Baum, E., & Sive, B. C. (2015). Impact of Marcellus Shale Natural Gas Development in Southwest Pennsylvania on Volatile Organic Compound Emissions and Regional Air Quality. Environmental Science & Technology, 49(5), 3175–3184. https://doi.org/10.1021/es504315f.

Thoma, E. D., Deshmukh, P., Logan, R., Stovern, M., Dresser, C., & Brantley, H. L. (2017). Assessment of Uinta Basin oil and natural gas well pad pneumatic controller emissions. Journal of Environmental Protection, 8(4), 394.

Thompson, T. M., Shepherd, D., Stacy, A., Barna, M. G., & Schichtel, B. A. (2017). Modeling to Evaluate Contribution of Oil and Gas Emissions to Air Pollution. Journal of the Air & Waste Management Association, 67(4), 445–461. https://doi.org/10.1080/10962247.2016.1251508.

Thorpe, A. K., Duren, R. M., Conley, S., Prasad, K. R., Bue, B. D., Yadav, V., Foster, K. T., Rafiq, T., Hopkins, F. M., Smith, M. L., Fischer, M. L., Thompson, D. R., Frankenberg, C., McCubbin, I. B., Eastwood, M. L., Green, R. O., & Miller, C. E. (2020). Methane emissions from underground gas storage in California. Environmental Research Letters, 15(4), 045005. https://doi.org/10.1088/1748-9326/ab751d.

Thorpe, A. K., Frankenberg, C., Aubrey, A. D., Roberts, D. A., Nottrott, A. A., Rahn, T. A., Sauer, J. A., Dubey, M. K., Costigan, K. R., Arata, C., Steffke, A. M., Hills, S., Haselwimmer, C., Charlesworth, D., Funk, C. C., Green, R. O., Lundeen, S. R., Boardman, J. W., Eastwood, M. L., … McFadden, J. P. (2016). Mapping methane concentrations from a controlled release experiment using the next generation airborne visible/infrared imaging spectrometer (AVIRIS-NG). Remote Sensing of Environment, 179, 104–115. https://doi.org/10.1016/j.rse.2016.03.032.

Thorpe, A. K., Frankenberg, C., Thompson, D. R., Duren, R. M., Aubrey, A. D., Bue, B. D., Green, R. O., Gerilowski, K., Krings, T., Borchardt, J., Kort, E. A., Sweeney, C., Conley, S., Roberts, D. A., & Dennison, P. E. (2017). Airborne DOAS retrievals of methane, carbon dioxide, and water vapor concentrations at high spatial resolution: Application to AVIRIS-NG. Atmospheric Measurement Techniques, 10(10), 3833–3850. https://doi.org/10.5194/amt-10-3833-2017.

Tian, W., Wu, X., Shen, T., Zhang, Z., & Karla, S. (2017). Quantitative Prediction of Radon Concentration at Wellhead in Shale Gas Development. SPE Journal, 22(01), 235–243. https://doi.org/10.2118/180358-PA.

Townsend-Small, A., Botner, E. C., Jimenez, K. L., Schroeder, J. R., Blake, N. J., Meinardi, S., Blake, D. R., Sive, B. C., Bon, D., Crawford, J. H., Pfister, G., & Flocke, F. M. (2016a). Using stable isotopes of hydrogen to quantify biogenic and thermogenic atmospheric methane sources: A case study from the Colorado Front Range: Hydrogen Isotopes in the Front Range. Geophysical Research Letters, 43(21), 11,462-11,471. https://doi.org/10.1002/2016GL071438.

PSE

Townsend-Small, A., Marrero, J. E., Lyon, D. R., Simpson, I. J., Meinardi, S., & Blake, D. R. (2015). Integrating Source Apportionment Tracers into a Bottom-up Inventory of Methane Emissions in the Barnett Shale Hydraulic Fracturing Region. Environmental Science & Technology, 49(13), 8175–8182. https://doi.org/10.1021/acs.est.5b00057.

Townsend-Small, A., Ferrara, T. W., Lyon, D. R., Fries, A. E., & Lamb, B. K. (2016b). Emissions of coalbed and natural gas methane from abandoned oil and gas wells in the United States. Geophysical Research Letters, 43(5), 2283–2290. https://doi.org/10.1002/2015GL067623.

Turner, A. J., Jacob, D. J., Benmergui, J., Wofsy, S. C., Maasakkers, J. D., Butz, A., Hasekamp, O., & Biraud, S. C. (2016). A large increase in U.S. methane emissions over the past decade inferred from satellite data and surface observations. Geophysical Research Letters, 43(5), 2218–2224. https://doi.org/10.1002/2016GL067987.

Tyner, D. R., & Johnson, M. R. (2021). Where the Methane Is—Insights from Novel Airborne LiDAR Measurements Combined with Ground Survey Data. Environmental Science & Technology.

Ulrich, B. A., Mitton, M., Lachenmeyer, E., Hecobian, A., Zimmerle, D., & Smits, K. M. (2019). Natural Gas Emissions from Underground Pipelines and Implications for Leak Detection. Environmental Science & Technology Letters, 6(7), 401–406. https://doi.org/10.1021/acs.estlett.9b00291.

Umeozor, E. C., Jordaan, S. M., & Gates, I. D. (2018). On methane emissions from shale gas development. Energy, 152, 594–600. https://doi.org/10.1016/j.energy.2018.03.151.

United Nations Environment Program (UNEP) & Climate and Clean Air Coalition (CCAC). (2021). Global Methane Assessment (full report). Climate & Clean Air Coalition. https://www.ccacoalition.org/en/resources/global-methane-assessment-full-report.

US EIA (United States Energy Information Administration). (2021). U.S. Natural Gas Gross Withdrawals. https://www.eia.gov/dnav/ng/hist/n9010us2A.htm.

US EPA (United States Environmental Protection Agency). (2020). Inventory of U.S. Greenhouse Gas Emissions and Sinks: 1990-2018. https://www.epa.gov/sites/production/files/2020-04/documents/us-ghg-inventory-2020-main-text.pdf.

US EPA (United States Environmental Protection Agency). (2016). Control Techniques Guidelines for the Oil and Natural Gas Industry. EPA-453/B-16-001, 343.

US EPA (United States Environmental Protection Agency). (2018). Inventory of U.S. Greenhouse Gas Emissions and Sinks: 1990-2016 (EPA 430-R-18-003). https://www.epa.gov/sites/production/files/2018-01/documents/2018_complete_report.pdf.

US EPA (United States Environmental Protection Agency). (2021). Inventory of U.S. Greenhouse Gas Emissions and Sinks: 1990-2019 (EPA 430-R-21-005). https://www.epa.gov/ghgemissions/inventory-us-greenhouse-gas-emissions-and-sinks-1990-2019.

Varon, D. J., Jacob, D. J., Jervis, D., & McKeever, J. (2020). Quantifying time-averaged methane emissions from individual coal mine vents with GHGSat-D satellite observations. Environmental Science & Technology, 54(16), 10246–10253.

Varon, D., McKeever, J., Jervis, D., Maasakkers, J., Pandey, S., Houweling, S., Aben, I., Scarpelli, T., & Jacob, D. (2019). Satellite discovery of anomalously large methane point sources from oil/gas production. Geophysical Research Letters, 46(22), 13507–13516.

PSE

Vaughn, T. L., Bell, C. S., Pickering, C. K., Schwietzke, S., Heath, G. A., Pétron, G., Zimmerle, D. J., Schnell, R. C., & Nummedal, D. (2018). Temporal variability largely explains top-down/bottom-up difference in methane emission estimates from a natural gas production region. Proceedings of the National Academy of Sciences, 115(46), 11712–11717. https://doi.org/10.1073/pnas.1805687115.

Vaughn, T. L., Bell, C. S., Yacovitch, T. I., Roscioli, J. R., Herndon, S. C., Conley, S., Schwietzke, S., Heath, G. A., Pétron, G., & Zimmerle, D. (2017). Comparing facility-level methane emission rate estimates at natural gas gathering and boosting stations. Elem Sci Anth, 5(0), 71. https://doi.org/10.1525/elementa.257.

Vaughn, T. L., Luck, B., Williams, L., Marchese, A. J., & Zimmerle, D. (2021). Methane Exhaust Measurements at Gathering Compressor Stations in the United States. Environmental Science & Technology, 55(2), 1190–1196.

Verhulst, K. R., Karion, A., Kim, J., Salameh, P. K., Keeling, R. F., Newman, S., Miller, J., Sloop, C., Pongetti, T., Rao, P., Wong, C., Hopkins, F. M., Yadav, V., Weiss, R. F., Duren, R. M., & Miller, C. E. (2017). Carbon dioxide and methane measurements from the Los Angeles Megacity Carbon Project – Part 1: Calibration, urban enhancements, and uncertainty estimates. Atmospheric Chemistry and Physics, 17(13), 8313–8341. https://doi.org/10.5194/acp-17-8313-2017.

von Fischer, J. C., Cooley, D., Chamberlain, S., Gaylord, A., Griebenow, C. J., Hamburg, S. P., Salo, J., Schumacher, R., Theobald, D., & Ham, J. (2017). Rapid, Vehicle-Based Identification of Location and Magnitude of Urban Natural Gas Pipeline Leaks. Environmental Science & Technology, 51(7), 4091–4099. https://doi.org/10.1021/acs.est.6b06095.

von Stackelberg, K., Buonocore, J., Bhave, P. V., & Schwartz, J. A. (2013). Public health impacts of secondary particulate formation from aromatic hydrocarbons in gasoline. Environmental Health, 12(1), 19. https://doi.org/10.1186/1476-069X-12-19.

Walters, K., Jacobson, J., Kroening, Z., & Pierce, C. (2015). PM 2.5 Airborne Particulates Near Frac Sand Operations. Journal of Environmental Health, 78(4), 8–12.

Weller, Z. D., Hamburg, S. P., & von Fischer, J. P. (2020). A National Estimate of Methane Leakage from Pipeline Mains in Natural Gas Local Distribution Systems. Environmental Science & Technology, 54(22), 8958-8967. https://doi.org/10.1021/acs.est.0c00437.

Weller, Z. D., Roscioli, J. R., Daube, W. C., Lamb, B. K., Ferrara, T. W., Brewer, P. E., & von Fischer, J. C. (2018). Vehicle-Based Methane Surveys for Finding Natural Gas Leaks and Estimating Their Size: Validation and Uncertainty. Environmental Science & Technology, acs.est.8b03135. https://doi.org/10.1021/acs.est.8b03135.

Weyant, C. L., Shepson, P. B., Subramanian, R., Cambaliza, M. O. L., Heimburger, A., McCabe, D., Baum, E., Stirm, B. H., & Bond, T. C. (2016). Black Carbon Emissions from Associated Natural Gas Flaring. Environmental Science & Technology, 50(4), 2075–2081. https://doi.org/10.1021/acs.est.5b04712.

Williams, J. P., Regehr, A., & Kang, M. (2021). Methane Emissions from Abandoned Oil and Gas Wells in Canada and the United States. Environmental Science & Technology, 55(1), 563–570. https://doi.org/10.1021/acs.est.0c04265.

Williams, P. J., Reeder, M., Pekney, N. J., Risk, D., Osborne, J., & McCawley, M. (2018). Atmospheric impacts of a natural gas development within the urban context of Morgantown, West Virginia. Science of The Total Environment, 639, 406–416. https://doi.org/10.1016/j.scitotenv.2018.04.422.

Willyard, K. A., & Schade, G. W. (2019). Flaring in two Texas shale areas: Comparison of bottom-up with top-down volume estimates for 2012 to 2015. Science of The Total Environment, 691, 243–251. https://doi.org/10.1016/j.scitotenv.2019.06.465.

PSE

Wisen, J., Chesnaux, R., Werring, J., Wendling, G., Baudron, P., & Barbecot, F. (2020). A portrait of wellbore leakage in northeastern British Columbia, Canada. Proceedings of the National Academy of Sciences, 117(2), 913–922. https://doi.org/10.1073/pnas.1817929116.

Wong, C. K., Pongetti, T. J., Oda, T., Rao, P., Gurney, K. R., Newman, S., Duren, R. M., Miller, C. E., Yung, Y. L., & Sander, S. P. (2016). Monthly trends of methane emissions in Los Angeles from 2011 to 2015 inferred by CLARS-FTS observations. Atmospheric Chemistry and Physics; Katlenburg-Lindau, 16(20), 13121–13130. http://dx.doi.org/10.5194/acp-16-13121-2016.

Wunch, D., Toon, G. C., Hedelius, J. K., Vizenor, N., Roehl, C. M., Saad, K. M., Blavier, J.-F. L., Blake, D. R., & Wennberg, P. O. (2016). Quantifying the loss of processed natural gas within California's South Coast Air Basin using long-term measurements of ethane and methane. Atmospheric Chemistry and Physics, 16(22), 14091–14105. https://doi.org/10.5194/acp-16-14091-2016.

Yacovitch, T. I., Daube, C., Vaughn, T. L., Bell, C. S., Roscioli, J. R., Knighton, W. B., Nelson, D. D., Zimmerle, D., Pétron, G., & Herndon, S. C. (2017). Natural gas facility methane emissions: Measurements by tracer flux ratio in two US natural gas producing basins. Elem Sci Anth, 5(0), 69. https://doi.org/10.1525/elementa.251.

Yacovitch, T. I., Herndon, S. C., Pétron, G., Kofler, J., Lyon, D., Zahniser, M. S., & Kolb, C. E. (2015). Mobile Laboratory Observations of Methane Emissions in the Barnett Shale Region. Environmental Science & Technology, 49(13), 7889–7895. https://doi.org/10.1021/es506352j.

Yang, S., Lan, X., Talbot, R., & Liu, L. (2019). Characterizing anthropogenic methane sources in the Houston and Barnett Shale areas of Texas using the isotopic signature δ13C in CH4. Science of The Total Environment, 696, 133856. https://doi.org/10.1016/j.scitotenv.2019.133856.

Yin, Y., Mastalerz, M., Lennon, J. T., Drobniak, A., & Schimmelmann, A. (2020). Characterization and microbial mitigation of fugitive methane emissions from oil and gas wells: Example from Indiana, USA. Applied Geochemistry, 118, 104619. https://doi.org/10.1016/j.apgeochem.2020.104619.

Yuan, B., Kaser, L., Karl, T., Graus, M., Peischl, J., Campos, T. L., Shertz, S., Apel, E. C., Hornbrook, R. S., Hills, A., Gilman, J. B., Lerner, B. M., Warneke, C., Flocke, F. M., Ryerson, T. B., Guenther, A. B., & de Gouw, J. A. (2015). Airborne flux measurements of methane and volatile organic compounds over the Haynesville and Marcellus shale gas production regions: AIRBORNE EDDY COVARIANCE FLUX. Journal of Geophysical Research: Atmospheres, 120(12), 6271–6289. https://doi.org/10.1002/2015JD023242.

Zavala-Araiza, D., Alvarez, R. A., Lyon, D. R., Allen, D. T., Marchese, A. J., Zimmerle, D. J., & Hamburg, S. P. (2017). Super-emitters in natural gas infrastructure are caused by abnormal process conditions. Nature Communications, 8(1), 14012. https://doi.org/10.1038/ncomms14012.

Zavala-Araiza, D., Herndon, S. C., Roscioli, J. R., Yacovitch, T. I., Johnson, M. R., Tyner, D. R., Omara, M., & Knighton, B. (2018). Methane emissions from oil and gas production sites in Alberta, Canada. Elem Sci Anth, 6(1), 27. https://doi.org/10.1525/elementa.284.

Zavala-Araiza, D., Lyon, D. R., Alvarez, R. A., Davis, K. J., Harriss, R., Herndon, S. C., Karion, A., Kort, E. A., Lamb, B. K., Lan, X., Marchese, A. J., Pacala, S. W., Robinson, A. L., Shepson, P. B., Sweeney, C., Talbot, R., Townsend-Small, A., Yacovitch, T. I., Zimmerle, D. J., & Hamburg, S. P. (2015a). Reconciling divergent estimates of oil and gas methane emissions. Proceedings of the National Academy of Sciences, 112(51), 15597–15602. https://doi.org/10.1073/pnas.1522126112.

PSE

Zavala-Araiza, D., Lyon, D., Alvarez, R. A., Palacios, V., Harriss, R., Lan, X., Talbot, R., & Hamburg, S. P. (2015b). Toward a Functional Definition of Methane Super-Emitters: Application to Natural Gas Production Sites. Environmental Science & Technology, 49(13), 8167–8174. https://doi.org/10.1021/acs.est.5b00133.

Zavala-Araiza, D., Omara, M., Gautam, R., Smith, M. L., Pandey, S., Aben, I., Almanza-Veloz, V., Conley, S., Houweling, S., Kort, E. A., & others. (2021). A tale of two regions: Methane emissions from oil and gas production in offshore/onshore Mexico. Environmental Research Letters, 16(2), 024019.

Zhang, Y., Gautam, R., Pandey, S., Omara, M., Maasakkers, J. D., Sadavarte, P., Lyon, D., Nesser, H., Sulprizio, M. P., Varon, D. J., Zhang, R., Houweling, S., Zavala-Araiza, D., Alvarez, R. A., Lorente, A., Hamburg, S. P., Aben, I., & Jacob, D. J. (2020). Quantifying methane emissions from the largest oil-producing basin in the United States from space. Science Advances, 6(17), eaaz5120. https://doi.org/10.1126/sciadv.aaz5120.

Zimmerle, D. J., Pickering, C. K., Bell, C. S., Heath, G. A., Nummedal, D., Pétron, G., & Vaughn, T. L. (2017). Gathering pipeline methane emissions in Fayetteville shale pipelines and scoping guidelines for future pipeline measurement campaigns. Elem Sci Anth, 5(0), 70. https://doi.org/10.1525/elementa.258.

Zimmerle, D. J., Vaughn, T., Luck, B., Lauderdale, T., Keen, K., Harrison, M., Marchese, A., Williams, L., & Allen, D. (2020). Methane Emissions from Gathering Compressor Stations in the U.S. Environmental Science & Technology, 54(12), 7552–7561. https://doi.org/10.1021/acs.est.0c00516.

Zimmerle, D. J., Williams, L. L., Vaughn, T. L., Quinn, C., Subramanian, R., Duggan, G. P., Willson, B., Opsomer, J. D., Marchese, A. J., Martinez, D. M., & Robinson, A. L. (2015). Methane Emissions from the Natural Gas Transmission and Storage System in the United States. Environmental Science & Technology, 49(15), 9374–9383. https://doi.org/10.1021/acs.est.5b01669.

