Evaluating economic opportunities and challenges for energy recovery from methane leaks during wastewater treatment

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Abstract

Methane leaks from wastewater treatment represent the loss of biogas that can be used to generate onsite energy, offsetting costs and improving efficiency. Here, we characterize emissions from water resource recovery facilities by compiling measurement data and calculating biogas-production normalized leak rates for facilities with anaerobic digestion. For plants where biogas data were unavailable, we developed an empirical method to estimate production using annual data from 43 facilities. However, we find notable differences in production-normalized leak rates from measurement data where biogas data was available (mean: 12% [95% CI: 8-17%], median: 8%) and those where production was empirically derived (mean: 34% [95% CI: 28-41%], median: 23%). Considering different techno-economic scenarios for leak rates and gas capturability, we find the largest 5% of facilities in the United States could recover over $100,000/year/facility in currently forgone revenue by capturing gas by capturing gas leaked at rates as low as 3%; at rates ≥25%, accrued value could reach several million dollars. We conducted a Monte Carlo simulation to determine the financial cost of methane leaks considering existing energy recovery facilities in the United States, with different scenarios for the underlying leak distribution, and find median annual loss could range from $13 million to $42 million nationwide.

Introduction

Methane, an energy-rich molecule and the primary constituent of biogas, is produced biologically through the anaerobic microbial degradation of organic material.[1] For water resource recovery facilities (WRRFs), methane generated through wastewater treatment serves as a revenue stream when used onsite for heat and energy, or when upgraded to natural gas quality and sold for external use.[2] Yet with current emissions potentially larger than government estimates by as much as two- to threefold, methane leaks could represents a substantial loss of revenue for WRRFs.[3,4] A recent study estimated that nationally in the United States, WRRFs emit 0.5 – 0.9 million metric tons (MMT) CH_4/year,[4] equivalent to 20 – 35% of natural gas production in the state of California in 2023 (2.3 MMT CH_4).[5]

The economic impact of leak detection and repair (LDAR) depends on the size of the methane source, from where it originates within a facility, and the extent to which it can be captured. Leaks are primarily associated with anaerobic digestion,[3] and thus reducing emissions will likely directly translating to increased biogas production and utilization. However, unintentional emissions can also occur in upstream and downstream wastewater treatment processes, e.g. from anaerobic conditions in aeration basins or sewers,[6] where they are less readily capturable. Additionally, current estimates of total methane emissions are highly sensitive to emission factors based on a limited number of measurement studies.[7] Where actual emissions lie within current uncertainty bounds will impact the economics of strategies individual facilities use to find and repair sources of fugitive methane. While several recent studies report emissions factors for WRRFs, given the wide range of methane measurement techniques, study designs implemented, and the inherent variability across facilities, it remains unclear what approach individual facilities should use when evaluating the economics of mitigating leak rates.

A small number of studies examine the economics of methane leaks from WRRFs. A Danish survey of methane leaks at 11 WRRFs that used biogas for electricity generation found a positive net present value on mitigation efforts for 8 of these facilities on a 20-year time horizon.[8] Another European study found more favorable economics, using a Monte Carlo simulation to estimate a median LDAR payback period of 6.3 years for biogas plants generating electricity and heat, and 1.0 year for those upgrading biogas for injection into the natural gas grid.[9] However, these studies broadly addressed biogas plants, including but not focused on WRRFs. Additionally, technical costs and incentives in Europe will not necessarily translate to facilities located in the United States, where economic analysis is, to the best of our knowledge, limited to LDAR for the oil and gas sector.[10,11] Finally, data availability on representative leak rates, biogas production, and costs are often not published in the scientific literature difficult to obtain, posing a further challenge to conducting economic analyses.

This work fills several gaps in the current literature by characterizing production normalized emissions from WRRFs in the United States and evaluating the economic opportunities from capturing AD methane for use onsite. Additionally, due to the limited availability of biogas data, we developed an empirical approach for estimating biogas production based on 1-year data from 47 facilities in the United States, a data-based alternative to typical engineering rules of thumb often used to estimate WRRF electricity generation (see Hodson et al., 2026 for examples[12]). Informed by our analysis of production-normalized emissions, we modelled the revenue streams available to WRRFs if fugitive methane were used for onsite heat and power, considering ranges in leak rate, gas capturability, and facility size. Finally, we used a Monte Carlo simulation to estimate national revenue potential from methane leaks at WRRFs with energy recovery capabilities in the United States.