PSE

Acronyms and Abbreviations

Term	Definition
AMCV	air monitoring comparison values (TCEQ)
Alt-FEMP	Alberta Energy Regulator's Alternative Fugitive Emissions Management Program
ATV	all-terrain vehicle
AVIRIS-NG	airborne visible-infrared imaging spectrometer - next generation
BenMAP	benefits mapping and analysis program
BTEX	benzene, toluene, ethylbenzene, and xylene
BU	bottom-up
CalEPA	California Environmental Protection Agency
CAMx	comprehensive air modeling with extensions
CAP	criteria air pollutant
CEMS	continuous emissions monitoring system
CARB	California Air Resources Board
DRE	destruction removal efficiency
ECCC NIR	Environment and Climate Change Canada National Inventory Report
EF	emissions factor
EIA	Energy Information Administration
EM27/SUN	Bruker Optics portable solar-viewing spectrometer
EPA	Environmental Protection Agency (U.S.)
ESL	effects screening levels
FCC	fluidized-bed catalytic cracking
FEAST	fugitive emissions abatement simulation toolkit
FID	flame ionization detector
FLIR	forward-looking infrared
GAO	Global Airborne Observatory
GeoCarb	Geostationary Carbon Observatory
GHG	greenhouse gas

PSE

Term	Definition
GHGI	greenhouse gas inventory (U.S. EPA)
GHGRP	Greenhouse Gas Reporting Program (U.S. EPA)
GHGSat	greenhouse gas satellite
GHGSat-D	greenhouse gas satellite demonstrator
GMD	Global Monitoring Division (NOAA)
GML	gas mapping LiDAR
GOSAT	greenhouse gases observing satellite
GOSAT-2	greenhouse gases observing satellite 2
HAP	hazardous air pollutant
HBACV	health-based comparison value
HDAP	health-damaging air pollutant
HI	hazard indices
HyTES	hyperspectral thermal emission spectrometer
INECC	Instituto Nacional de Ecología y Cambio Climático
IVOC	intermediate volatile organic compounds
LDAR	leak detection and repair
LiDAR	light detection and ranging
LNG	liquefied natural gas
MARKAL	market allocation
MATES	multiple air toxics exposure study
MDL	minimum detection limits
MERLIN	methane remote sensing lidar mission
MFP	mobile flux plane
MethaneSAT	methane satellite
N/A	not available
NAAQS	national ambient air quality standards
NASA	National Aeronautics and Space Administration

PSE

Term	Definition
NEI	national emissions inventory
NGLs	natural gas liquids
NIOSH	National Institute of Occupational Safety and Health
NMHC	non-methane hydrocarbon
NMVOC	non-methane volatile organic compounds
NOAA	National Oceanic and Atmospheric Administration
NO_x	nitrogen oxides
OEHHA	Office of Environmental Health Hazard Assessment
OGD	oil and gas development
OGI	optical gas imaging
ONG	oil and natural gas
OSHA	Occupational Safety and Health Administration
OTM 33A	other test method 33A (EPA)
PACE	Princeton atmospheric chemistry experiment
PADDs	Petroleum Administration for Defense Districts
PA DEP	Pennsylvania Department of Environmental Protection
PAH	polycyclic aromatic hydrocarbon
PAN	peroxyacyl nitrates
PC	pneumatic controllers
PEL	permissible exposure limits
PM	particulate matter
$PM_{2.5}$	particulate matter, 2.5 microns or less
PM_{10}	particulate matter, 10 microns or less
PRISMA	preferred reporting items for systematic reviews and meta-analyses
PRISMA	precursore iperspettrale della missione applicativa
PSE	Physicians Scientists and Engineers for Healthy Energy
PTR-MS	proton-transfer-reaction mass-spectrometry

PSE

Term	Definition
REL	recommended exposure limits
REL	reference exposure levels
RfC	reference concentration
ROGER	repository for oil and gas energy research
RSL	regional screening levels
SCAQMD	South Coast Air Quality Monitoring District
SCFM	standard cubic feet per minute
SCIAMACHY	scanning imaging absorption spectrometer for atmospheric chartography
SOE	on site emissions
TCEQ	Texas Commission on Environmental Quality
TD	top-down
TEOM	tapered element oscillating microbalance
TROPOMI	tropospheric monitoring instrument
TxRRC	Texas Railroad Commission
UAV	unmanned aerial vehicle
UGS	underground gas storage
VIIRS	visible infrared imaging radiometer suite
VOC	volatile organic compound
WRAP	Western Regional Air Partnership

PSE

Units of Measurement

Unit	Definition
ft	feet
Gg/yr	gigagrams per year
g/hr	grams per hour
g/s	grams per second
hr	hour
kg	kilograms
kg/hr	kilograms per hour
kg/s	kilograms per second
km	kilometers
Kt	kilotons
m	meters
mg/hr	milligrams per hour
mi	miles
MT	metric tons
MT/hr	metric tons per hour
ppb	parts per billion
ppm	parts per million
tg/yr	teragrams per year
t/hr	metric tons per hour
$\mu g/m^3$	micrograms per cubic meter

PSE

Chemical Formulas

Chemical Formula	Compound Name
C_2H_2	acetylene
C_2H_6	ethane
C_3H_8	propane
CH_4	methane
CO	carbon monoxide
CO_2	carbon dioxide
H_2S	hydrogen sulfide
HNO_3	nitric acid
N_2O	nitrous oxide
NH_3	ammonia
NO_2	nitrous dioxide
NO_x	nitrogen oxides
O_3	ozone
SF_6	sulfur hexafluoride
SO_2	sulfur dioxide
SO_x	sulfur oxides
$\delta^{13}C\text{-}CH_4$	carbon-13 methane
$\delta^2H\text{-}CH_4$	deuterated methane
$\delta D\text{-}CH_4$	deuterated methane

PSE

Appendix A. Setting the stage: Knowledge and research gaps in the oil and gas industry in North America 2011–2014

Table of Contents

PSE

List of Tables

List of Figures

This section "sets the stage" by summarizing the state of knowledge and general research agenda that preceded our literature review for both methane and health-damaging air pollutants (HDAPs). This section encompasses literature published in 2011–2014 — the period directly preceding the primary literature review (January 2015 to August 2020).

Methane (CH_4) is a potent greenhouse gas and is the primary constituent of natural gas. Thus, anytime natural gas is released or leaked at any point of the oil or natural gas supply chains, the emitted methane impacts climate change. Methane also contributes to the formation of ground-level ozone in some areas. Additionally, there is growing concern about the other pollutants present in natural gas, as well as air pollutants created following the combustion of natural gas. Methane is somewhat unique from a climate-forcing perspective, because methane's potency as a greenhouse gas is a function of time spent in the atmosphere, and this potency declines over time. For this reason, methane is considered a short-lived climate pollutant in comparison to carbon dioxide (carbon dioxide is the benchmark-equivalent metric used to estimate the greenhouse gas effect). For example, the instantaneous global warming potential — the amount of heat absorbed by methane when it's immediately released into the atmosphere — is 120 times greater than that of carbon dioxide (CO_2). However, methane's ability to absorb heat decreases over time, resulting in a much less potent, but still significant 28x greater warming potential than CO_2 after 100 years (Balcombe et al. 2018). Thus, estimating the climate impacts from methane requires a value-based judgment around whether to consider more immediate impacts or longer-term impacts associated with a changing climate.

Global atmospheric methane concentrations have more than doubled in the past 150 years, in conjunction with global industrialization and urbanization (IPCC 2019), rising on average 6.9 +/- 2.7 ppb per year between 2007–2015 (Nisbet et al. 2016). Looking back even further, the growing concentration of atmospheric methane since 1750 accounts for approximately 17% of the subsequent growth in radiative forcing. The degree of warming is multiplied further by a factor of approximately two when including greenhouse gases produced during the degradation of methane in the atmosphere (Allen 2016; Myhre et al. 2013; US EPA 2021). Not surprisingly, reducing anthropogenic sources of methane was recently labeled as the "most powerful lever" to slow global warming in the near term, according to the recent United Nations Global Methane Assessment (UNEP & CCAC, 2021).

While it is clear that atmospheric methane concentrations have risen steadily since the mid-2000s, understanding the underlying causes of this increase remains a highly active area of scientific study. Four major hypotheses have emerged to account for the increase (Methane from Oil & Gas - Methane Tracker 2020 - Analysis n.d.):

- The natural mechanisms that break down methane in the atmosphere are becoming weaker.
- There has been a rise in biogenic sources of methane (e.g., from agriculture or waste).

PSE

- There has been a rise in natural sources of emissions (e.g., wetlands and other flood zones).
- There has been a rise in emissions from the extraction of fossil fuels.

These hypotheses remain an area of active research with no clear leading explanation. The only way to reduce this uncertainty — and better understand abatement opportunities — is to continue to improve data transparency and expand and deepen measurement activities.

Focusing on the last bullet point above, the intensification of natural gas development from unconventional geologic sources in North America had already exerted a major influence on the global energy prospectus by 2010. The rapid and disparate growth of the industry in some regions previously devoid of industrial activity challenged many areas of science and regulation. The coincident rapid development of unconventional hydrocarbon extraction and global rise in atmospheric methane led to an increased focus on methane emissions associated with hydrocarbon production, transportation, and consumption. As such, leaks of natural gas during production, processing, transmission, and distribution throughout the oil and gas sector quickly became an intense area of scientific inquiry.

Health-damaging air pollutants (HDAPs) — particulates and gaseous volatile or semi-volatile compounds that are hazardous to human health — are also emitted from the oil and gas sector. These pollutants include naturally-occurring constituents of petroleum products, products of complete or incomplete combustion processes used for hydrocarbon development or transport, and additive compounds used to facilitate hydrocarbon development or odorize final natural gas products. While methane emissions hold implications at a global scale in the context of a changing climate, HDAP emissions contribute to more locally- and regionally-realized health risks and impacts. A wide array of populations may be exposed to HDAP emissions from the oil and gas sector, including but not limited to those residing near oil and natural gas extraction sites (e.g., upstream), near storage facilities (e.g., midstream), or further down the distribution line as fuel sources make their way into industrial, residential, or commercial settings (e.g., downstream).

Methane associated with the oil and gas industry: 2010–2014

In North America in particular, early research on methane emissions from oil and natural gas was driven in part by a desire to understand the climate implications of the rapid switching from coal to natural gas for electricity generation. Research in the early 2010s addressed the lifecycle opportunity costs of fuel switching, noting that while direct CO_2 emissions are lower from gas combustion than coal combustion to generate an equivalent amount of electricity, a certain threshold level of methane leakage exists whereby net climate benefits of switching from coal to gas could be nullified. Nonetheless, over all timeframes, every methane molecule that escapes from the natural gas supply chain erodes the carbon dioxide (CO_2) emissions advantage that natural gas has over other fossil fuels, particularly in the near term. The best estimates suggested that methane leakage ranging from 1–5% could nullify the climactic

benefits of switching to natural gas, depending on the ultimate end use (e.g., electricity generation, transportation). The review by Alvarez et al. (2012), concluded that methane leakage from well to power plant combustion must remain below 3.2% for natural gas to realize any climate benefits over any fuel type (e.g., coal power generation, oil).

One of the primary knowledge gaps in 2011 was the accuracy of the underlying assumptions used to estimate methane leakage from natural gas operations, complicated by rapid changes in the scope and intensity of the industry (e.g., the growth of high-volume hydraulic fracturing in unconventional resources such as shale formations). Because the vast majority of methane emissions factors for natural gas operations were first derived from estimates produced in a 1996 joint report between the U.S. Environmental Protection Agency (U.S. EPA) and the Gas Research Institute (and later updated in 2010), studies in the early 2010s called into question the applicability of existing "conventional'' natural gas lifecycle methane leakage rates to emerging "unconventional" production and well stimulation methods (e.g., well flowback) (US EPA 2010). These early studies proposed a wide range of methane leakage estimates, some suggesting that unconventional shale wells emit twice as much methane as conventional wells and that natural gas leakage may be several times greater than the annual U.S. EPA National Greenhouse Gas Inventory (GHGI) (Howarth et al. 2011, 2012). Such high leakage rates would severely degrade any net climate benefits in switching from coal. Overall, studies during this germinal period reached a broad set of methane leakage estimates and conclusions that in some cases differed substantially (Burnham et al. 2012; Cathles 2012; Howarth et al. 2011, 2012; Jiang et al. 2011; Levi 2012; Pétron et al. 2012, 2013).

The systematic review by Brandt et al. (2014) summarized the tension at the time between a lack of methane emissions data across the natural gas supply chain within the backdrop of climate change:

"…this uncertainty range [1%–10% leakage along the natural gas supply chain] makes it difficult to make policy decisions regarding whether to promote natural gas as a bridge fuel to a low carbon economy."

In sum, policy is hard in the face of uncertain science. Perhaps the only point of widespread agreement that emerged from the early methane emissions and life cycle assessment studies was the need to reduce the high degree of uncertainty of methane emissions from oil and natural gas production and use. Importantly, this research gap seemed to preempt advancement of wide scale mitigation efforts. Or in other words, any kind of methane mitigation efforts first required a more complete understanding of the relative source contributions across the industry. Miller et al. (2013) explicitly noted at the time:

"Successful regulation of greenhouse gas emissions requires knowledge of current methane emissions sources."

Thus, the primary set of recommendations at the time was to rapidly conduct representative direct methane measurements at the facility level across the supply chain, in addition to

PSE

verifying "bottom-up" component-level emissions with "top-down" atmospheric measurements at various spatial and temporal scales. The terms "bottom-up" and "top-down" are general scientific terms that describe two different approaches of data gathering and processing. Within this context, top-down methane studies always start with measurements of the concentration of methane in the atmosphere, and then apply different modeling approaches to estimate the mass of methane emitted per unit time, space, or source(s). Bottom-up methane estimates take a disaggregated approach and rely on emissions measurements made directly from components or at the site level with the goal of obtaining a statistically representative sample of sources, then extrapolating to all sources. Bottom-up methods can entail any combination of stack test data, manufacturer data, emissions factors, engineering estimates, activity factors, and on-site measurements. Inventories such as the U.S. GHGI use a bottom-up approach that estimates emissions based on the product of activity and emissions factors summed across all respective sources within a distinct temporal and spatial resolution (Harriss et al. 2015; Heath et al. 2015).

In response, the 2012–2014 period witnessed a host of both bottom-up and top-down atmospheric studies geared towards verification of methane emissions and emissions factors for components and activities primarily in the upstream (Allen et al. 2013; Caulton et al. 2014; Jeong et al.2014; Karion et al. 2013; Miller et al. 2013; Peischl et al. 2013; Pétron et al. 2012). A range of sampling techniques were also deployed, including: on-site direct measurements from operating components (Allen et al. 2013); stationary tower sampling (Miller et al. 2013; Pétron et al. 2012); mobile automobile sampling (Pétron et al. 2012); the use of aircraft equipped to measure atmospheric methane enhancement originating from ground-level sources (Caulton et al. 2014; Jeong et al. 2014; Miller et al. 2013; Peischl et al. 2013); and the first use of satellite observations (Wecht et al. 2014). Around this time, we also saw the advent of the term "super-emitter," used to describe the seemingly omnipresent, yet uncertain number of large leaks suspected to be contributing disproportionately to overall emissions.

Much progress was made in this short time frame in advancing methods to use atmospheric measurements to estimate methane emissions that could be scaled to other unmeasured sources, but it quickly became apparent that apportioning the observed atmospheric methane concentrations to the appropriate sources was particularly challenging for oil and natural gas systems. Brandt et al. (2014) suggested that the greatest challenge for atmospheric studies was source attribution — or determining the set of sources where the observed methane originated. This was further complicated by the highly integrated, yet spatially dispersed oil and natural gas supply chain that was often co-located with natural sources of methane such as animal husbandry or landfills.

Nonetheless, this period witnessed substantial advancement in deriving scaled methane emission estimates from collected atmospheric methane concentrations. Some of these techniques included the use of mixing ratios of methane to other chemical markers or gasses co-emitted with methane (e.g., VOCs, propane) (Miller et al. 2013; Peischl et al. 2013; Pétron et al. 2012); the mass-balance approach (e.g., Karion et al. 2013); and the first use of an

atmospheric transport model coupled with geostatistical inverse modelling by Miller et al. (2013). We also saw the first studies collect primary methane emissions data from the midstream and downstream natural gas supply chains (Jeong et al. 2014; Peischl et al. 2013), as the majority of early studies focused only on upstream production processes.

While much progress was made during this relatively short period, the conflicting findings of the initial lifecycle assessments were generally echoed during this initial phase of direct measurement and atmospheric sampling studies. Not until 2014 did we begin to see some scientific consensus around methane emissions from oil and natural gas systems. In addition, 2014 represented a crucial time period where scientific understanding solidified on the outsized and largely underestimated impacts that the oil and natural gas sector were having on the atmosphere. Multiple major critical review articles or national-level sampling campaign studies were published in this time period, capping the previous 20 years of technical literature on methane emissions (Allen 2014b; Brandt et al. 2014; Heath et al. 2014; McGarry et al. 2014). The critical review *Methane Leaks from North American Natural Gas Systems*" by Brandt et al. (2014) collected all known top-down studies that reported measurement-based methane emissions estimates within North America and concluded that:

"Measurements across all scales show that official inventories consistently underestimate actual CH_4 emissions, with the natural gas and oil sectors as important contributors."

More specifically, in aggregating all top-down studies to date, Brandt et al. (2014) found that an estimated 14 Tg/year (7–21 Tg/year) of U.S. methane emissions were not accounted for in the U.S. national emissions inventory — representing approximately 50% (25%–100%) of the total man-made methane emissions in the United States. However, a large degree of uncertainty around this estimate remained, and the authors were unable to specifically quantify the component- or system-level contributions of methane emissions from the oil and natural gas systems.

The remainder of this section provides a deeper look at the key challenges, research gaps, and research recommendations of the 2011–2014 time period related to collection of primary methane measurement data. The section is framed around three interrelated themes that emerged during this period, which are also revisited extensively throughout the 2015–2020 results and conclusion sections: (1) top-down vs. bottom-up studies; (2) estimation and attribution methodologies and sampling technologies; and (3) super-emitters.

Top-down and bottom-up studies: Challenges, research gaps, and recommendations, 2011–2014

The umbrella Issue during 2011–2014 was the uncertainty regarding the mass of methane emitted from the oil and gas sector, rooted in the lack of agreement between bottom-up and top-down derived methane emission estimates. The framing of this issue was largely based around new methane emissions data collection efforts that were compared to existing methane emissions inventories such EPA's GHGI, which is a bottom-up approach. Inherently,

PSE

both bottom-up and top-down approaches have strengths and weaknesses. Top-down estimates provide an aggregate of all emissions within the boundary conditions considered and provide an important comparison for the bottom-up estimates. However, top-down estimates cannot easily distinguish emissions from specific source types, limiting the development of informed mitigation strategies. Bottom-up estimates are resource intensive, and may not provide sufficient statistical characterization of each source type to accurately estimate total emissions (Harriss et al. 2015).

Bottom-up estimates typically rely on assumptions regarding methane emission factors for individual pieces of equipment and processes, while top-down estimates rely on field-based or satellite-based atmospheric measurements, usually at the regional scale. However, bottom-up studies have also relied on some direct measurements at the individual facility level. On-site measurements for bottom-up studies can take many forms, but typically entail acoustic emissions detection; Hi-Flow sampling; or optical gas imaging (e.g., infrared cameras). While both acoustic and Hi-Flow sampling methods are capable of producing emissions rates during normal operation, not all fixtures and emissions points on-site are accessible to these types of sampling systems, potentially leading to an underestimation of emissions. Optical gas imaging offers the ability to identify leaks from afar and is often the first line of defense; however, the technology is generally unable to provide quantitative estimates, only qualitative leak detection. Overall, bottom-up emission estimates have also been limited by their inability to accurately characterize outliers (e.g., super-emitters) and overall intermittency of emissions due to short sampling durations on-site. Nonetheless, both bottom-up and top-down approaches share many overlapping and interrelated challenges — some of which would not be known without the other.

Recall that the best scientific estimates of methane emissions from the entire natural gas supply chain ranged from slightly over 1% to greater than 10% (Allen et al. 2013; Brandt et al. 2014; Harrison et al. n.d.; Howarth et al. 2011; McGarry et al. 2014; Wigley 2011). Even the U.S. Greenhouse Gas Inventory (GHGI) varied substantially from year to year in the late 2000s, driven primarily by the uncertainties in the production upstream sector. What was not uncertain, however, was the strong consensus that new measurement data were necessary to begin to reconcile the remaining discrepancies and provide more targeted research recommendations to ultimately inform mitigation measures. In light of the fact that the best estimates at the time were all leveraged against the U.S. GHGI — a type of "bottom-up" approach — the logical research recommendation writ large was to provide direct verification of these estimates using both direct measurement and other "top-down" atmospheric sampling methods.

Overall, top-down estimates have typically exhibited higher uncertainties regarding source identification compared to bottom-up, likely in part due to the presence of emissions intermittency and unsuspecting super-emitters, and inherent challenges of sample design and monitoring systems themselves. Comparisons between methods are useful for validation purposes and elucidating uncertainties, particularly when boundary conditions align spatially,

PSE

temporally, and sectorally. More generally, comparing results from both approaches, when collected simultaneously, has been found to be a useful study design. Both methodologies exhibit complimentary pros and cons, including the increased likelihood of capturing key emissions sources at the component level. As Heath et al. (2015) noted:

"In their current states, neither measurement- (top-down or bottom-up) nor inventory-based estimates of methane emissions are considered "gold standard" benchmarks, yet both can be useful."

In recognizing the emerging scientific challenges at the time, a host of critical review studies were published in 2014 that cataloged the previous 20 years of natural gas methane leakage measurements. These studies were an attempt to reconcile the explanation for differences in newer top-down sampling studies and existing bottom-up estimates (Allen 2014b; Brandt et al. 2014; Heath et al. 2014; McGarry et al. 2014). McGarry et al. (2014) analyzed a total of 35 journal articles, government reports, NGO reports, and industry reports from January 2010 to June 2014 and documented methane emissions estimates in comparison to the U.S. EPA's GHGI estimates. Of the 24 peer-reviewed articles, 11 indicated that U.S. EPA's estimates were likely underestimating the true emissions, while another 11 believed that U.S. EPA accurately assessed emissions, leaving two studies that believed U.S. EPA had overestimated.

The aforementioned disagreement between bottom-up and top-down studies up to 2014 resulted from both the inaccuracy of inventories themselves as well as measurement uncertainties in the atmospheric studies (Brandt et al. 2014). Given the generalized assumptions required in bottom-up approaches and the highly variable (and rapidly changing) oil and natural gas industry in the late 2000s, the level of disagreement observed should not have been surprising. Bottom-up approaches must first develop emissions factors by conducting some direct measurements as a function of activity level (e.g. throughput of natural gas). These estimates are then assumed to be representative of the true underlying distribution of source strengths, but the representativeness of these samples was strongly called into question early on (Allen 2014a, 2014b; Brandt et al. 2014; Heath et al. 2014; McGarry et al. 2014). Early top-down studies also exhibited their fair number of uncertainties and challenges which are discussed in greater detail in Section 3.

In an attempt to formalize the scientific understandings around top-down vs. bottom-up estimates, one of the first questions posed by Brandt et al. (2014) was:

"Why might emissions inventories be underpredicting what is observed in the atmosphere?"

The potential reasons offered by Brandt et al. (2014) harkened back to the early life-cycle assessment studies, whereby key assumptions underpinning bottom-up inventory methods were likely insufficient. These included:

- Uncertain infrastructure, whereby device counts used in inventories are contradictory, incomplete, and of unknown representativeness.

- The sampling of devices that are not representative of current technologies. For example, hydraulic fracturing and horizontal drilling were not utilized in the 1990s when the U.S. EPA emissions factors were first established.
- Resource constraints leading to small sample sizes, as well as possible sampling selection bias with self-selected cooperating facilities.
- Long-tailed distributions whereby more high-emissions sources exist than would be expected in a normal distribution (i.e., super-emitters).

The review by Heath et al. (2015) largely echoed each of these issues, noting that the four major challenges related to the accuracy of inventories were:

- Completeness and accuracy of inventoried sources;
- Representativeness of emission samples;
- Quantifying uncertainty; and
- Variable and heavily skewed distributions of emissions within a source category (i.e., super-emitters).

The critical review by Allen (2014b) that explicitly focused on reconciling bottom-up and top-down measurements concluded that the wide variation in top-down estimates (larger than bottom-up estimates) at this time was due to:

- Differences between the estimation and measurement approaches themselves;
- Large population of disparate, heterogeneous sources; and
- Extreme values of emissions rates from individual sources that are much larger than the population average (i.e., super-emitters).

Notably, all three reviews pointed out that the most fundamental assumption of any inventory is knowing the number of all contributing sources. Each review also identified super-emitters as a unique challenge, which is discussed below. At the time, there was much uncertainty around the number of systems and individual sources in operation across the U.S. (that is still plaguing inventories today). The best estimates in the U.S. as of 2014 were 500,000 existing gas wells; 10,000 gathering stations, which encompassed ~100,000 compressor stations and other related equipment; and ~200,000 miles of gathering pipelines in the upstream (e.g., production, gathering and boosting) sector. An estimated 580 processing plants functioned to dehydrate wet gas or remove impurities in the processing sector. The transmission and storage sector (i.e., midstream) included an additional ~320,000 miles of high-volume, long-distance pipelines and ~9,000 more compressors operating in concert with ~400 underground gas storage facilities. And finally, the distribution or downstream sector included ~2.1 million miles of smaller diameter pipelines and ~62 million buildings, each with their own gas service lines.

There were also known missing sources and activity-types in these early natural gas inventories, which admittedly were not included due to a lack of emissions factors:

- Hydraulic fracturing of oil wells;
- Abandoned oil and gas wells and other derelict infrastructure;
- Gathering pipelines;
- Methane dissolved in wastewater and emitted from flowback ponds, pits, and retention ponds;
- Geologic seeps;
- Drilling mud and other solid waste;
- Well work-overs;
- Well testing and maintenance (e.g., blowdowns); and
- Leakage from portions of the downstream local distribution systems, including off-take points for industrial users or behind-the-meter emissions.

Both Heath et al. (2015) and Brandt et al. (2014) also called into question the representativeness of direct measurement sampling to date. Oftentimes, emissions factors for a specific source are derived from a small number of samples that simply cannot be fully representative of the distribution of real-world operating conditions. Meanwhile there are a number of factors and parameters that will be uniquely site-specific and likely operating within a certain steady-state condition that may be particular to that system and subject to changes over time. Additionally, the functional condition of the system (e.g., steady-state vs. abnormal) at the time of sampling is ultimately unknown, especially if sampling is for only a short duration. Emissions may also vary over time by day, season or by other external forces (e.g., weather, market forces). There are likely system-level variations that could be explained in part by differences between component manufacturers, construction engineers, site location, geologic formations and hydrocarbon make-up, state and federal regulations/enforcement, operator-specific, and more.

Relatedly, Heath et al. (2015) stressed the importance of the interrelationship between the representativeness of samples that are included in any inventory and the resulting impact on statistical uncertainty of the derived emissions estimates. Heath et al. (2015) therefore suggested that a representative sample should capture a range of key factors that may be associated with emissions variability such as:

- Equipment age;
- Equipment type;
- Maintenance and integrity management;
- Duty cycle vs. recommended life span;
- Operator practices; and
- Emissions controls installed and functional status.

Related to equipment age, equipment degrades with time and engineered systems in general tend to improve with time. Therefore, within this context, equipment age can act as proxy for the integrity of a system through two main ways. First, by providing an objective accounting of how long a system has been exposed to natural degradation processes such as corrosion and

PSE

stresses from thermal and abrasive forces. Second, age can proxy for the knowledge and regulatory standards at the time of construction that likely informed design, materials used, and technological competency.

Heath et al. (2015) made specific recommendations to inform the U.S. GHGI using bottom-up direct measurements. In addition to prioritizing unmeasured sources (e.g., abandoned wells, gathering lines, post-meter emissions), Heath et al. (2015) recommended aligning future direct measurement sampling with the existing inventory source categories noting that regulatory policies are typically outlined at individual industrial sectors. Heath et al. (2015) also notes that making this differentiation is not necessarily straightforward, and is even more complicated by the increasing co-location of various hydrocarbon products (e.g., condensates, natural gas liquids, oil) and associated separate waste streams.

As mentioned above, the other potential culprit put forth at this time to explain the underestimation of bottom-up inventories was the presence of super-emitters. A "super-emitter" — as colloquially defined during this period — was the name given to the few disproportionately large emitting sources observed in early upstream measurement studies. More technically, the observed emissions distributions demonstrated "heavy tails" or "fat tails," which means they exhibited more high-emitting sources than would be expected in a normal distribution. The consistent presence of disproportionately large leaks in early studies was one of the most logical explanations put forth at the time, and has largely stood up over the years and even expanded into other portions of the supply chain (Table A.1). In essence, if total emissions are dominated by a small number of rare, high-emitting sources, emissions factors-derived measures of central tendency (i.e., arithmetic mean of samples) will systematically underpredict total emissions. Brandt et al. (2014) cited 10 studies dating back to 1997 that demonstrated the presence of super-emitters

PSE

Table A.1. Studies demonstrating the presence of super-emitters. Adapted[1] from Brandt et al. (2014) and Innovative Environmental Solutions, Inc. (2015).

% of sources contributing to…	Percent of emissions	Component or sector	Region	Method	Study
44%	90%	Production	United States	Direct measurement of unloading events	Allen et al. 2013
10%	70 %	Production	United States	Analysis of reported emissions	Alvarez et al. 2012
Single leak	Increased from 104 kg/hr to 450 kg/hr	Processing	Alberta, Canada	Down-wind differential absorption LIDAR	Chambers 2004
Top 10 leaks across four facilities (>100,000 device measures)	35.7%–64.6%	Processing	Western United States	Direct measurement using Hi-Flow	Clearstone Engineering 2002
Top single leak	40%	Transmission compressors		Direct measurement using Hi-Flow	Cormack 2007
20%	80%				
Top single leak	>100,000x larger than valve and flange emissions factor	Valve and flanges (compressor)	United States	IR camera, Hi-Flow sampler	Harrison et al. 2011
Top single leak	70%	Blowdown (compressor)			
16%	Three orders magnitude larger than the median flow rate	Abandoned oil and gas wells	Pennsylvania		Kang et al. 2014
Top 10 leaks (2%)	58%	Processing, well sites, gathering compressor stations	United States	Direct measurement using Hi-Flow sampler and optical methods	National Gas Machinery Laboratory et al. 2006
Top 10 leaks	**80%**	**All stages**		**Various**	**Picard 2005**

PSE

% of sources contributing to…	% of emissions	Component or sector	Region	Method	Study
Top emitters	100x–10,000x larger than small emitters	All stages		Remote sampling via tracer methods	Shorter et al. 1997
23%	77%	Compressor stations and gas plants		Optical measurement and Hi-Flow sampler	Trefiak 2006

Identifying and properly accounting for super-emitters will likely continue to be a major challenge in estimating methane emissions. Notably, many of the same recommendations related to primary data collection apply also to super-emitters. In essence, the higher the likelihood of the presence of super-emitters or disproportionately large leaks, the greater the sample size needed to sufficiently characterize the underlying distribution. Or in other words, the smaller the sample size, the greater the likelihood of not capturing larger sources that may be driving the heavy-tail of the (unknown at the time) distribution. At the time, Brandt et al. (2014) noted that very few datasets report emissions measurements in sufficient detail to allow for full characterization of suspected distributions of emissions across various sources, again reiterating a similar data availability gap as noted previously.

Overall, the primary recommendation to address many of the key research gaps and challenges identified in 2014–2015 was a need for additional representative direct methane measurements across the oil and natural gas supply chain, and verification of bottom-up component level emissions via top-down atmospheric measurement. Heath et al. (2015) called for a simultaneous data collection effort that entailed coordinated bottom-up direct measures with top-down sampling at matching spatial, temporal, and sectoral scales. Allen (2014b) echoed many of the recommendations from Brandt et al. (2014) and Heath et al. (2015), suggesting a hybrid bottom-up and top-down approach to improve emissions estimates and noting that similar combination approaches have been successful in other industries (such as the automotive industry) to identify super-emitters and reduce emissions. However, Allen (2014b) stated that the approach deployed in the automotive industry tests all vehicles annually, which would be cost prohibitive and impractical for the oil and natural gas sector. He suggested that a more practical option — sticking with the automotive analogy — would be to adopt a "check engine light" model. This model combines an intelligent monitoring system (composed of a host of smart sensing devices near production facilities), which triggers further testing when certain thresholds are met, with routine ambient sampling systems that vary in spatiotemporal coverage (e.g., mobile or aircraft deployment and stationary monitoring systems). However, given the resource costs of direct measurements and atmospheric studies, advances related to methane inventories should also remain a key priority (Brandt et al. 2014).

PSE

Top-down sampling methodologies and source attribution methodologies: Challenges, research gaps, and recommendations 2011–2014

The goal of top-down measurement studies is to estimate the amount of methane emitted per time and area from a set of suspected sources, based upon measured methane concentrations collected away from those sources. Essentially, top-down sampling can be considered a set of reverse-engineering steps that take advantage of decades of atmospheric transport science. Brandt et al. (2014) stated that the greatest challenge for atmospheric studies was source attribution: determining the set of sources where the observed methane originated, inclusive of both natural and man-made sources. Even when considering recent technological advances, determining the amount of methane emitted from an unknown source — based only upon its atmospheric measurement — presents quite the challenge. As Brandt et al. (2014) noted,

"...atmospheric transport [of methane] is the end result of a complex, nonlinear, multi-scale dynamical process."

As noted above, the 2012–2014 period witnessed a host of top-down direct measurement and atmospheric studies, and an accompanying range of sampling techniques. The most common sampling approaches deployed for regional top-down estimates were:

- Remote sensing (e.g., satellite, scanning Fourier transform spectrometer);
- Total column measurements (ground-based Fourier transform spectrometer);
- Stationary tall tower site or tower network; and
- Airborne in situ (e.g., aircraft, unmanned aerial vehicles).

Brandt et al. (2014) noted that the most common instrumentation deployed in these studies are typically either whole air flask samples measured by gas chromatography, direct absorption spectroscopy typically used in aircraft systems, or spectroscopy of an air column. The accuracies of these analysis methods typically range from 2–3.5 ppb, and therefore would not have a substantial impact on the overall methane emissions estimates if properly maintained and calibrated as per manufacturer's recommendation.

By 2014, we had witnessed numerous different methodologies deployed in top-down studies to estimate methane emissions at the regional scale (i.e., methane flux) using the data collected by the various instruments and sampling modes above. Each of these exhibited various degrees of complexity, assumptions, and advantages/disadvantages related to the emissions regime. These generally included:

- Transport box models (mass balance models);
- Transport inversion modeling;
- Tracer-tracer; and
- Other source apportionment methods, including co-emitted fingerprinting and isotopic signatures.

PSE

Transport box models

Transport box models represent the most straightforward study design that takes advantage of air movement aloft of emissions sources of interest. If the prevailing wind direction and velocity are known, the amount of methane released can be reasonably predicted by sampling the upwind and downwind concentrations. A higher downwind methane concentration represents the presence of methane sources within the source area of interest. To obtain an emissions estimate for sources within the "box," the difference in downwind vs. upwind concentrations is multiplied by the average rate of air movement and the horizontal dimension of the box (e.g., production basin or whole city).

Mass balance estimation methods have typically relied on instantaneous measurements collected by mobile sampling systems such as aircraft or ground-based vehicles. While instantaneous measurements carry their own drawbacks (e.g., inability to capture temporal variability), the advantage of box models lies in their simplicity. Numerous studies have applied various forms of box models or mass balance approaches to estimate methane emissions (Karion et al. 2013; Peischl et al. 2013). In some cases these efforts have led to very high emissions rates (Karion et al. 2013) that called into question their ability to apportion methane to sources of interest (Allen 2014a; Brandt et al. 2014). If other methane sources (or sinks) such as livestock, landfills, or other natural sources are co-located within the area of interest, they can potentially inflate (or deflate) estimates, although these can be subtracted out if estimates are available from other inventories. Moreover, if the sources of interest are natural gas only, co-located oil production, if present, could further complicate source attribution. Allen (2014a) also noted that emissions characteristics of a well will change with time. Early in a well or basin's lifetime, associated emissions may be dominated by well completion activities. As the field matures, more liquids may be produced, requiring more frequent liquid unloadings. This type of change in steady-state emissions over time can introduce additional uncertainty for the regional box model approach. Brandt et al. (2014) estimated the uncertainty ranges for simple box model approaches are upwards of 50%.

Transport inversion modeling

Transport inversion models similarly rely on methane measurements collected in the atmosphere. However, in contrast to mass balance approaches, transport inversion approaches use some form of statistical modeling to reverse-engineer the unknown location where the methane originated from. By 2014, at least four studies had used some form of atmospheric transport modelling to estimate methane emissions (Brandt et al. 2014). The most recent, at that time, was performed by Miller et al. (2013), who used ~12,000 observations collected over two years (2007–2008) from aircraft and tall towers throughout the United States, in combination with wind fields produced with the Weather Research and Forecasting model (WRF) and the Stochastic Time-Inverted Lagrangian Transport model (STILT). The study reported a U.S. anthropogenic methane budget 1.5 times larger than U.S. EPA estimates. The authors also noted that methane inventories for other sectors exhibited similar deficiencies at

PSE

that time, and therefore can't be relied upon to differentiate fossil methane by subtracting out contributions from other known methane sources.

From Miller et al. (2013), the authors noted the benefit that a multi-sampling technique study design can have in these types of sample plus modelling studies, and other estimation approaches writ large. For example, the authors showed the importance of aircraft data in comparison with the ground-based measurements — without the aircraft data, ground-based measurements alone resulted in an upward bias due to an inability to properly account for the vertical redistribution of surface emissions. The combination of ground-level measures and top-down atmospheric measures provided both internal comparisons and a form of external validation for each dataset alone. Overall, Brandt et al. (2014) noted that the biggest challenge of transport modelling are the inherent difficulties in simulating atmospheric transport itself. For instance, inaccurately defining the mixing height can propagate through the modeling framework, resulting in potentially large errors that may not be reflected in the statistical uncertainty of the error (i.e., error bars) due to the modelling structure. This is an example of a type of "systemic" uncertainty that cannot be understood or expressed through statistics alone.

Tracer-tracer

The tracer-tracer method for estimating methane emissions relies on measuring a gas other than methane that is believed to relate in some way to methane. This relationship can be either through co-emissions themselves (e.g., other alkanes such as ethane) or by simple co-location, whereby the two gases will be affected similarly by atmospheric transport mechanisms (e.g., CO [carbon monoxide] and CO_2 [carbon dioxide] from various sources). Also within this category are deliberate releases of non-reactive tracer compounds (e.g., SF_6 [sulfur hexafluoride], N_2O [nitrous oxide], C_2H_2 [acetylene]) released at a known rate at or near the methane emissions source(s) of interest. From this study design, methane emissions can be determined by sampling both gases upwind and downwind, or multiple transects downwind (see Figure A.1) and then multiplying the known emissions rate of the tracer gas by the downwind concentration ratio of methane to the tracer. This approach requires the justification of three related assumptions, as outlined in Allen et al. (2014). These are: (1) emissions of the tracer gas are well characterized; (2) the gases of interest are well mixed at the point of downwind sample collection; and (3) the atmospheric processes affect both gases identically. Allen (2014a) noted that deliberate tracer-tracer approaches produce some of the most precise methane emissions estimates, and multiple studies at the time had deployed these methods in various settings, both near upstream production sources (Katzenstein et al. 2003; Mønster et al. 2014), and in settings that exhibited complex methane signatures such as Los Angeles (Hsu et al. 2010; Wennberg et al. 2012; Wunch et al. 2009).

PSE
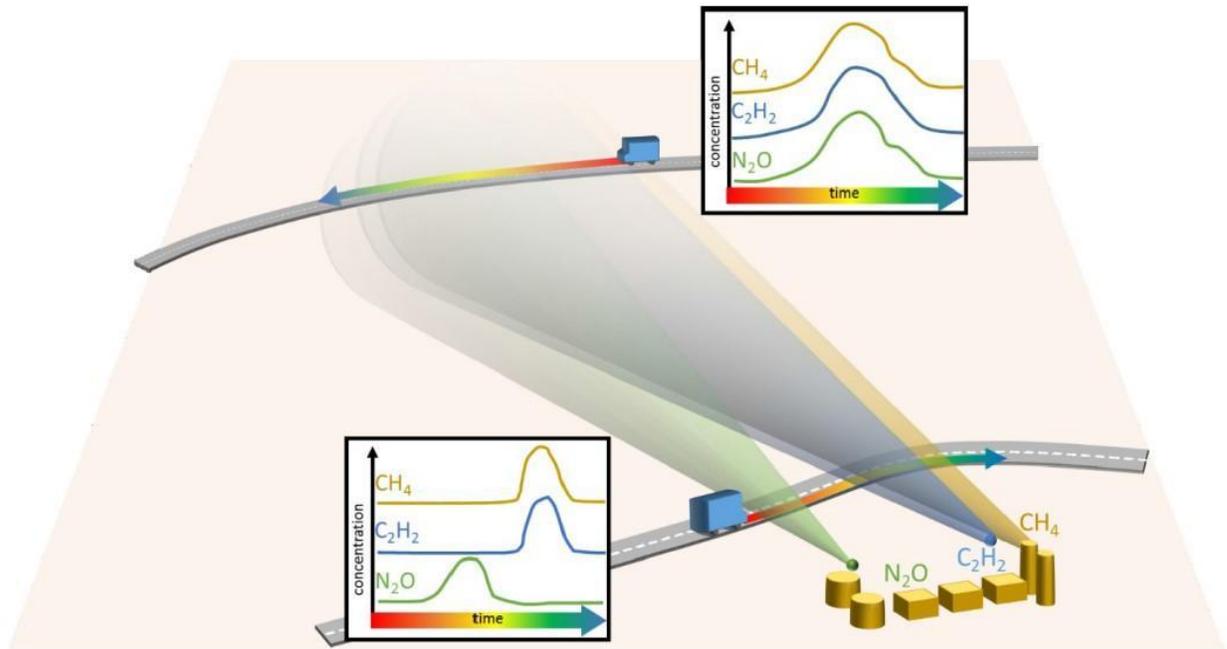
Figure A.1. Illustration of the tracer-tracer release technique using mobile measurement. The spatiotemporal offset of N_2O vs. C_2H_2 and methane (CH_4) in the near transect, compared to offsets at the far downwind transect, can indicate the relative location of the methane release — namely directly near the C_2H_2 release point. Source: Adapted[2] from Roscioli et al. (2015).

Tracer-tracer approaches have typically relied on ground-based sample collection modes only, notably tall tower and vehicle-based (Pétron et al. 2013). Multiple tracer-tracer studies in Los Angeles used carbon monoxide (CO) as a tracer (Hsu et al. 2010; Wennberg et al. 2012; Wunch et al. 2009), and took advantage of California's highly detailed annual CO inventory, estimated to have an uncertainty estimate of only 10% — much more accurate than any methane inventories to date. While using an existing inventory has its advantages, such as proxying for atmospheric transport, many other areas where oil and natural gas development takes place likely do not have as detailed inventories for other gases, limiting the generalizability of this approach to other areas. Moreover, leveraging bottom-up inventories in this fashion does entrain the associated bottom-up uncertainty onto the top-down estimate. The Petron et al. (2012) study sparked a debate related to this issue, which largely concluded that unless the non-methane overlapping inventory is vastly superior to that of methane (e.g., such as CO emissions in Los Angeles), the resulting estimates may not be sufficiently reliable.

---

[2] Reproduced as is permitted by the creative commons license. Content of the original figure not altered.

PSE

While not a tracer method per se, the use of source signatures such as isotopic measurements and alkane fingerprints had been proposed to overcome many issues with correctly apportioning methane sources in the top-down estimation studies. Because methane formation often occurs alongside other gases (depending on the external conditions), the relative abundances of these gases can help distinguish between source types such as fossil thermogenic methane (i.e., fossil) vs. ruminant biogenic methane. For example, the ratios of methane to certain non-methane hydrocarbons (e.g., ethane) are very high for biogenically-produced methane (e.g., from livestock). In contrast, much higher levels of alkanes, alkenes, alkynes, aromatics, and oxygenated VOCs are present with methane that was thermogenically-produced in underground strata. By 2014, numerous studies had used alkane signatures to apportion the proportional amount of methane to oil and natural gas system sources (Katzenstein et al. 2003; Miller et al. 2013; Pétron et al. 2014; Wennberg et al. 2012). While these methods were popular for good reason, Brandt et al. (2014) noted that alkane fingerprints can vary by production zone and gas compositions change following processing, limiting this method in the mid- and downstream. Allen (2014a) also noted that alkane fingerprints will likely change over time within a production system, but to a degree which is not fully understood.

Isotopic signatures can also indicate how and where methane formed (Townsend-Small et al. 2012; Wennberg et al. 2012). For methane, carbon isotopic ratios ($^{13}C/^{12}C$) are the most common signatures used for source apportionment. In essence, most methane molecules on Earth are composed of one atom of ^{12}C and four atoms of hydrogen; however, some organic substrates contain relatively elevated levels of ^{13}C that are constant over time. Distinct $^{13}C/^{12}C$ ratios exist; for example, methane from biomass burning has a different ratio compared to background atomspheric methane or methane that originated from a fossil source. While the methodology is promising, few studies at the time had attempted source apportionment for natural gas systems. Townsend-Small et al. (2014) sampled near multiple methane sources, such as power plants, oil fields, and landfills, and showed promising results; however, the sample sizes were small. Moreover, challenges exist related to instrumentation that requires extreme sensitivity to identify the very small changes to the isotopic signature, particularly for other thermogenically-formed sources of methane, such as geologic seepage, which are not related to oil and gas extraction (Heath et al. 2015).

Summary: Challenges, research gaps, and recommendations 2011–2014

As is common in other scientific disciplines facing growing uncertainties, the working assumption during this time period was that it was vital to separate and characterize methane sources before methane mitigation efforts could be recommended or deployed. Given the highly integrated, spatially dispersed, and heterogeneous nature of the North American oil and natural gas industry, direct measurement of the quantity of emissions from every source was untenable. Therefore, both bottom-up and top-down estimates must rely upon some form of statistical or other model-based approaches, which are inherently susceptible to a degree of uncertainty. Regardless of the extent of future direct measurement and atmospheric sampling,

PSE

a degree of uncertainty is inescapable. The goal and scientific challenge herein will continue to focus on reducing these uncertainties in the most cost-effective manner possible.

Efforts to reduce methane and co-pollutant emissions have been limited by a lack of reliable data. However, much progress has been made over the past decade, supported by a wide variety of measurement technologies and related scientific study to evaluate effectiveness. Moreover, a lack of perfect information should not impede progress to reduce emissions. Recent research has helped get a better understanding of the diverse processes and conditions that cause natural gas leakage. Overall, the major research gaps and recommendations during this time period were:

- Reported emissions variability overall suggested a poor understanding of the sources potentially driving the observed excess methane levels leading to uncertainty in where to improve the science.

- The least well-constrained sources that could potentially explain the observed excess methane were abandoned oil and gas wells, geologic seeps, and local distribution systems.

    o Other sources of methane not included within the U.S. GHGI included gathering pipelines, methane dissolved in produced water, solid waste sources, well work-overs that are not completions, and well testing activities.

    o Local distribution leaks that occur downstream of customer meters associated with end-use appliances are particularly uncertain, and are not included in the U.S. GHGI. Some evidence suggests that local distribution leaks alone could effectively double the overall methane leakage rate.

- Regional and multi-state studies focused on both upstream and downstream sources found larger excess methane emissions than national scale studies, possibly due to averaging effects from atmospheric processes. Trade-offs exist between spatial and temporal study designs — regional studies may collect many samples over a short period of time, as compared to national studies that use fewer samples across space.

- Overall, the frequency and root causes of high-emitting sources was unknown.

- The contribution of intermittent, short-duration events was unknown.

- Emissions ratios for top-down studies are more variable than bottom-up; such top-down atmospheric studies must infer fluxes by accounting for atmospheric transport and assumptions therein (e.g., steady-state, homogenous wind conditions).

- Sampling scale and location must be considered when attempting to attribute emissions to subsectors to reduce uncertainties associated with co-located sources and gas composition variability.

PSE

- Methods for attributing total methane concentrations to their ground source need to be improved given the complex and co-located methane source regimes in many areas of the United States.

- New attribution methods are needed to differentiate other geologic sources of methane, such as methane from geologic seepage or coal-bed methane, that exhibit identical isotopic signatures to fossil methane.

- Continued advancement of downscaling methodologies to improve the spatial resolution capabilities in apportioning specific methane sources down to individual facilities.

- Additionally, many current solutions were already being adopted and some present opportunities to be cost-effective. These included:

  - Reduced emissions completions;

  - Leak detection and repair; and

  - New methods, models, and measurement systems to rapidly identify and repair the small fraction of high-emitting sources; however, initial identification remains a challenge given the breadth of oil and natural gas infrastructure currently in place.

Health-damaging air pollutants associated with the oil and gas industry, 2011–2014

While there is significant evidence that the oil and natural gas sector is a major source of methane emissions in the United States, fewer studies have examined its role as a source of health-damaging air pollutants (HDAPs)[3]. Additionally, as a greenhouse gas, methane emissions hold implications at the global scale, in both the near- and long-term. In contrast, the benefits of mitigating HDAP emissions are realized locally and regionally in the near-term for populations that live near upstream, midstream, and downstream from sources of oil and gas-associated emissions. While there has been substantial research on health co-benefits of CO_2 emission-reduction strategies in transportation, electricity, buildings, and industry, there is a paucity of research on the health benefits of policies that reduce methane emissions and co-emitted HDAPs prior to 2015, particularly for emissions from the oil and gas supply chain.

This section provides an assessment of the peer-reviewed literature from 2011–2014 to determine the current state of the science regarding HDAPs co-emitted with methane from the

---

[3] In this study, HDAPs are defined as any air pollutant (including gases and particulates) that is hazardous to human health when exposed.

PSE

oil and gas sector as of 2015, including: (1) enumerating the HDAPs associated with the oil and gas sector; (2) reviewing HDAPs co-emitted with oil and gas sector methane emissions; (3) summarizing findings from 2011–2014 studies examining health hazards and risks associated with HDAP emissions from the oil and gas sector; and (4) discussing of the inclusion of HDAPs in 2011–2014 emission inventories.

Health-damaging air pollutants emitted by the oil and gas industry 2011– 2014

The literature prior to 2015 provides ample evidence that health-damaging air pollutants are emitted by the oil and gas sector. The majority of the studies focus on emissions from upstream oil and gas activities (Adgate et al. 2014; Bloomdahl et al. 2014; Brown et al. 2014; Colborn et al. 2014; Edwards et al. 2014; Helmig et al. 2014; Macey et al. 2014; Oltmans et al. 2014; Roy et al. 2014; Thompson et al. 2014; Warneke et al. 2014; Zavala-Araiza et al. 2014; Zielinska et al. 2014), while a fewer number of studies evaluate HDAPs from midstream and downstream activities (Bozlaker et al. 2013; Lewis et al. 2012; Nelson 2013). Broadly, HDAPs associated with the upstream oil and gas sector include many naturally-occurring volatile compounds found in petroleum products; odorants added to processed natural gas; proppants used for well stimulation; products of combustion and incomplete combustion; and volatile and semi-volatile chemical additives used for routine well maintenance and/or well stimulation.

Petroleum reservoirs contain hundreds of petroleum hydrocarbons, which make up the largest fraction of petroleum. Petroleum hydrocarbons include compounds such as benzene, toluene, ethylbenzene, and xylene (BTEX), and various alkanes (e.g., n-hexane), many of which have known or suspected toxic effects. For example, benzene is a known human carcinogen and hematological toxicant, and chronic exposures to ethylbenzene, toluene, and xylene have been associated with carcinogenicity, neurotoxicity, nervous system effects, and/or reproductive toxicity (National Cancer Institute 2019; OEHHA 2019). Petroleum may also contain trace metals, some of which are hazardous to human health. Cadmium, lead, and nickel — trace metals commonly found in crude oil — are known human carcinogens and developmental and reproductive toxicants, and are associated with other toxic effects (Lord 1991).

Many petroleum hydrocarbons are volatile or semi-volatile compounds (VOC or SVOC, respectively) that are recognized as hazardous air pollutants (HAPs) in the United States under the Clean Air Act (CAA).[4] These emissions standards are intended to prevent adverse health risks (non-cancer and cancer) from specific source types, including from the oil and gas

---

[4] The Clean Air Act (CAA) [42 USC Section 7401 et seq. (1970)], passed in 1970 and last amended in 1990, is a federal law that gives the U.S. EPA broad authority to regulate air emissions from stationary and mobile sources, and to implement air pollution prevention and control programs nationwide (US EPA 2020). Sources of HAP emissions are controlled through a set of standards, as outlined in CAA Section 112 National Emission Standards for Hazardous Air Pollutants (NESHAP) (US EPA 2020).

PSE

industry (US EPA 2020). While some VOCs that comprise petroleum are directly associated with adverse health impacts (e.g., benzene), others (mainly non-methane VOCs) are precursors that lead to the secondary formation of ground-level ozone (e.g., VOCs, NO_x) and $PM_{2.5}$ (e.g., CO, NO_x), both of which are federally defined as criteria air pollutants and are associated with adverse respiratory impacts under specific exposure scenarios.

Literature published prior to 2015 found upstream oil and gas activities were associated with a variety of non-methane VOCs and HDAPs, including benzene, toluene, hexane, formaldehyde, carbon monoxide, nitrogen oxides, and particulate matter. Diesel-powered equipment or gas turbines used during drilling and well stimulation activities (e.g., hydraulic fracturing); flaring or the controlled burning of natural gas from flare stacks; and diesel trucks used to transport equipment and waste products, can impair local air quality by emitting incomplete combustion byproducts. The incomplete combustion of diesel and natural gas used as energy sources to facilitate oil and gas exploration and production results in localized increases of HDAPs, including formaldehyde, carbon monoxide, nitrogen oxides, and particulate matter. Exposure to diesel exhaust (sometimes referred to as diesel PM) and benzene near oil and gas sites is a recognized respiratory health hazard and known human carcinogen (McCawley 2013). Additionally, particulate matter and ozone are secondary pollutants, and can form as oil and gas-associated pollutants interact with other reactive compounds in the atmosphere, and as products of combustion from equipment, trucks, and flare stacks.

Relative to upstream sources, HDAP emissions from midstream and downstream sources are less understood. From 2011–2014, there were a handful of peer-reviewed studies that evaluated HDAP emissions from petroleum refineries in the United States (Bozlaker et al. 2013; Lewis et al. 2012; Nelson 2013). We did not identify any studies from 2011–2014 that evaluated HDAP emissions associated with midstream processes (transmission and storage). Similarly, we did not find any studies focused on HDAP emissions associated with other downstream sources from 2011–2014, including from distribution pipelines, metering and regulating stations, customer meters, gas stations, end-user appliances, or buildings.

The usage of catalysts in petroleum refineries is a potential source of toxic air contaminants from downstream processes (Lewis et al. 2012). Both solid and liquid catalysts are used in petroleum refining, with the vast majority of solid catalysts comprised of precious metals (e.g., platinum and rhenium), base metals (e.g., cobalt and nickel), and zeolites (e.g., aluminosilicates) (Lewis et al. 2012). Both liquid and solid catalysts are potential sources for human exposure via inhalation during loading and unloading activities, especially in the absence of proper emission control measures. Liquid catalysts may generate vapors, while solid catalysts can produce dust when disturbed.

For example, elevated levels of airborne silica are associated with fluidized-bed catalytic cracking (FCC) (Bozlaker et al. 2013). Petroleum refining, including the use of catalysts during

FCC, also releases "light lanthanoids,"[5] including lanthanum-rich coarse particulate matter (PM_{10}). Vanadium-enriched PM_{10} is associated with other downstream processes, including fuel oil combustion, mainly from ships, oil-fired boilers, and industrial power plants using heavy oil (Bozlaker et al. 2013).

Various compounds with strong odors occur naturally in petroleum reservoirs or are added to processed natural gas for leak detection and safety. Compounds with odors can adversely impact the physical and mental health of those experiencing odors, as well as interfere with daily activities and social well-being. Broadly, peer-reviewed studies have associated odors with acute physical symptoms such as headaches, nausea, eye and throat irritation, respiratory symptoms including wheezing, and psychosocial stress (Avery et al. 2004; Heaney et al. 2011; Horton et al. 2009; Schiffman et al. 1995, 2005).

Hydrogen sulfide (H_2S) is an odorant gas with a low odor threshold, which means its odor can be detected at low concentrations ranging from 8 to 130 parts per billion (ppb) (NRC, 2010). Most human organ systems are susceptible to the toxic effects of hydrogen sulfide, particularly mucus membranes, the central nervous system, the respiratory system, the cardiovascular system, and the gastrointestinal system (Reiffenstein et al. 1992). Exposure to hydrogen sulfide is associated with known acute health symptoms, including irritation of the eyes, nose and throat, nausea, vomiting, and headaches. Hydrogen sulfide is also a chemical asphyxiant that can cause death at relatively low concentrations (Schiffman et al. 2005; Wing et al. 2008).

Additionally, a series of thiol compounds containing a sulfhydryl group (R-SH) (i.e., mercaptans) are intentionally added to processed natural gas at low concentrations (e.g., 5–10 ppb), typically at the point where gas enters the distribution network. Without the addition of mercaptans or other organosulfur compounds, processed natural gas is odorless and undetectable without dedicated equipment. Overall, there is very little toxicological and epidemiological research on organosulfur compounds typically used for natural gas odorization, especially at chronic or sub-chronic, low-dose levels. Studies of mercaptan exposure in humans have been limited to case reports in occupational settings, however, direct adverse health effects included eye, dermal, and respiratory irritation, loss of a sense of smell, difficulty breathing, cyanosis, headache, disorientation similar to drunkenness, lung congestion, kidney damage, convulsions, and coma. Human health-based regulations exist only for two mercaptan constituents (ethyl and methyl mercaptan) that are not commonly used to odorize gas, though much conflicting evidence exists pertaining to specific mercaptan use. There is evidence that odorant-associated self-reported health effects can manifest at concentrations below standard ambient monitoring instrument detection limits, yet above the human odor threshold. While the literature published from 2011–2014 indicates that various

---

[5] Light lanthanoids include: 57 lanthanum (La), 58 cerium (Ce), 59 praseodymium (Pr) and 60 neodymium (Nd).

PSE

odorant compounds with health relevance are present in natural gas, limited data are available on emissions and exposures.

Numerous hazardous chemical additives are also used throughout all phases of the oil and gas supply chain. Some additives are either volatile and semi-volatile, and may present an airborne health risk (Hudgins 1992, 1994; Kelland 2014).

Studies on health-damaging air pollutants co-emitted with oil and gas-associated methane emissions

Estimates of methane and other greenhouse gas emissions (GHG) (e.g., CO_2) from the oil and gas sector comprised the majority of studies provided in the 2011–2014 literature. Relatively few studies reported emissions estimates of health-damaging air pollutants (HDAPs), along with methane, and even fewer quantified the relationship between methane and associated HDAPs. Even so, the limited evidence published from 2011–2014 suggests that HDAPs (i.e., non-methane VOCs) are often co-emitted with methane. Researchers may directly measure methane and various HDAPs concurrently, or rely on emission factors to estimate proportions of co-emitted air pollutants from a known, characterized source. Emissions ratios are commonly used for source attribution, and ratios of methane and VOCs can be used to identify oil and gas sources and combustion sources, including emissions from vehicle exhaust. Upstream oil and gas sources can be identified by evaluating ratios of methane and ethane, a secondary component in natural gas, or other short-chain hydrocarbons (i.e., C1-C5 alkanes). Several studies apply VOC emissions ratios to identify and distinguish between thermogenic (in particular oil and gas) methane emissions (Gilman et al. 2013; Koss et al. 2015; Pétron et al. 2014; Warneke et al. 2014).

For upstream oil and gas emissions, methane and 101 other chemicals were detected concurrently in the air near a Texas production site, including several HDAPs: benzene, 1,3-butadiene, carbon disulfide, carbonyl sulfide, chloromethane, tetrachloroethane, toluene, and xylenes (Rich et al. 2014). Significant positive correlations with methane were found for 15 VOCs, including ozone precursors (pentane, heptane, and butane) and several HDAPs: hexachlorobutadiene, tetrachloroethene (PCE), and 1,2,4-trichlorobenzene (Rich et al. 2014). In a companion study, elevated concentrations of methane (>3 ppm) and hydrogen sulfide (>4.7 ppb) were also detected at the fenceline of Texas production sites in the, although there was no explicit assessment of their association (Eapi et al. 2014).

A critical review conducted by Moore et al. (2014) summarizes the most commonly emitted pollutants reported in the peer-reviewed literature. This review identified methane as a major pollutant in every stage of the oil and natural gas lifecycle. In addition, pre-production (i.e., exploration, drilling, hydraulic fracturing, well completions) and production (e.g., processing) activities in the upstream sector — as well as end uses — were all identified as sources of non-methane VOCs, many of which qualify as HDAPs, including benzene, toluene, and particulate matter (Figure A.2). Oil and gas transmission, storage, and distribution infrastructure may also

PSE

emit HDAPs; however, the literature prior to 2015 provides insufficient evidence to support (or refute) this notion, highlighting a major research gap.

Figure A.2. Air pollutants emitted to the atmosphere during specific stages of the oil and natural gas life cycle. Source: Reproduced with permission[6] from Moore et al. (2014). BTEX: Benzene, toluene, ethylbenzene, and xylenes. NO$_X$: Nitrogen oxides. PM$_{2.5}$: Fine particulate matter (diameter <2.5 microns).

Studies evaluating health risks associated with oil and gas-associated health-damaging air pollutant emissions

Upstream

Upstream oil and gas production sites not only produce HDAPs, they also impact regional air quality (Field et al. 2014; Helmig et al. 2014; Pétron et al. 2012, 2014; Roy et al. 2014; Thompson et al. 2014). The majority of studies from 2011–2014 that assessed air quality as a function of distance found that concentrations of various hazardous and other air pollutants can be even higher in close proximity to active oil and gas development (Brown et al. 2014; Colborn et al. 2014; Macey et al. 2014; McKenzie et al. 2012). Intermittent spikes of emissions from oil and gas activities and equipment have also been observed (Allen 2014a; Brown et al. 2014), which may

---

PSE

have a limited influence on regional air pollutant concentrations but are likely to be associated with increased exposures to populations in close proximity to emission sources. As such, studies that focus on regional concentrations of air pollutants associated with oil and gas development may underestimate low- to moderate-level chronic exposures (Pétron et al. 2014).

Few studies compared observed air pollutant concentrations to health-based guidance values. Macey et al. (2014) analyzed air samples from locations in five different states using a community- based monitoring approach. They found that concentrations of eight volatile chemicals, including benzene, formaldehyde, hexane, and hydrogen sulfide, exceeded federal guidelines (ATSDR minimal risk levels (MRLs)) (ATSDR, 2018) and U.S. EPA Integrated Risk Information System (IRIS) cancer risk levels at multiple sampling sites in close proximity to production infrastructure. Residents who collected the samples reported a range of common health symptoms plausibly attributable to HDAP exposure, including headaches, dizziness or light-headedness, irritated, burning, or running nose, nausea, and sore or irritated throat (Macey et al. 2014).

In an industry-funded study, Bunch et al. (2014) compared volatile organic compound (VOC) concentration data from air quality monitors at six locations in the Barnett Shale in Texas, using state and federal health-based guidance values. The authors concluded that shale gas activities did not result in community-wide exposures to VOCs at concentrations that would pose a health concern. However, the use of regional air quality monitoring data in this study preclude the evaluation of exposures to more local air quality impacts such as those documented by Macey et al. (2014).

A single peer-reviewed health risk assessment focused on upstream oil and gas sites was published from 2011–2014. McKenzie et al. (2012) found both noncancer health risks associated with subchronic HDAP exposures and lifetime cancer risks were greater for residents living within ½ mile (805 m) from oil and gas wells, as compared to those living beyond ½ mile. Increased noncancer health risks were observed specifically for respiratory, neurological, and hematological target organ systems. Increased risk was driven primarily by exposure to trimethylbenzenes, xylenes, and aliphatic hydrocarbons; slightly elevated excess lifetime cancer risk estimates were also driven by benzene exposure (McKenzie et al. 2012).

Peer-reviewed public health studies published from 2011–2014 primarily used self-reported data on symptoms or other health outcomes (Saberi et al. 2014; Steinzor et al. 2013), as opposed to relying on more robust outcome datasets (e.g., electronic medical records). However, one peer-reviewed analytical epidemiological study was published between 2011 and 2014 (McKenzie et al. 2014). In a retrospective cohort study in Colorado, McKenzie et al. (2014) examined the association between exposure to upstream oil and gas development and adverse birth outcomes. Risk for congenital heart defects was 30% greater among infants born to mothers who lived in the highest exposure category compared to those with no wells within a 16 km (52,493 ft) radius of maternal residence. Infants born to mothers in the highest density

PSE

of gas development also had twice the odds of being born with neural tube defects as compared to infants born to mothers living with no wells within a 16 km (52,493 ft) radius (McKenzie et al. 2014). Exposure was inversely associated with preterm birth and a positive association of small magnitude was observed with fetal growth. No association was observed for oral clefts.

Midstream and downstream

As described above, Moore et al. (2014) noted an absence of studies examining HDAPs in the midstream and downstream oil and gas supply chain. As such, little research had been conducted on the health risks associated with exposure to HDAPs from midstream and downstream sources from 2011–2014, with one exception. Handling of heavy metal catalysts used in the oil refining process produced airborne concentrations of cobalt and chromium at levels above the Occupational Safety and Health Administration's (OSHA) established permissible exposure limits (PEL), as well as above Threshold Limit Values (TLVs) established by the American Conference of Governmental Industrial Hygienists (ACGIH) (Lewis et al. 2012).

Oil and gas-associated health-damaging air pollutant emissions in emission inventories

The U.S. Environmental Protection Agency (EPA) publishes an emission inventory every three years, referred to as the National Emissions Inventory (NEI) (US EPA 2016). It provides detailed emissions data from point sources, nonpoint sources, on-road and nonroad sources, and event sources. The inventory encompasses the emission of criteria air pollutant precursors, criteria air pollutants (CAP), and hazardous air pollutants (HAPs). More specifically, the NEI considers emissions information for the six CAPs and 187 federally-listed HAPs (US EPA 2016). Emissions data are compiled from local, tribal, and state industries and researchers, however, there can be gaps in the data if information is not received from industry facilities and in such cases, can be filled with other data. The NEI reports on the major sources of pollutant emissions as well as the change in key pollutant emissions compared to the prior NEI report.

The 2014 NEI included reporting categories for exploration and production activities, conventional and unconventional processes, specific equipment, and equipment counts in their emissions calculations. Table A.2 below (sourced from US EPA 2014 NEI Version 1.0), summarizes the percent change of CAPs, VOCs, and HAPs, organized by anthropogenic and biogenic sources for the continental U.S., Alaska, Hawai'i, tribal lands, and territories (note that values in the table do not include off-shore emissions). The greatest total emissions in 2014 were for CO, PM_{10}, and VOCs. Comparing 2011 to 2014, reductions were seen in all categories of pollutants except for PM_{10}, with SO_2 accounting for the greatest reduction in emissions between 2011 and 2014 (reduction of 26%) (US EPA EIAG 2017). According to the report, the decrease in HAP emissions between 2011 and 2014 is predominantly (~50%) a function of changing methods used to estimate biogenic methanol (US EPA EIAG 2017). The HDAP species with the greatest reductions were toluene, hexane, xylene, and benzene from mobile sources.

PSE

Table A.2. Total Emissions, All Sectors, 2011 NEI v2 vs. 2014 NEI v1. Source: US EPA 2014 NEI Version 1.0.

Pollutant	Anthropogenic, x1000 Tons (Man-made)		Biogenic, x1000 Tons (Natural)		Total, x1000 Tons		Percent Change from 2011 to 2014
	2011	2014	2011	2014	2011	2014	
CO	75,760	63,252	6,528	6,635	82,288	69,907	-15
NH₃	4,316	3,869	NA	22.2	4,316	3,891	-10
NOₓ	14,574	12,643	1,018	903	15,592	13,546	-13
PM₁₀	20,907	24,506	NA	NA	20,907	24,506	17
PM₂.₅	6,306	6,223	NA	NA	6,306	6,223	-1.3
SO₂	6,557	4,812	NA	NA	6,557	4,812	-26
VOCs	18,169	16,478	39,653	38,679	57,822	55,157	-5
Pb	0.80	0.73	NA	NA	0.80	0.73	-9
BC (same as EC)	567	446	NA	NA	567	446	-21
Total HAPs	3,107	3,020	5,968	5,295	9,074	8,315	-8

CO: carbon monoxide, **NH₃**: ammonia, **NOₓ**: nitrogen oxides, **PM₁₀**: particulate matter less than 10 microns, **PM₂.₅**: particulate matter less than 2.5 microns, **SO₂**: sulfur dioxide, **VOCs**: volatile organic compounds, **Pb**: lead, **BC**: black carbon, **EC**: elemental carbon, **HAPs**: hazardous air pollutants.

Considering only stationary source emissions, oil and gas production (28%) and natural gas consumption (22%) were significant sources of NOₓ emissions. Oil and gas production (48%) were a significant contributor to VOC emissions, and natural gas combustion (40%) was a major source of black carbon emissions (US EPA EIAG, 2017).
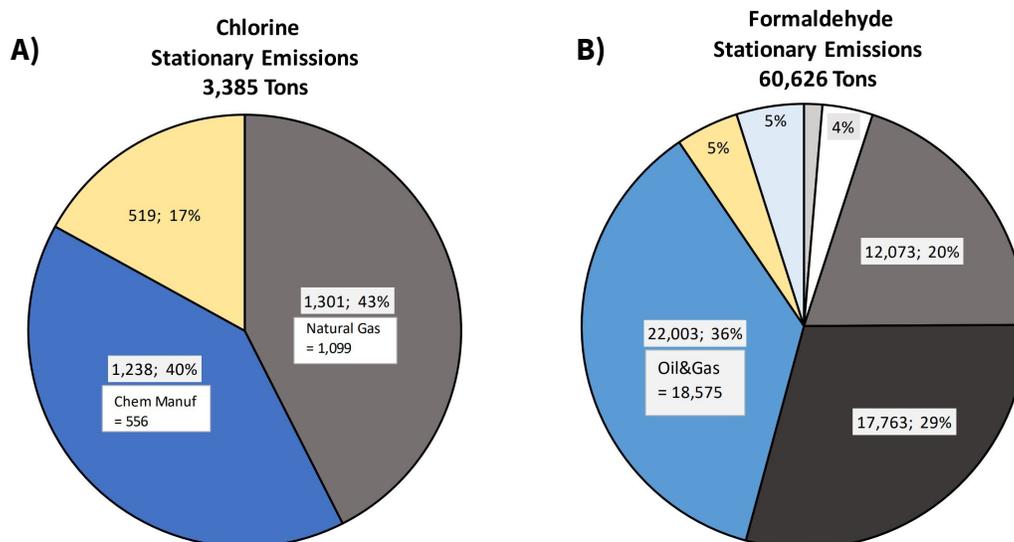
Figure A.3. (A) Chlorine stationary emissions and (B) formaldehyde stationary emissions (US EPA EIAG 2017).

The NEI also reported on a select few HAPs that were identified as 2011 National Air Toxics Assessment health risk drivers: acrolein (non-cancer), diesel-PM10 (non-cancer), formaldehyde (cancer), and chlorine (non-cancer). Detailed analysis of the stationary sources for these four pollutants showed that industrial boilers using natural gas (43%) are the major contributor to chlorine emissions (Figure A.3-A). Oil and gas production (36%) contributes significantly to emissions (refer to Figure A.3-A) (US EPA EIAG 2017).

The petroleum and related industries category was a significant overall source of HDAP emissions in 2014 (Figure A.4-A). Comparing the 2011 NEI to the 2014 NEI, emissions for the petroleum and related industries categories actually increased for several important HDAPs, including VOCs, NOx, CO, and ammonia (Figure A.4-B).
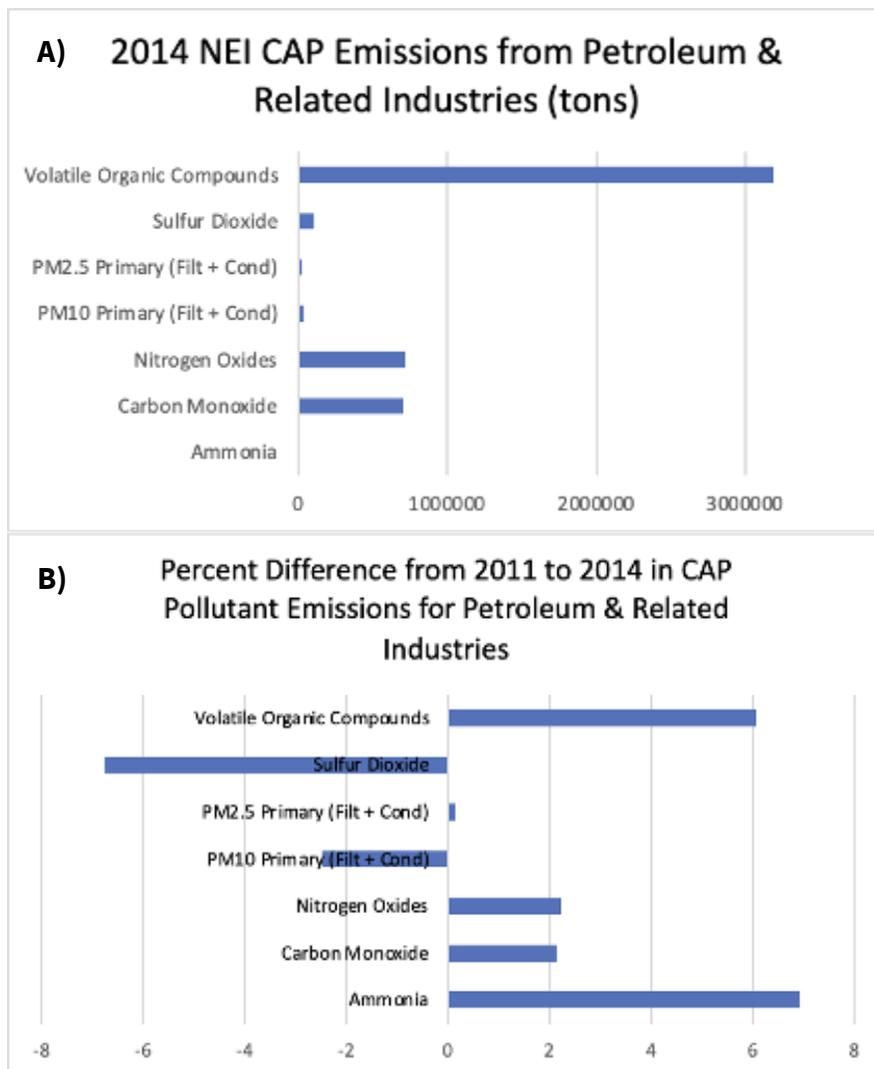
Figure A.4. (A) Total CAP emissions in 2014 and (B) percent difference of CAP emissions 2014 to 2011 (Source: US EPA 2016).

For the oil and gas production segment (upstream emissions), VOCs were the dominant emissions, followed by nitrogen oxides (NO_x) and carbon monoxide (CO). (Although not reported in the NEI as such, VOCs and NO_x can also impact air quality as major ozone precursors.) For the petroleum refineries segment (downstream emissions), nitrous oxides are the dominant emissions, followed by sulfur dioxide (SO_2), VOCs, and CO. National emissions summaries are shown below (Figure A.5). VOCs were the top pollutants emitted from oil and gas production, with nitrogen oxides and carbon monoxide being in the top three pollutant emissions by quantity. There was greater variation between states with regard to the top species emitted by petroleum refineries. However, the predominant top three chemicals were nitrogen oxides, sulfur dioxide, VOCs, and carbon monoxide. Values for each of the figures were extracted from the EPA's NEI data queries function on the EPA NEI website (US EPA 2016).

PSE
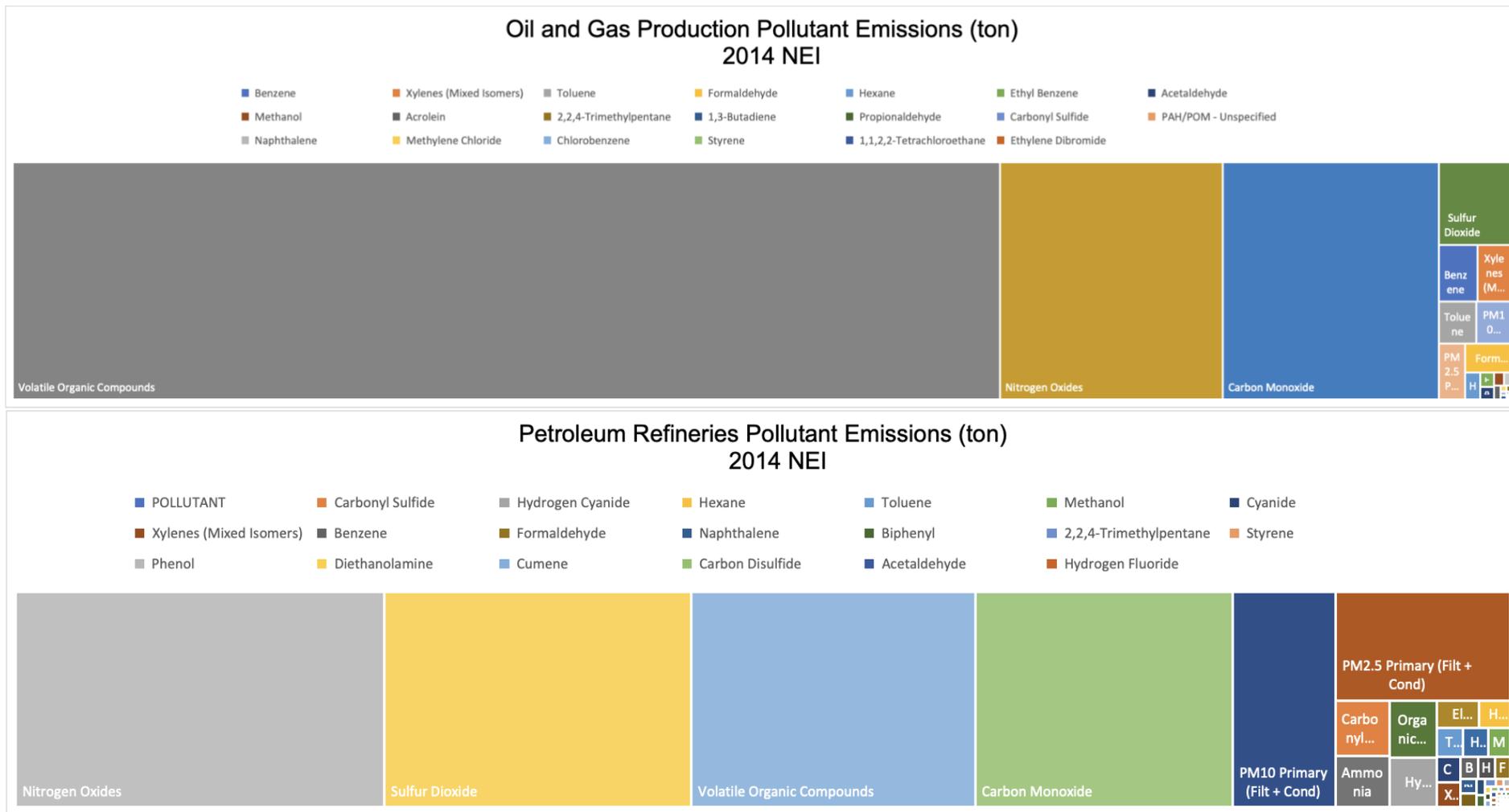
Figure A.5. (A) 2014 Oil and gas production emissions and (B) 2014 Petroleum refineries emissions (Source: US EPA, 2016).

Health-damaging air pollutants: Challenges, research gaps, and recommendations

Based on the evidence reviewed herein, the literature published from 2011–2014 is most complete for upstream oil and gas sources. Oil- and gas-production activities were associated with a variety of non-methane VOCs and health damaging air pollutants (HDAPs), including benzene, toluene, and other alkanes (n-hexane, propane, ethane), formaldehyde, carbon monoxide, nitrogen oxides, and particulate matter. Furthermore, many HDAPs were specifically co-emitted with methane, including benzene, 1,3-butadiene, carbon disulfide, carbonyl sulfide, chloromethane, tetrachloroethane, toluene, xylene, hexachlorobutadiene, tetrachloroethene (PCE), 1,2,4-trichlorobenzene, and chloroform.

This review identified few studies focused on downstream sources (distribution) and no studies focused on HDAP emissions from midstream sources (transmission and storage). The few studies that did evaluate HDAP emissions from downstream sources focused on petroleum refineries and found unloading and loading activities to be a significant source of cobalt, chromium, silica, and metal-enriched coarse particulate matter (PM_{10}) emissions, including lanthanum. Non-routine operations at refineries are also a significant source of HDAPs, but are seemingly underestimated and underreported HDAP sources.

Several knowledge gaps emerge from this review of the 2011–2014 literature on HDAP emissions from the oil and gas sector. First, relative to the upstream segment, there has been little research on HDAP emissions from midstream (e.g., transmission pipelines, compressor or pumping stations, storage facilities) and downstream (e.g., distribution systems, fueling stations, buildings) sources. Second, there are substantial limits in the spatial (e.g., local vs. regional) and temporal (e.g., capturing short-lived, high emissions events vs. long-term averages) resolution of oil and gas HDAP emissions. Finally, the National Emissions Inventory from 2011 and 2014 demonstrated that the oil and gas industry is both a considerable source of HDAP emissions and that some HDAPs have increased in the most recent inventory (e.g., VOCs, which include some HDAPs, nitrogen oxides, carbon monoxide, and ammonia, with a slight increase in $PM_{2.5}$ emissions).

More concerning with respect to human health is the paucity of epidemiologic investigations of exposure to HDAP emissions from the oil and gas sector, with only a few studies in 2011–2014 focused on upstream oil and gas development. Local exposures remain generally uncharacterized, although modeling and preliminary studies have indicated that intermittent emissions spikes may produce health risks for local populations. Building a robust body of knowledge on the health risks associated with oil and gas HDAP emissions will thus likely entail greater spatial and temporal resolution of emissions events in subsequent iterations of research programs.

PSE

References

Adgate, J. L., Goldstein, B. D., & McKenzie, L. M. (2014). Potential Public Health Hazards, Exposures and Health Effects from Unconventional Natural Gas Development. *Environmental Science & Technology*, *48*(15), 8307–8320. https://doi.org/10.1021/es404621d.

Allen, D. (2016). Attributing Atmospheric Methane to Anthropogenic Emission Sources. *Accounts of Chemical Research*, *49*(7), 1344–1350. https://doi.org/10.1021/acs.accounts.6b00081.

Allen, D. T. (2014a). Atmospheric Emissions and Air Quality Impacts from Natural Gas Production and Use. *Annual Review of Chemical and Biomolecular Engineering*, *5*(1), 55–75. https://doi.org/10.1146/annurev-chembioeng-060713-035938.

Allen, D. T. (2014b). Methane emissions from natural gas production and use: Reconciling bottom-up and top-down measurements. *Current Opinion in Chemical Engineering*, *5*, 78–83. https://doi.org/10.1016/j.coche.2014.05.004.

Allen, D. T., Torres, V. M., Thomas, J., Sullivan, D. W., Harrison, M., Hendler, A., Herndon, S. C., Kolb, C. E., Fraser, M. P., Hill, A. D., Lamb, B. K., Miskimins, J., Sawyer, R. F., & Seinfeld, J. H. (2013). Measurements of methane emissions at natural gas production sites in the United States. *Proceedings of the National Academy of Sciences*, *110*(44), 17768–17773. https://doi.org/10.1073/pnas.1304880110.

Alvarez, R. A., Pacala, S. W., Winebrake, J. J., Chameides, W. L., & Hamburg, S. P. (2012). Greater focus needed on methane leakage from natural gas infrastructure. *Proceedings of the National Academy of Sciences*, *109*(17), 6435–6440. https://doi.org/10.1073/pnas.1202407109.

ATSDR (Agency for Toxic Substances & Disease Registry). (2018). *Minimal Risk Levels (MRLs)*. https://www.atsdr.cdc.gov/mrls/pdfs/atsdr_mrls.pdf.

Avery, R. C., Wing, S., Marshall, S. W., & Schiffman, S. S. (2004). Odor from industrial hog farming operations and mucosal immune function in neighbors. *Archives of Environmental Health*, *59*(2), 101–108. https://doi.org/10.3200/AEOH.59.2.101-108.

Balcombe, P., Speirs, J. F., Brandon, N. P., & Hawkes, A. D. (2018). Methane emissions: Choosing the right climate metric and time horizon. *Environmental Science: Processes & Impacts*, *20*(10), 1323–1339. https://doi.org/10.1039/C8EM00414E.

Bloomdahl, R., Abualfaraj, N., Olson, M., & Gurian, P. L. (2014). Assessing worker exposure to inhaled volatile organic compounds from Marcellus Shale flowback pits. *Journal of Natural Gas Science and Engineering*, *21*, 348–356. https://doi.org/10.1016/j.jngse.2014.08.018.

Bozlaker, A., Buzcu-Güven, B., Fraser, M. P., & Chellam, S. (2013). Insights into PM10 sources in Houston, Texas: Role of petroleum refineries in enriching lanthanoid metals during episodic emission events. *Atmospheric Environment*, *69*, 109–117. https://doi.org/10.1016/j.atmosenv.2012.11.068.

Brandt, A. R., Heath, G. A., Kort, E. A., O'Sullivan, F., Pétron, G., Jordaan, S. M., Tans, P., Wilcox, J., Gopstein, A. M., Arent, D., Wofsy, S., Brown, N. J., Bradley, R., Stucky, G. D., Eardley, D., & Harriss, R. (2014). Methane Leaks from North American Natural Gas Systems. *Science*, *343*(6172), 733–735. https://doi.org/10.1126/science.1247045.

Brown, D., Weinberger, B., Lewis, C., & Bonaparte, H. (2014). Understanding exposure from natural gas drilling puts current air standards to the test. *Reviews on Environmental Health*, *29*(4), 277–292. https://doi.org/10.1515/reveh-2014-0002.

PSE

Bunch, A. G., Perry, C. S., Abraham, L., Wikoff, D. S., Tachovsky, J. A., Hixon, J. G., Urban, J. D., Harris, M. A., & Haws, L. C. (2014). Evaluation of impact of shale gas operations in the Barnett Shale region on volatile organic compounds in air and potential human health risks. *Science of The Total Environment*, *468–469*, 832–842. https://doi.org/10.1016/j.scitotenv.2013.08.080.

Burnham, A., Han, J., Clark, C. E., Wang, M., Dunn, J. B., & Palou-Rivera, I. (2012). Life-Cycle Greenhouse Gas Emissions of Shale Gas, Natural Gas, Coal, and Petroleum. *Environmental Science & Technology*, *46*(2), 619–627. https://doi.org/10.1021/es201942m.

Cathles, L. M. (2012). Assessing the greenhouse impact of natural gas: GREENHOUSE IMPACT OF NATURAL GAS. *Geochemistry, Geophysics, Geosystems*, *13*(6), n/a-n/a. https://doi.org/10.1029/2012GC004032.

Caulton, D. R., Shepson, P. B., Santoro, R. L., Sparks, J. P., Howarth, R. W., Ingraffea, A. R., Cambaliza, M. O. L., Sweeney, C., Karion, A., Davis, K. J., Stirm, B. H., Montzka, S. A., & Miller, B. R. (2014). Toward a better understanding and quantification of methane emissions from shale gas development. *Proceedings of the National Academy of Sciences*, *111*(17), 6237–6242. https://doi.org/10.1073/pnas.1316546111.

Chambers. (2004). Optical Measurement Technology for Fugitive Emissions from Upstream Oil and Gas Facilities. 79.

Clearstone Engineering. (2002). Identification and Evaluation of Opportunities to Reduce Methane Losses at Four Gas Processing Plants. Gas Technology Institute.

Colborn, T., Schultz, K., Herrick, L., & Kwiatkowski, C. (2014). An Exploratory Study of Air Quality Near Natural Gas Operations. *Human and Ecological Risk Assessment: An International Journal*, *20*(1), 86–105. https://doi.org/10.1080/10807039.2012.749447.

Cormack, J. (2007). Energy Management Workshop for Upstream and Midstream Operations: Increasing Revenue through Process Optimization & Methane Emissions Reduction. Global Methane Initiative.

Eapi, G. R., Sabnis, M. S., & Sattler, M. L. (2014). Mobile measurement of methane and hydrogen sulfide at natural gas production site fence lines in the Texas Barnett Shale. *Journal of the Air & Waste Management Association*, *64*(8), 927–944. https://doi.org/10.1080/10962247.2014.907098.

Edwards, P. M., Brown, S. S., Roberts, J. M., Ahmadov, R., Banta, R. M., deGouw, J. A., Dubé, W. P., Field, R. A., Flynn, J. H., Gilman, J. B., Graus, M., Helmig, D., Koss, A., Langford, A. O., Lefer, B. L., Lerner, B. M., Li, R., Li, S.-M., McKeen, S. A., … Zamora, R. (2014). High winter ozone pollution from carbonyl photolysis in an oil and gas basin. *Nature*, *514*(7522), 351–354. https://doi.org/10.1038/nature13767.

Field, R. A., Soltis, J., & Murphy, S. (2014). Air quality concerns of unconventional oil and natural gas production. *Environ. Sci.: Processes Impacts*, *16*(5), 954–969. https://doi.org/10.1039/C4EM00081A.

Gilman, J. B., Lerner, B. M., Kuster, W. C., & de Gouw, J. A. (2013). Source Signature of Volatile Organic Compounds from Oil and Natural Gas Operations in Northeastern Colorado. *Environmental Science & Technology*, *47*(3), 1297–1305. https://doi.org/10.1021/es304119a.

Harrison, M. R., Galloway, K. E., Hendler, A., Shires, T. M., Foss, M., Thomas, J., & Spinhirne, J. (2011). *Natural Gas Indsutr Methane Emission Factor Improvement Study*. Prepared for Lisa Hanle, United States Environmental Protection Agency. https://dept.ceer.utexas.edu/ceer/GHG/files/FReports/XA_83376101_Final_Report.pdf.

Harrison, M. R., Shires, T. M., Wessels, J. K., & Michael, R. (n.d.). *Methane Emissions from the Natural Gas Industry*.

Harriss, R., Alvarez, R. A., Lyon, D., Zavala-Araiza, D., Nelson, D., & Hamburg, S. P. (2015). Using Multi-Scale Measurements to Improve Methane Emission Estimates from Oil and Gas Operations in the Barnett Shale Region, Texas. *Environmental Science & Technology*, *49*(13), 7524–7526. https://doi.org/10.1021/acs.est.5b02305.

PSE

Heaney, C. D., Wing, S., Campbell, R. L., Caldwell, D., Hopkins, B., Richardson, D., & Yeatts, K. (2011). Relation between malodor, ambient hydrogen sulfide, and health in a community bordering a landfill. *Environmental Research*, *111*(6), 847–852. https://doi.org/10.1016/j.envres.2011.05.021.

Heath, G. A., O'Donoughue, P., Arent, D. J., & Bazilian, M. (2014). Harmonization of initial estimates of shale gas life cycle greenhouse gas emissions for electric power generation. *Proceedings of the National Academy of Sciences*, *111*(31), E3167–E3176. https://doi.org/10.1073/pnas.1309334111.

Heath, G., Warner, E., Steinberg, D., & Brandt, A. (2015). Estimating US Methane Emissions from the Natural Gas Supply Chain. Approaches, Uncertainties, Current Estimates, and Future Studies. National Renewable Energy Lab.(NREL), Golden, CO (United States).

Helmig, D., Thompson, C. R., Evans, J., Boylan, P., Hueber, J., & Park, J.-H. (2014). Highly Elevated Atmospheric Levels of Volatile Organic Compounds in the Uintah Basin, Utah. *Environmental Science & Technology*, *48*(9), 4707–4715. https://doi.org/10.1021/es405046r.

Horton, R. A., Wing, S., Marshall, S. W., & Brownley, K. A. (2009). Malodor as a Trigger of Stress and Negative Mood in Neighbors of Industrial Hog Operations. *American Journal of Public Health*, *99*(Suppl 3), S610–S615. https://doi.org/10.2105/AJPH.2008.148924.

Howarth, R. W., Santoro, R., & Ingraffea, A. (2011). Methane and the greenhouse-gas footprint of natural gas from shale formations: A letter. *Climatic Change*, *106*(4), 679–690. https://doi.org/10.1007/s10584-011-0061-5.

Howarth, R. W., Santoro, R., & Ingraffea, A. (2012). Venting and leaking of methane from shale gas development: Response to Cathles et al. *Climatic Change*, *113*(2), 537–549. https://doi.org/10.1007/s10584-012-0401-0.

Hsu, Y.-K., VanCuren, T., Park, S., Jakober, C., Herner, J., FitzGibbon, M., Blake, D. R., & Parrish, D. D. (2010). Methane emissions inventory verification in southern California. *Atmospheric Environment*, *44*(1), 1–7.

Hudgins, C. M. (1992). Chemical Treatments and Usage in Offshore Oil and Gas Production Systems. *Journal of Petroleum Technology*, *44*(05), 604–611. https://doi.org/10.2118/22477-PA.

Hudgins, C. M. (1994). Chemical Use in North Sea Oil and Gas E&P. *Journal of Petroleum Technology*, *46*(01), 67–74. https://doi.org/10.2118/24106-PA.

Innovative Environmental Solutions, Inc. (2015). *A COMPARATIVE ANALYSIS: METHANE EMISSION STUDIES OF NATURAL GAS INDUSTRY OPERATIONS*. https://www.ingaa.org/File.aspx?id=24463.

IPCC (Intergovernmental Panel on Climate Change). (2019). 2018: Global Warming of 1.5°C. An IPCC Special Report on the impacts of global warming of 1.5°C above pre-industrial levels and related global greenhouse gas emission pathways, in the context of strengthening the global response to the threat of climate change, sustainable development, and efforts to eradicate poverty. https://www.ipcc.ch/sr15/download/.

Jeong, S., Millstein, D., & Fischer, M. L. (2014). Spatially Explicit Methane Emissions from Petroleum Production and the Natural Gas System in California. *Environmental Science & Technology*, *48*(10), 5982–5990. https://doi.org/10.1021/es4046692.

Jiang, M., Michael Griffin, W., Hendrickson, C., Jaramillo, P., VanBriesen, J., & Venkatesh, A. (2011). Life cycle greenhouse gas emissions of Marcellus shale gas. *Environmental Research Letters*, *6*(3), 034014. https://doi.org/10.1088/1748-9326/6/3/034014.

Kang, M., Kanno, C. M., Reid, M. C., Zhang, X., Mauzerall, D. L., Celia, M. A., Chen, Y., & Onstott, T. C. (2014). Direct measurements of methane emissions from abandoned oil and gas wells in Pennsylvania. *Proceedings of the National Academy of Sciences*, *111*(51), 18173–18177. https://doi.org/10.1073/pnas.1408315111.

PSE

Karion, A., Sweeney, C., Pétron, G., Frost, G., Michael Hardesty, R., Kofler, J., Miller, B. R., Newberger, T., Wolter, S., Banta, R., Brewer, A., Dlugokencky, E., Lang, P., Montzka, S. A., Schnell, R., Tans, P., Trainer, M., Zamora, R., & Conley, S. (2013). Methane emissions estimate from airborne measurements over a western United States natural gas field: CH4 EMISSIONS OVER A NATURAL GAS FIELD. *Geophysical Research Letters*, *40*(16), 4393–4397. https://doi.org/10.1002/grl.50811.

Katzenstein, A. S., Doezema, L. A., Simpson, I. J., Blake, D. R., & Rowland, F. S. (2003). Extensive regional atmospheric hydrocarbon pollution in the southwestern United States. *Proceedings of the National Academy of Sciences*, *100*(21), 11975–11979.

Kelland, M. A. (2014). Production Chemicals for the Oil and Gas Industry, Second Edition. CRC Press.

Koss, A. R., de Gouw, J., Warneke, C., Gilman, J. B., Lerner, B. M., Graus, M., Yuan, B., Edwards, P., Brown, S. S., Wild, R., Roberts, J. M., Bates, T. S., & Quinn, P. K. (2015). Photochemical aging of volatile organic compounds associated with oil and natural gas extraction in the Uintah Basin, UT, during a wintertime ozone formation event. *Atmospheric Chemistry and Physics Discussions*, *15*(5), 6403–6444. https://doi.org/10.5194/acpd-15-6403-2015.

Levi, M. A. (2012). Comment on "Hydrocarbon emissions characterization in the Colorado Front Range: A pilot study" by Gabrielle Pétron et al.: COMMENTARY. *Journal of Geophysical Research: Atmospheres*, *117*(D21), n/a-n/a. https://doi.org/10.1029/2012JD017686.

Lewis, R. C., Gaffney, S. H., Le, M. H., Unice, K. M., & Paustenbach, D. J. (2012). Airborne concentrations of metals and total dust during solid catalyst loading and unloading operations at a petroleum refinery. *International Journal of Hygiene and Environmental Health*, *215*(5), 514–521. https://doi.org/10.1016/j.ijheh.2011.10.006.

Lord, C. J. (1991). Determination of trace metals in crude oil by inductively coupled plasma mass spectrometry with microemulsion sample introduction. *Analytical Chemistry*, *63*(15), 1594–1599. https://doi.org/10.1021/ac00015a018.

Macey, G. P., Breech, R., Chernaik, M., Cox, C., Larson, D., Thomas, D., & Carpenter, D. O. (2014). Air concentrations of volatile compounds near oil and gas production: A community-based exploratory study. *Environmental Health*, *13*(1), 82. https://doi.org/10.1186/1476-069X-13-82.

McCawley, M. (2013). Air, noise, and light monitoring results for assessing environmental impacts of horizontal gas well drilling operations (ETD-10-project). West Virginia Department of Environmental Protection. https://www.cdc.gov/niosh/nioshtic-2/20049534.html.

McGarry, J., Network, C. C. A., & Flamm, C. (2014). US Methane Leakage from Natural Gas Systems: A Literature Review.

McKenzie, L. M., Guo, R., Witter, R. Z., Savitz, D. A., Newman, L. S., & Adgate, J. L. (2014). Birth outcomes and maternal residential proximity to natural gas development in rural Colorado. *Environmental Health Perspectives*, *122*(4), 412–417. https://doi.org/10.1289/ehp.1306722.

McKenzie, L. M., Witter, R. Z., Newman, L. S., & Adgate, J. L. (2012). Human health risk assessment of air emissions from development of unconventional natural gas resources. *Science of The Total Environment*, *424*, 79–87. https://doi.org/10.1016/j.scitotenv.2012.02.018.

Methane from oil & gas – Methane Tracker 2020 – Analysis. (n.d.). IEA. Retrieved July 10, 2021, from https://www.iea.org/reports/methane-tracker-2020/methane-from-oil-gas.

Miller, S. M., Wofsy, S. C., Michalak, A. M., Kort, E. A., Andrews, A. E., Biraud, S. C., Dlugokencky, E. J., Eluszkiewicz, J., Fischer, M. L., Janssens-Maenhout, G., Miller, B. R., Miller, J. B., Montzka, S. A., Nehrkorn, T., & Sweeney, C. (2013). Anthropogenic emissions of methane in the United States. *Proceedings of the National Academy of Sciences*, *110*(50), 20018–20022. https://doi.org/10.1073/pnas.1314392110.

PSE

Mønster, J. G., Samuelsson, J., Kjeldsen, P., Rella, C. W., & Scheutz, C. (2014). Quantifying methane emission from fugitive sources by combining tracer release and downwind measurements – A sensitivity analysis based on multiple field surveys. *Waste Management*, *34*(8), 1416–1428. https://doi.org/10.1016/j.wasman.2014.03.025.

Moore, C. W., Zielinska, B., Pétron, G., & Jackson, R. B. (2014). Air Impacts of Increased Natural Gas Acquisition, Processing, and Use: A Critical Review. *Environmental Science & Technology*, *48*(15), 8349–8359. https://doi.org/10.1021/es4053472.

Myhre, G., Shindell, D., Bréon, F.-M., Collins, W., Fuglestvedt, J., Huang, J., Koch, D., Lamarque, J.-F., Lee, D., Mendoza, B., Nakajima, T., Robock, A., Stephens, G., Takemura, T., & Zhang, H. (2013). Anthropogenic and Natural Radiative Forcing. In Stocker, T.F., D. Qin, G.-K. Plattner, M. Tignor, S. K. Allen, J. Boschung, A. Nauels, Y. Xia, V. Bex, & P. M. Midgley (Eds.), *Climate Change 2013 – The Physical Science Basis: Working Group I Contribution to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change* (pp. 659–740). Cambridge University Press. https://doi.org/DOI: 10.1017/CBO9781107415324.018.

National Cancer Institute. (2019, January 14). *Cancer-Causing Substances—Benzene*. https://www.cancer.gov/about-cancer/causes-prevention/risk/substances/benzene.

National Gas Machinery Laboratory, Clearstone Engineering, & Innovative Environmental Solutions, Inc. (2006). EPA Phase II Aggregate Site Report Cost-Effective Directed Inspection and Maintenance Control Opportunities at Five Gas Processing Plants and Upstream Gatllering Compressor Stations and Well Sites. Prepared for Roger Fernandez, US EPA Natural Gas STAR Program.

Nelson, T. P. (2013). An examination of historical air pollutant emissions from us petroleum refineries. *Environmental Progress & Sustainable Energy*, *32*(2), 425–432. https://doi.org/10.1002/ep.11713.

Nisbet, E. G., Dlugokencky, E. J., Manning, M. R., Lowry, D., Fisher, R. E., France, J. L., Michel, S. E., Miller, J. B., White, J. W. C., & Vaughn, B. (2016). Rising atmospheric methane: 2007–2014 growth and isotopic shift. *Global Biogeochemical Cycles*, *30*(9), 1356–1370.

NRC (National Research Council). (2010). Acute Exposure Guideline Levels for Selected Airborne Chemicals: Volume 9. https://doi.org/10.17226/12978.

OEHHA (Office of Environmental Health Hazard Assessment). (2019, November). *Acute, 8-hour and Chronic Reference Exposure Level (REL) Summary*. https://oehha.ca.gov/air/general-info/oehha-acute-8-hour-and-chronic-reference-exposure-level-rel-summary.

Oltmans, S., Schnell, R., Johnson, B., Pétron, G., Mefford, T., & Neely, R. (2014). Anatomy of wintertime ozone associated with oil and natural gas extraction activity in Wyoming and Utah. *Elementa: Science of the Anthropocene*, *2*, 000024. https://doi.org/10.12952/journal.elementa.000024.

Peischl, J., Ryerson, T. B., Brioude, J., Aikin, K. C., Andrews, A. E., Atlas, E., Blake, D., Daube, B. C., de Gouw, J. A., Dlugokencky, E., Frost, G. J., Gentner, D. R., Gilman, J. B., Goldstein, A. H., Harley, R. A., Holloway, J. S., Kofler, J., Kuster, W. C., Lang, P. M., … Parrish, D. D. (2013). Quantifying sources of methane using light alkanes in the Los Angeles basin, California: SOURCES OF METHANE IN L.A. *Journal of Geophysical Research: Atmospheres*, *118*(10), 4974–4990. https://doi.org/10.1002/jgrd.50413.

Pétron, G., Frost, G. J., Trainer, M. K., Miller, B. R., Dlugokencky, E. J., & Tans, P. (2013). Reply to comment on "Hydrocarbon emissions characterization in the Colorado Front Range-A pilot study" by Michael A. Levi: PéTRON ET AL. *Journal of Geophysical Research: Atmospheres*, *118*(1), 236–242. https://doi.org/10.1029/2012JD018487.

PSE

Pétron, G., Frost, G., Miller, B. R., Hirsch, A. I., Montzka, S. A., Karion, A., Trainer, M., Sweeney, C., Andrews, A. E., Miller, L., Kofler, J., Bar-Ilan, A., Dlugokencky, E. J., Patrick, L., Moore, C. T., Ryerson, T. B., Siso, C., Kolodzey, W., Lang, P. M., … Tans, P. (2012). Hydrocarbon emissions characterization in the Colorado Front Range: A pilot study. *Journal of Geophysical Research: Atmospheres*, *117*(D4). https://doi.org/10.1029/2011JD016360.

Pétron, G., Karion, A., Sweeney, C., Miller, B. R., Montzka, S. A., Frost, G. J., Trainer, M., Tans, P., Andrews, A., Kofler, J., Helmig, D., Guenther, D., Dlugokencky, E., Lang, P., Newberger, T., Wolter, S., Hall, B., Novelli, P., Brewer, A., … Schnell, R. (2014). A new look at methane and nonmethane hydrocarbon emissions from oil and natural gas operations in the Colorado Denver-Julesburg Basin: Hydrocarbon emissions in oil & gas basin. *Journal of Geophysical Research: Atmospheres*, *119*(11), 6836–6852. https://doi.org/10.1002/2013JD021272.

Picard, D. (2005). Modern Technolgies of Detection and Elimination of Methane Leakages from Natural Gas Systems.

Reiffenstein, R. J., Hulbert, W. C., & Roth, S. H. (1992). Toxicology of hydrogen sulfide. *Annual Review of Pharmacology and Toxicology*, *32*, 109–134. https://doi.org/10.1146/annurev.pa.32.040192.000545.

Rich, A., Grover, J. P., & Sattler, M. L. (2014). An exploratory study of air emissions associated with shale gas development and production in the Barnett Shale. *Journal of the Air & Waste Management Association*, *64*(1), 61–72. https://doi.org/10.1080/10962247.2013.832713.

Roscioli, J. R., Yacovitch, T. I., Floerchinger, C., Mitchell, A. L., Tkacik, D. S., Subramanian, R., Martinez, D. M., Vaughn, T. L., Williams, L., Zimmerle, D., Robinson, A. L., Herndon, S. C., & Marchese, A. J. (2015). Measurements of methane emissions from natural gas gathering facilities and processing plants: Measurement methods. *Atmospheric Measurement Techniques*, *8*(5), 2017–2035. https://doi.org/10.5194/amt-8-2017-2015.

Roy, A. A., Adams, P. J., & Robinson, A. L. (2014). Air pollutant emissions from the development, production, and processing of Marcellus Shale natural gas. *Journal of the Air & Waste Management Association*, *64*(1), 19–37. https://doi.org/10.1080/10962247.2013.826151.

Saberi, P., Propert, K., Powers, M., Emmett, E., & Green-McKenzie, J. (2014). Field Survey of Health Perception and Complaints of Pennsylvania Residents in the Marcellus Shale Region. *International Journal of Environmental Research and Public Health*, *11*(6), 6517–6527. https://doi.org/10.3390/ijerph110606517.

Schiffman, S. S., Miller, E. A., Suggs, M. S., & Graham, B. G. (1995). The effect of environmental odors emanating from commercial swine operations on the mood of nearby residents. *Brain Research Bulletin*, *37*(4), 369–375. https://doi.org/10.1016/0361-9230(95)00015-1.

Schiffman, S. S., Studwell, C. E., Landerman, L. R., Berman, K., & Sundy, J. S. (2005). Symptomatic Effects of Exposure to Diluted Air Sampled from a Swine Confinement Atmosphere on Healthy Human Subjects. *Environmental Health Perspectives*, *113*(5), 567–576. https://doi.org/10.1289/ehp.6814.

Shorter, J. H., McManus, J. B., Kolb, C. E., Allwine, E. J., Siverson, R., Lamb, B. K., Mosher, B. W., Harriss, R. C., Howard, T., & Lott, R. A. (1997). Collection of leakage statistics in the natural gas system by tracer methods. *Environmental Science & Technology*, *31*(7), 2012–2019.

Steinzor, N., Subra, W., & Sumi, L. (2013). Investigating Links between Shale Gas Development and Health Impacts through a Community Survey Project in Pennsylvania. *NEW SOLUTIONS: A Journal of Environmental and Occupational Health Policy*, *23*(1), 55–83. https://doi.org/10.2190/NS.23.1.e.

Thompson, C. R., Hueber, J., & Helmig, D. (2014). Influence of oil and gas emissions on ambient atmospheric non-methane hydrocarbons in residential areas of Northeastern Colorado. *Elementa: Science of the Anthropocene*, *3*, 000035. https://doi.org/10.12952/journal.elementa.000035.

PSE

Townsend-Small, A., Tyler, S. C., Pataki, D. E., Xu, X., & Christensen, L. E. (2012). Isotopic measurements of atmospheric methane in Los Angeles, California, USA: Influence of "fugitive" fossil fuel emissions: LOS ANGELES METHANE EMISSIONS. *Journal of Geophysical Research: Atmospheres*, *117*(D7), n/a-n/a. https://doi.org/10.1029/2011JD016826.

Trefiak, T. (2006). Pilot study: Optical leak detection and measurement. Conoco Phillips.

UNEP & CCAC (United Nations Environment Programme and Climate and Clean Air Coalition). (2021). *Global Methane Assessment (full report)*. Climate & Clean Air Coalition. https://www.ccacoalition.org/en/resources/global-methane-assessment-full-report.

US EPA (United States Environmental Protection Agency). (2010). *GREENHOUSE GAS EMISSIONS REPORTING FROM THE PETROLEUM AND NATURAL GAS INDUSTRY BACKGROUND TECHNICAL SUPPORT DOCUMENT*. https://www.epa.gov/sites/production/files/2015-05/documents/background-tsd-posted-4-12-10-epa-hq-oar-2009-0923-0027.pdf.

US EPA (United States Environmental Protection Agency). (2020, June 5). *National Emission Standards for Hazardous Air Pollutants (NESHAP)*. Stationary Sources of Air Pollution. https://www.epa.gov/stationary-sources-air-pollution/national-emission-standards-hazardous-air-pollutants-neshap-9.

US EPA EIAG (United States Environmental Protection Agency Emission Inventory and Analysis Group). (2017). Profile of Version 1 of The 2014 National Emissions Inventory. 24.

US EPA (United States Environmental Protection Agency). (2016, September 22). *2014 National Emissions Inventory (NEI) Data* [Other Policies and Guidance]. US EPA. https://www.epa.gov/air-emissions-inventories/2014-national-emissions-inventory-nei-data.

US EPA (United States Environmental Protection Agency). (2021). *Inventory of U.S. Greenhouse Gas Emissions and Sinks, 1990-2019: Executive Summary* (EPA 430-R-21-005). https://www.epa.gov/sites/production/files/2021-04/documents/us-ghg-inventory-2021-chapter-executive-summary.pdf.

Warneke, C., Geiger, F., Edwards, P. M., Dube, W., Pétron, G., Kofler, J., Zahn, A., Brown, S. S., Graus, M., Gilman, J. B., Lerner, B. M., Peischl, J., Ryerson, T. B., de Gouw, J. A., & Roberts, J. M. (2014). Volatile organic compound emissions from the oil and natural gas industry in the Uintah Basin, Utah: Oil and gas well pad emissions compared to ambient air composition. *Atmospheric Chemistry and Physics*, *14*(20), 10977–10988. https://doi.org/10.5194/acp-14-10977-2014.

Wecht, K. J., Jacob, D. J., Frankenberg, C., Jiang, Z., & Blake, D. R. (2014). Mapping of North American methane emissions with high spatial resolution by inversion of SCIAMACHY satellite data: NORTH AMERICA METHANE EMISSION INVERSION. *Journal of Geophysical Research: Atmospheres*, *119*(12), 7741–7756. https://doi.org/10.1002/2014JD021551.

Wennberg, P. O., Mui, W., Wunch, D., Kort, E. A., Blake, D. R., Atlas, E. L., Santoni, G. W., Wofsy, S. C., Diskin, G. S., Jeong, S., & Fischer, M. L. (2012). On the Sources of Methane to the Los Angeles Atmosphere. *Environmental Science & Technology*, *46*(17), 9282–9289. https://doi.org/10.1021/es301138y.

Wigley, T. M. L. (2011). Coal to gas: The influence of methane leakage. *Climatic Change*, *108*(3), 601. https://doi.org/10.1007/s10584-011-0217-3.

Wing, S., Horton, R. A., Marshall, S. W., Thu, K., Tajik, M., Schinasi, L., & Schiffman, S. S. (2008). Air Pollution and Odor in Communities Near Industrial Swine Operations. *Environmental Health Perspectives*, *116*(10), 1362–1368. https://doi.org/10.1289/ehp.11250.

Wunch, D., Wennberg, P. O., Toon, G. C., Keppel-Aleks, G., & Yavin, Y. G. (2009). Emissions of greenhouse gases from a North American megacity. *Geophysical Research Letters*, *36*(15). https://doi.org/10.1029/2009GL039825.

PSE

Zavala-Araiza, D., Sullivan, D. W., & Allen, D. T. (2014). Atmospheric Hydrocarbon Emissions and Concentrations in the Barnett Shale Natural Gas Production Region. *Environmental Science & Technology*, *48*(9), 5314–5321. https://doi.org/10.1021/es405770h.

Zielinska, B., Campbell, D., & Samburova, V. (2014). Impact of emissions from natural gas production facilities on ambient air quality in the Barnett Shale area: A pilot study. *Journal of the Air & Waste Management Association*, *64*(12), 1369–1383. https://doi.org/10.1080/10962247.2014.954735.

PSE

Appendix B. Methodology

List of Tables

PSE

Table B.1. Complete list of Boolean search terms used for upstream, midstream, and downstream methane database search.

Sector	Boolean search terms
Upstream	TS=(("natural gas" OR NG OR methane OR CH$4 OR "Greenhouse Gas*" OR GHG) NEAR/10 (emission* OR leak* OR venting OR loss OR blowdown OR flux* OR dispersion OR migrat* OR accident OR explosion OR blowout OR trace* OR "top-down" OR "bottom-up" OR "source apportionment" OR "super emitter")) AND TS=('oil and gas' OR shale OR petroleum OR 'natural gas' OR 'shale gas' OR 'tight gas' OR 'tight resource' OR 'shale oil' OR 'tight oil' OR 'unconventional gas' OR 'unconventional oil' OR 'unconventional resource' OR 'conventional gas' OR 'conventional oil' OR 'conventional resource' OR 'natural gas liquids' OR drilling OR 'well stimulation' OR 'hydraulic fracturing' OR fracking OR flowback OR 'produced water' OR flar* OR 'coalbed methane' OR inject* OR "supply chain" OR condensate* OR separat* OR well OR "well head" OR wellbore OR "casing head" OR "well pad" OR "abandoned well" OR pump* OR compressor* OR "storage vessel" OR pneumatic OR dehydrator* OR pipeline* OR processing OR fraction*) AND TS=('U.S.' OR 'United States' OR USA OR Canada OR 'North* America' OR Alabama OR Alaska OR Arizona OR Arkansas OR California OR Colorado OR Connecticut OR Delaware OR Florida OR Georgia OR Idaho OR Hawaii OR Illinois OR Indiana OR Iowa OR Kansas OR Kentucky OR Louisiana OR Maine OR Maryland OR Massachusetts OR Michigan OR Minnesota OR Mississippi OR Missouri OR Montana OR Nebraska OR Nevada OR 'New Hampshire' OR 'New Jersey' OR 'New Mexico' OR 'New York' OR 'North Carolina' OR 'North Dakota' OR Ohio OR Oklahoma OR Oregon OR Pennsylvania OR 'Rhode Island' OR 'South Carolina' OR 'South Dakota' OR Tennessee OR Texas OR Utah OR Vermont OR Virginia OR Washington OR 'West Virginia' OR Wisconsin OR Wyoming OR 'Washington DC' OR 'Washington D.C.' OR 'D.C.' OR 'District of Columbia' OR Anadarko OR Ardmore OR Arkoma OR Appalachian OR Devonian OR Bakken OR Barnett OR Chattanooga OR Cherokee OR Delaware OR 'Denver-Julesburg' OR 'Eagle Ford' OR Fayetteville OR 'Fort Worth' OR 'Greater Green River Basin' OR 'Front Range' OR Haynesville OR Inglewood OR Marcellus OR Monterey OR Niobrara OR Permian OR 'Powder River' OR Piceance OR Rogersville OR Saskatchewan OR San Juan OR Uinta OR Utica OR Wattenberg OR Williston OR 'Wind River Basin' OR Woodford OR Wolfcamp OR 'Four Corners' OR 'Canadian Oil Sands') NOT TI=(biogas OR "greenhouse gas" OR cattle OR cow OR "liquified natural gas" OR LNG OR "diesel-methane")
Midstream	TS=(("natural gas" OR NG OR methane OR CH$4 OR "Greenhouse Gas*" OR GHG) NEAR/10 (emission* OR leak* OR flux* OR vent* OR blowdown OR blowout))

AND TS=(pipeline* OR transmission OR "high volume*" OR "trunk line*" OR refiner* OR "storage facilit*" OR "underground storage" OR "gas storage" OR "storage system" OR UGS OR UNGS OR compressor* OR "depleted oil*" OR "salt cavern*" OR "salt dome*" OR dehydration OR "gathering facilit*" OR boosting OR "processing station*" OR "gathering and boosting" OR "gathering and processing" OR "gathering system*" OR "gas gathering" OR "gas processing" OR "gas treatment" OR Separator* OR sweetening OR hinshaw OR midstream OR "regulat* station*" OR "meter* station*")

NOT TI=(wetland* OR landfill* OR rice OR permafrost OR dairy OR dairies OR wastewater OR waste OR forest OR peat* OR LNG OR biogas OR cattle OR cow* OR "liquified natural gas" OR "diesel-methane" OR fertilize*)

NOT TS=(Afghanistan OR Albania OR Algeria OR Andorra OR Angola OR Antigua OR Barbuda OR Argentina OR Armenia OR Australia OR Austria* OR Azerbaijan OR Baden* OR Bahama OR Bahrain OR Bangladesh OR Barbados OR Bavaria* OR Belarus OR Belgium OR Belize OR Benin OR Dahomey OR Bolivia OR Bosnia OR Herzegovina OR Botswana OR Brazil OR Brunei OR Brunswick OR Lüneburg OR Bulgaria OR "Burkina Faso" OR "Upper Volta" OR Burma OR Burundi OR "Cabo Verde" OR Cambodia OR Cameroon OR "Cayman Islands" OR "Central African Republic" OR Chad OR Chile OR China OR Colombia OR Comoros OR Congo OR "Costa Rica" OR Croatia OR Cuba OR Cyprus OR Czechia OR Czechoslovakia OR Denmark OR Djibouti OR Dominica OR "Dominican Republic" OR Ecuador OR Egypt OR "El Salvador" OR Guinea OR Eritrea OR Estonia OR Eswatini OR Ethiopia OR Fiji OR Finland OR France OR Gabon OR Gambia OR Germany OR Ghana OR Greece OR Grenada OR Guatemala OR Guinea OR Guyana OR Haiti OR Hanover* OR Hesse* OR Honduras OR Hungary OR Iceland OR India OR Indonesia OR Iran OR Iraq OR Ireland OR Israel OR Italy OR Jamaica OR Japan OR Jordan OR Kazakhstan OR Kenya OR Yugoslavia* OR Kiribati OR Korea OR Kosovo OR Kuwait OR Kyrgyzstan OR Laos OR Latvia OR Lebanon OR Lesotho OR Liberia OR Libya OR Liechtenstein OR Lithuania OR Luxembourg OR Madagascar OR Malawi OR Malaysia OR Maldives OR Mali OR Malta OR "Marshall Islands" OR Mauritania OR Mauritius OR Micronesia OR Moldova OR Monaco OR Mongolia OR Montenegro OR Morocco OR Mozambique OR Namibia OR Nassau* OR Nauru OR Nepal OR Netherlands OR "New Zealand" OR Nicaragua OR Niger OR Nigeria OR Norway OR |

PSE

Sector	Boolean search terms
	Oldenburg* OR Oman OR Pakistan OR Palau OR Panama OR Paraguay OR Peru OR Philippines OR Piedmont-Sardinia* OR Poland OR Portugal OR Qatar OR Romania OR Russia OR Rwanda OR Samoa OR "Saudi Arabia" OR Schaumburg-Lippe* OR Senegal OR Serbia OR Seychelles OR "Sierra Leone" OR Singapore OR Slovakia OR Slovenia OR "Solomon Islands" OR Somalia OR "South Africa" OR Spain OR "Sri Lanka" OR Sudan OR Suriname OR Sweden OR Switzerland OR Syria OR Tajikistan OR Tanzania OR Thailand OR Timor-Leste OR Togo OR Tonga OR Trinidad OR Tobago OR Tunisia OR Turkey OR Turkmenistan OR Tuvalu OR Uganda OR Ukraine OR Soviet OR "United Arab Emirates" OR "United Kingdom" OR Uruguay OR Uzbekistan OR Vanuatu OR Venezuela OR Vietnam OR Württemberg* OR Yemen OR Zambia OR Zimbabwe OR Europe OR Asia OR Africa OR Oceania OR Mediterranean OR UK OR EU)
Downstream	TS=(("natural gas" OR methane OR CH$4) NEAR/10 (emission* OR leak* OR flux*)) AND TS=(supply-chain OR "distribution infrastructure" OR pipeline* OR city-gate* OR service* OR gas-meter* OR metering OR behind-the-meter OR post-meter OR urban OR home* OR house* OR single-family OR apartment* OR household OR building* OR end-use* OR gas-station* OR water-heater* OR appliance* OR residen* OR furnace* OR cook* OR oven* OR fireplace* OR grill* OR clothes-dryer* OR quiescent OR boiler* OR utility OR power-plant* OR generator* OR turbine* OR electric* OR transformer* OR stack OR city OR cities OR urban OR air-basin* OR "distribution system" OR "distribution sector") NOT TI=(wetland* OR landfill*)

PSE

Table B.2. Complete list of Boolean search terms used for upstream, midstream, and downstream HDAPs database search.

Sector	Boolean search terms
Upstream	TS=(("oil and gas" OR shale OR petroleum OR "natural gas" OR "shale gas" OR "tight gas" OR "tight resource" OR "shale oil" OR "tight oil" OR "unconventional gas" OR "unconventional oil" OR "unconventional resource" OR "conventional gas" OR "conventional oil" OR "conventional resource" OR "natural gas liquids" OR drilling OR "well stimulation" OR "hydraulic fracturing" OR fracking OR flowback OR "produced water" OR flar* OR "coalbed methane" OR inject* OR "supply chain" OR condensate* OR separat* OR well OR "well head" OR wellbore OR "casing head" OR "well pad" OR "abandoned well" OR pump* OR compressor* OR "storage vessel" OR pneumatic OR dehydrator* OR pipeline* OR processing OR fraction*)) AND TS=("U.S." OR "United States" OR Canada OR "North America" OR Alabama OR Alaska OR Arizona OR Arkansas OR California OR Colorado OR Connecticut OR Delaware OR Florida OR Georgia OR Idaho OR Hawaii OR Illinois OR Indiana OR Iowa OR Kansas OR Kentucky OR Louisiana OR Maine OR Maryland OR Massachusetts OR Michigan OR Minnesota OR Mississippi OR Missouri OR Montana OR Nebraska OR Nevada OR "New Hampshire" OR "New Jersey" OR "New Mexico" OR "New York" OR "North Carolina" OR "North Dakota" OR Ohio OR Oklahoma OR Oregon OR Pennsylvania OR "Rhode Island" OR "South Carolina" OR "South Dakota" OR Tennessee OR Texas OR Utah OR Vermont OR Virginia OR Washington OR "West Virginia" OR Wisconsin OR Wyoming OR "Washington DC" OR "Washington D.C." OR "D.C." OR "District of Columbia" OR Anadarko OR Ardmore OR Arkoma OR Appalachian OR Devonian OR Bakken OR Barnett OR Chattanooga OR Cherokee OR Delaware OR "Denver-Julesburg" OR "Eagle Ford" OR Fayetteville OR "Fort Worth" OR "Greater Green River Basin" OR "Front Range" OR Haynesville OR Inglewood OR Marcellus OR Monterey OR Niobrara OR Permian OR "Powder River" OR Piceance OR Rogersville OR Saskatchewan OR San Juan OR Uinta OR Utica OR Wattenberg OR Williston OR "Wind River Basin" OR Woodford OR Wolfcamp OR "Four Corners" OR "Canadian Oil Sands") AND TS=("air concentration" OR "air monitoring" OR "air pollution" OR "air pollution emission" OR "air quality" OR "air quality impact" OR "atmospheric emission" OR benzene OR "black carbon" OR BTEX OR "chemical mixture" OR "hydrogen sulfide" OR H2S OR "hazardous air pollutant*" OR HAP OR HAPS OR "hydrocarbon emission" OR "nitrogen dioxide" OR NO2 OR "non-methane hydrocarbon" OR "non-methane volatile organic compounds" OR NMVOC* OR ozone OR O3 OR "particulate matter" OR PM2.5 OR PM10 OR PM OR "sulfur dioxide" OR SO2 OR "toxic emission" OR "toxic hydrocarbon emission" OR "volatile compound" OR "volatile organic compound" OR VOC* OR H2S OR "hydrogen sulfide") AND TS= (inhal* OR expos* OR biomarker OR biomonitor OR endocrine OR carcin* OR tox* OR impact OR hazard OR risk OR health OR safety OR anxiety OR stress OR depression OR symptom OR epidemiology OR trauma OR risk* OR human) NOT TI= (wetland* OR landfill* OR rice OR permafrost OR dairy OR wastewater OR waste OR forest OR peat* OR LNG OR agriculture)
Midstream	TS=(("oil and gas" OR shale OR petroleum OR "natural gas" OR "shale gas" OR "tight gas" OR "tight resource" OR "shale oil" OR "tight oil" OR "unconventional gas" OR "unconventional oil" OR "unconventional resource" OR "conventional gas" OR "conventional oil" OR "conventional resource" OR "natural gas liquids" OR drilling OR "well stimulation" OR "hydraulic fracturing" OR fracking OR flowback OR "produced water" OR flar* OR inject*) NEAR/10 (pipeline* OR transmission OR "high volume*" OR "trunk line*" OR refiner* OR "storage facilit*" OR "underground storage" OR "gas storage" OR "storage system" OR UGS OR UNGS OR compressor* OR "depleted oil*" OR "salt cavern*" OR "salt dome*" OR dehydration OR "gathering facilit*" OR boosting OR "processing station*" OR "gathering and boosting" OR "gathering and processing" OR "gathering system*" OR "gas gathering" OR "gas processing" OR "gas treatment" OR separator* OR sweetening OR hinshaw OR midstream OR "regulat* station*" OR "meter* station*")) AND TS=("air concentration" OR "air monitoring" OR "air pollution" OR "air pollution emission" OR "air quality" OR "air quality impact" OR "atmospheric emission" OR benzene OR "black carbon" OR BTEX OR "chemical mixture" OR "hydrogen sulfide" OR H2S OR "hazardous air pollutant*" OR HAP OR HAPS OR "hydrocarbon emission" OR "nitrogen dioxide" OR NO2 OR "non-methane hydrocarbon" OR "non-methane volatile organic compounds" OR NMVOC* OR ozone OR O3 OR "particulate matter" OR PM2.5 OR PM10 OR PM OR "sulfur dioxide" OR SO2 OR "toxic emission" OR "toxic hydrocarbon emission" OR "volatile compound" OR "volatile organic compound" OR VOC* OR H2S OR "hydrogen sulfide" OR inhal* OR expos* OR biomarker OR biomonitor OR endocrine OR carcin* OR tox* OR impact OR hazard OR risk OR health OR safety OR anxiety OR stress OR depression OR symptom OR epidemiology OR trauma OR risks OR human) NOT TI=(wetland* OR landfill* OR rice OR permafrost OR dairy OR dairies OR wastewater OR waste OR forest OR peat* OR LNG OR biogas OR cattle OR cow* OR "liquified natural gas" OR "diesel-methane" OR fertilize* OR Afghanistan OR Albania OR Algeria OR Andorra OR Angola OR Antigua OR Barbuda OR Argentina OR Armenia OR Australia OR Austria* OR Azerbaijan OR Baden* OR Bahama OR Bahrain OR Bangladesh OR Barbados OR Bavaria* OR Belarus OR Belgium OR Belize OR Benin OR Dahomey OR Bolivia OR Bosnia OR Herzegovina OR Botswana

PSE

	OR Brazil OR Brunei OR Brunswick OR Lüneburg OR Bulgaria OR "Burkina Faso" OR "Upper Volta" OR Burma OR Burundi OR "Cabo Verde" OR Cambodia OR Cameroon OR "Cayman Islands" OR "Central African Republic" OR Chad OR Chile OR China OR Colombia OR Comoros OR Congo OR "Costa Rica" OR Croatia OR Cuba OR Cyprus OR Czechia OR Czechoslovakia OR Denmark OR Djibouti OR Dominica OR "Dominican Republic" OR Ecuador OR Egypt OR "El Salvador" OR Guinea OR Eritrea OR Estonia OR Eswatini OR Ethiopia OR Fiji OR Finland OR France OR Gabon OR Gambia OR Germany OR Ghana OR Greece OR Grenada OR Guatemala OR Guinea OR Guyana OR Haiti OR Hanover* OR Hesse* OR Honduras OR Hungary OR Iceland OR India OR Indonesia OR Iran OR Iraq OR Ireland OR Israel OR Italy OR Jamaica OR Japan OR Jordan OR Kazakhstan OR Kenya OR Yugoslavia* OR Kiribati OR Korea OR Kosovo OR Kuwait OR Kyrgyzstan OR Laos OR Latvia OR Lebanon OR Lesotho OR Liberia OR Libya OR Liechtenstein OR Lithuania OR Luxembourg OR Madagascar OR Malawi OR Malaysia OR Maldives OR Mali OR Malta OR "Marshall Islands" OR Mauritania OR Mauritius OR Micronesia OR Moldova OR Monaco OR Mongolia OR Montenegro OR Morocco OR Mozambique OR Namibia OR Nassau* OR Nauru OR Nepal OR Netherlands OR "New Zealand" OR Nicaragua OR Niger OR Nigeria OR Norway OR Oldenburg* OR Oman OR Pakistan OR Palau OR Panama OR Paraguay OR Peru OR Philippines OR Piedmont-Sardinia* OR Poland OR Portugal OR Qatar OR Romania OR Russia OR Rwanda OR Samoa OR "Saudi Arabia" OR Schaumburg-Lippe* OR Senegal OR Serbia OR Seychelles OR "Sierra Leone" OR Singapore OR Slovakia OR Slovenia OR "Solomon Islands" OR Somalia OR "South Africa" OR Spain OR "Sri Lanka" OR Sudan OR Suriname OR Sweden OR Switzerland OR Syria OR Tajikistan OR Tanzania OR Thailand OR Timor-Leste OR Togo OR Tonga OR Trinidad OR Tobago OR Tunisia OR Turkey OR Turkmenistan OR Tuvalu OR Uganda OR Ukraine OR Soviet OR "United Arab Emirates" OR "United Kingdom" OR Uruguay OR Uzbekistan OR Vanuatu OR Venezuela OR Vietnam OR Württemberg* OR Yemen OR Zambia OR Zimbabwe OR Europe OR Asia OR Africa OR Oceania OR Mediterranean OR UK OR EU)
Downstream	TS=(("natural gas" OR methane OR CH$4) NEAR/10 (emission* OR leak* OR flux*)) AND TS=('air concentration' OR 'air monitoring' OR 'air pollution' OR 'air pollution emission' OR 'air quality' OR 'air quality impact' OR 'atmospheric emission' OR 'benzene' OR 'black carbon' OR BTEX OR 'chemical mixture' OR 'hydrogen sulfide' OR H2S OR 'hazardous air pollutant*' OR HAP OR HAPS OR 'hydrocarbon emission' OR 'nitrogen dioxide' OR NO2 OR 'non-methane hydrocarbon' OR 'non-methane volatile organic compounds' OR NMVOC* OR ozone OR O3 OR 'particulate matter' OR PM2.5 OR PM10 OR PM OR 'sulfur dioxide' OR SO2 OR 'toxic emission' OR 'toxic hydrocarbon emission' OR 'volatile compound' OR 'volatile organic compound' OR VOC* OR H2S OR 'hydrogen sulfide') AND TS= (inhal* OR expos* OR biomarker OR biomonitor OR endocrine OR carcin* OR tox* OR impact OR hazard OR risk OR health OR safety OR anxiety OR stress OR depression OR symptom OR epidemiology OR trauma OR risk OR "distribution infrastructure" OR pipeline OR city$gate OR service* OR "gas$meter" OR "metering" OR "behind the meter" OR "post meter" OR urban OR home OR "single$family" OR apartment OR household OR building OR "end$use*" OR "gas station" OR "water$heater*" OR appliance* OR residen* OR furnace OR cook* OR oven OR fireplace OR grill OR "clothes$dryer" OR quiescent OR boiler* OR utility OR "service line" OR stove*) NOT TI= (wetland* OR landfill* OR rice OR permafrost OR dairy OR wastewater OR waste OR forest OR peat* OR LNG OR agriculture)

Table B.3. PRISMA framework for systematic review of publications relevant to methane emissions.

Flag - for Additional Review?	Questions for others to review/clarify Fill in background color Red to flag
Is this Study Classified Correctly?	Some studies may be misclassified and belong in the HDAPs PRISM or they may not contain useful information and should not be reviewed. **Add additional context in column A if unsure.
ROGER or WOS Search?	Did the article come from the Web of Science word search or from the ROGER database?
Full MLA Citation & link to PDF	e.g., Shonkoff et al., 2019a
Year	Year published
Paper Title	Full title
Annotated PDF	If you prefer to highlight/add comments to PDF, you can re-upload marked-up PDF here: Link here **Please name with author year and your initials (e.g., Shonkoff_2019a_DM)
Reviewer	Your name
Article Type	Research Article Critical Review Commentary Policy Analysis Technology Paper Perspective Government Document Book Chapter Conference Proceeding Other (write-in)
Is this study duplicated in the HDAPs PRISM (Y/N)	Follow link to HDAP PRISM **If yes - please enter your name in HDAP review sheet ASAP
High-level Contribution/Implications of Study	Describe the high-level contribution/implication of study: e.g., (improve the overall methane emissions budget of California state in supporting new climate policies) All verbatim language should be enclosed in "double quote" and Inferred information or notes should be in italics

PSE

	**note it is okay is there is redundant information in this column with other columns - please include
(Subjective) Rank Study in terms of importance to lit. review	Low = very little contribution (e.g., no new data) Medium = moderate contribution (new data but very limited in scale) High = Strong contribution (systematic review - cross-sectoral - high impact journal) **Writing will begin with high contribution studies
Candid thoughts? **Take home message?**	In your own words, is this study useful, complicated? e.g., Contains new data, but not useful for our review
Identified Research Gaps	Typically located near end of manuscript - AKA the "more research needed" section
Health-damaging/ co-pollutants included? **(Write-in)**	N/A (-------) if only methane measured, include the other pollutants if included. Abbreviations okay. e.g., (BTEX, H2S, NO2, Alkanes, Other hydrocarbons) **If uncertain, please include
Implication/Opportunity to co-reduce HDAPs? **(Write-in)**	If so, provide additional detail: 1) Does the system in question deal with raw unprocessed gas that may contain HDAPs? 2) If methane emissions addressed, would HDAPs also by default?
Hydrocarbon Type	- Natural Gas - Oil - Oil and Natural Gas - Oil & Associated Gas - Natural Gas Liquids - Processed Natural Gas - Compressed Natural Gas - Other
Geologic Extraction Source Type	- Conventional Gas/Oil - Shale/Tight Gas/Oil - Coalbed Methane - Migrated Hydrocarbons - Unknown/Unidentified - N/A (Midstream & Downstream)

PSE

Upstream **(Production & Processing)**	(Y/N) If reviewed study reported methane emissions from upstream sources. **Note - some sources are present across multiple segments - refer to "Source Glossary" Sheet to make determination
Upstream Subsector	Any specific source, operation, component, etc. e.g., Production & Processing > Separators **refer to "Source Glossary" Sheet to make determination
Midstream **(Transmission & Storage)**	(Y/N) If midstream sources.
Midstream Subsector and units	Any specific source, operation, component, etc. e.g., Transmission & Storage > Gathering & Boosting
Downstream **(Distribution)**	(Y/N) If reviewed study reported methane emissions from downstream sources.
Downstream Subsector	Any specific source, operation, component, etc. e.g., Distribution > metering station
Whole Supply Chain	(Y/N) If reported methane emissions from the whole supply chain sources. **note Yes if study provides estimates or LCA of total emissions - at least two sectors should be selected as "Y" in the columns to the left
Primary Data Collection **(Y/N)**	New data collected that did not exist prior to this study?
Data Collection / Analysis Time Span	Be as specific as possible: e.g., MM/DD/YYYY - MM/DD/YYYY
The state of operating conditions	- Steady-state on - Steady-state off - Accidental release - Combination of types - Unknown/not reported

PSE

Generic - Geographic Scale of Measurement/ Sampling Campaign	- Global - USA - Canada - Mexico - Basin - Field - Sub-field (e.g., well-pad) - State - City - Regional (confined unit) - Multi-regional (disparate areas)
Specific - Geography of Measurement **(Include state(s))	E.g., (Marcellus - PA, WV) e.g., (San Mateo; Santa Clara counties - CA) **please include state(s) that were included in the campaign (add additional info when possible)
Approach / Methodology	Include high-level detail on study methods. e.g., (Direct measurements collected using optical gas imaging cameras at 23 compressor stations)
Secondary Source Description	Describe any secondary sources used in the study e.g., (TCEQ Database)
Type of Emissions & Units	1. Concentration (ug/m3, ppb, etc.) 2. Flux 3. Both conc. & flux 4. Scaled Emissions Factors 5. Other (explain) e.g., flux (Gg/year) **Note - multiple types may be included
Measurements Scaled up to Estimate Total Emissions or Emissions Factors?	(Y/N) **Emissions may be scaled up to geographic area or by a component or sub-sector/sector
Scaling Methodology **Include units	Describe if emissions were scaled by geography, sector-wide component-level, other - include units) **If included, also note the scaling method - either top-down, bottom-up, or combo

PSE

	e.g., (well casing / bottom-up)
Geography Scale of scaled emissions	If Basin-level or less, please describe further (e.g., Marcellus, Boston)
Emissions Mitigation Recommendations	These recommendations will likely be verbatium from the study - make sure to put all verbatium language in "double quotes" and inferred infomation in italics.
Funder Type	Private Federal State Other
Funder Description	Copy and paste the "Acknowledgements" section (or equivalent) - make sure to put all verbatium language in "double quotes" and inferred infomation in italics.
Policy Recommendations	**note it is okay if there is redundant information in this column with other columns - please include
Explicit Research Limitations	All verbatium language should be enclosed in "double quote" Inferred information or notes should be in italics **note it is okay is there is redundant information in this column with other columns - please include
Notes and Comments & Screenshots	Any additional information the reviewer found interesting or noteworthy. Also if any confusion may have arisen here in regards to sector emissions or multiple sector emissions - that can be clarified here. Also if any tables or figures were saved as screenshots - they can be uploaded here, and shortcut link can be shared below. Please save the screenshot as Author, year (e.g., Lebel et al, 2020)

PSE

Table B.4. PRISMA framework for systematic review of publications relevant to HDAP emissions.

Flag - for Additional Review?	Questions for others to review/clarify Fill in background color Red to flag
Is this Study Classified Correctly?	Some studies may misclassified and belong in the HDAPs PRISM or they may not contain useful information and should not be reviewed. **Add additional context in column A if unsure.
ROGER or WOS Search?	Did the article come from the Web of Science word search or from the ROGER database?
Full MLA Citation & link to PDF	e.g., Shonkoff et al., 2019a
Year	Year published
Paper Title	Full title
Annotated PDF	If you prefer to highlight/add comments to PDF, you can re-upload marked-up. **Please name with author year and your initials (e.g., Shonkoff_2019a_DM)
Reviewer	First name
Article Type	Research Article Critical Review Commentary Policy Analysis Technology Paper Perspective Government Document Book Chapter Conference Proceeding Other (write-in)
Is this study duplicated in the Methane PRISM? **(Y/N)**	Follow link to HDAP PRISM **If yes - please enter your name in HDAP review sheet ASAP
High-level Contribution/Implications of Study	Describe the high-level contribution/implication of study: e.g., (improve the overall methane emissions budget of California state in supporting new climate policies) All verbatim language should be enclosed in "double quote" and Inferred information or notes should be in italics **note it is okay is there is redundant information in this column with other columns - please include

PSE

(Subjective) Rank Study in terms of importance to lit. review ****Do not leave blank****	Low = very little contribution (e.g., no new data) Medium = moderate contribution (new data but very limited in scale) High = Strong contribution (systematic review - cross-sectoral - high impact journal) **Writing will begin with high contribution studies
Candid thoughts?	In your own words, is this study useful, complicated? e.g., Contains new data, but not useful for our review; FLAG if the study is not helpful and indicate which sections you evaluated (e.g., Abstract, results)
Identified Research Gaps (Write-in) ****Do not leave blank****	Typically located near end of manuscript - AKA the "more research needed" section
Quantitative Methane Data Reported?	Yes/No
Type of Emissions & Units	1. Concentration 2. Flux 3. Both conc. & flux 4. Scaled Emissions Factors 5. Other (explain) e.g., flux (Gg/year) **Note - multiple types may be included
Upstream (Production & Processing)	(Y/N) If reviewed study reported HDAP emissions from upstream sources. **Note - some sources are present across multiple segments - refer to "Source Glossary" Sheet to make determination
Upstream Subsector	Any specific source, operation, component, etc. e.g., Production & Processing > Separators **refer to "Source Glossary" Sheet to make determination
Midstream (Transmission & Storage)	(Y/N) If midstream sources.
Midstream Subsector and units	Any specific source, operation, component, etc. e.g., Transmission & Storage > Gathering & Boosting

PSE

Downstream **(Distribution)**	(Y/N) If reviewed study reported HDAP emissions from downstream sources.
Downstream Subsector	Any specific source, operation, component, etc. e.g., Distribution > metering station
Whole Supply Chain	(Y/N) If reported HDAP emissions from the whole supply chain sources. **note Yes if study provides estimates or LCA of total emissions - at least two sectors should be selected as "Y" in the columns to the left
Health-damaging co-pollutants included? Abbreviations okay	N/A (-------) if only methane measured, include the other pollutants if included. Abbreviations okay. e.g., (BTEX, H2S, NO2) **Please include if uncertain if health-damaging or create a flag for additional review
Primary Data Collection **(Y/N)**	New data collected that did not exist prior to this study?
Data Collection / Analysis Time Span	Be as specific as possible: e.g., MM/DD/YYYY - MM/DD/YYYY
Generic - Full geographic Scale of Measurement/ Sampling Campaign	- Global - USA - Canada - Mexico - Basin - Field - Sub-field (e.g., well-pad) - State - City - Regional (confined unit) - Multi-regional (disparate areas)
Specific - Geography of Measurement **(Include state(s))	E.g., (Marcellus - PA, WV) e.g., (San Mateo; Santa Clara counties - CA) **please ALWAYS include state(s) that were included in the campaign (add additional info when possible)
Secondary Source Description	Describe any secondary sources used in the study e.g., (TCEQ Database)

PSE

Sampling Methods - Sample Size/Collection Method	Pertinent information: - Type: e.g., 24-hr integrated grab samples - N = # grab samples - Mobile ground-based/aerial-based survey - Statistical/Deterministic modeling (e.g., Gaussian) - Hybrid methods (describe) **if multiple pollutants reported, please distiguish
Hydrocarbon Type	- Natural Gas - Oil - Oil and Natural Gas - Oil & Associated Gas - Natural Gas Liquids - Processed Natural Gas - Compressed Natural Gas - Other
Geologic Extraction Source Type	- Conventional Gas/Oil - Shale/Tight Gas/Oil - Coalbed Methane - Migrated Hydrocarbons - Unknown/Unidentified - N/A (Midstream & Downstream)
Exposure/Health Analysis Conducted? (Y/N)	**Can include air concentraiton comparisons to air quality standards. (e.g., modeled PM2.5 exceeds the 24-hr NAAQS standards 5% of year) **Study may estimate health impacts without quantitative HDAP data (e.g., proximity)
If Health/Expsosure, what Approach / Methodology?	Examples may include: - Indicator metric used (e.g., proximity) - Concentration compared to stds. - Exposure assessment - Air and human health risk assessment - Undefined **Continuum generally follows: Risk > Exposure > Concentration > Proximity
Evaluated Health Impacts	If so, provide additional detail: 1) What health outcomes measured (e.g., excess cancer risk, hospitalizations, pre-term birth) 2) Magnitude and direction of impacts 3) Describe population characteristics, if available
Emissions/impacts Mitigation Recommendations	These recommendations will likely be verbatium from the study - make sure to put all verbatium language in "double quotes" and inferred infomation in italics.

Funder Type	Private Federal State Other
Funder Description	Copy and paste the "Acknowledgements" section (or equivalent) - make sure to put all verbatium language in "double quotes" and inferred infomation in italics.
Policy Recommendations (if any)	**note it is okay is there is redundant information in this column with other columns - please include
Explicit Research Limitations	All verbatium language should be enclosed in "double quote" Inferred information or notes should be in italics **note it is okay is there is redundant information in this column with other columns - please include
Notes and Comments & Screenshots	Any additional information the reviewer found interesting or noteworthy. Also if any confusion may have arisen here in regards to sector emissions or multiple sector emissions - that can be clarified here. Also if any tables or figures were saved as screenshots - they can be uploaded here, and shortcut link can be shared below. Please save the screenshot as Author, year. fig#, initials (e.g., Lebel_2020_Fig1_EL)

PSE