Datasets and Methods

Estimating biogas production based on reported flow rates

Methane emission factors are typically calculated using methane leak rates, which are in turn normalized by either the facility's treated wastewater flowrate or biogas production. However, biogas production rate at facilities is often not reported in the current literature (discussed further below). Thus, we developed an empirical method for estimating biogas production based on facility flow rate using raw data described in Chini and Stillwell (2018)[13], provided to the authors upon request. This dataset includes 1-year, facility-level flow and biogas production data from 2012 for 47 facilities, provided in response to Freedom of Information Act requests. We used data from 42 facilities in our analysis, removing 5 facilities in quality control (four due to implausible biogas production given facility size; one due to reported flow rates lower than the known flow rate of the facility). To obtain consistent units of biogas production as kg CH_4/hour, we assumed 55 MJ/kg CH_4 (100 scf CH_4/therm),[14] and that biogas is 65% (v/v) CH_4.[15] We determined the equation of best fit (**Equation 1**) between biogas production and flow rate using a linear regression with a fixed y-intercept at the origin (see Supplementary Information for statistical details).

$$Biogas\ generation \left[\frac{kg\ CH_4}{hour}\right] = 0.00148 * Flow\ \left[\frac{m^3}{day}\right] \qquad \textbf{Equation 1}$$

Data on methane emissions from water resource recovery facilities

We synthesized methane leak data reported previously in one literature review that compiled 136 measurements from 90 WRRF sites[16] and four subsequently published original measurement studies[8,17–19], resulting in a total of 181 datapoints. The literature-based study compiled emission factor data through automated literature mining and subsequently manual extraction of methane leak, flow rates, and treatment process information for each plant. Where presence or absence of anaerobic digestion was not specified, we checked the original source literature. The measurement studies monitored CH_4 at WRRFs using different methods for estimating methane concentration and emissions rate. Moore et al (2023 and 2025) measured methane mole fraction on a vehicle-mounted sensor and estimated emissions rate using a plume-integrated inverse Gaussian plume model with Bayesian source rate inference.[17,18] Fredenslund et al., 2023 used the tracer gas dispersion method to estimate whole plant methane emissions[8], and Gålfalk and Bastviken, 2025 implemented a mass-balance method using data collected from vertical wall drone flights performed perpendicular to the prevailing wind direction.[19] Key parameters of data sources are summarized in **Supplementary Table S2**.

All measurement studies reported methane leak rates, and presence or absence of anaerobic digestion onsite. Note that here we use the term "leaks" broadly, as reported methane leaks may also include intentional venting as part of routine operation. Song et al. 2023 and Moore et al. (2023 and 2025) reported methane leak rates on a mass flow basis (e.g. kg CH_4/hour or similar) alongside volumetric flow rate of treated wastewater for each facility. However, they did not provide biogas production rates during the measurement period.[16–18] Fredenslund et al, 2023 reported mass-flow methane leak rates and biogas production rates, but did not report facility flow rate.[8] Gålfalk and Bastviken (2025) reported methane leak rate, and provided annual biogas production rate for each facility upon request.[19]

Developing dataset of production-normalized emissions rate

Production normalized emission (%) is methane leak rate (kgCH_4/hour) divided by biogas production as kgCH_4/hour, assuming biogas is 65% (v/v) methane.[15] Note that methane leak rates could include natural gas emissions for process and building heating. Emissions from natural gas may artificially increase the production-normalized emissions rate as these values are based on only biogas production. Of the 181 measurements in our dataset, 34 included an associated biogas production rate. For data where biogas production was not reported in the source study (n=147, over 80% of leak measurement), we estimated biogas production rate using **Equation 1**.

Economics of methane leak detection and repair at WRRFs

We calculated the potential annual energy offset of methane leaks if gas were captured and used to meet onsite heat and power needs. To convert volume of methane into electricity production, we assumed 55.6 MJ per kg CH_4 (higher heating value, HHV)[20], and a lean burning reciprocating engine with an electrical efficiency of 32.6% (based on HHV) and a power-to-heat ratio of 0.86.[21] We set energy prices to $0.09/kWh for electricity and $0.008/MJ natural gas, based on the median prices for facilities with CHP in the United States, using on the 2023 industrial rate reported by for the states in which these facilities are located.[22,23] All monetary values use a currency year of 2023 for U.S. dollars.

To apply this analysis to actual facilities in the United States, we used previously reported location and flow rate data on 321 facilities with biogas energy recovery.[7] We estimated the total national financial revenue loss from methane leaks at these facilities using a Monte Carlo simulation that varied key input parameters to the calculations described above. For facility leak rate, we considered three different scenarios: 1) bootstrapping leak rate from the entire production-normalized emissions dataset, including measurements where biogas production was interpolated from flow rate 2) bootstrapping leak rate from a data subset where biogas production was available in the original study (i.e. excluding measurements where biogas was interpolated with **Equation 1**) 3) assuming a log-normal (heavy-tail) distribution with a median leak rate of 5% to represent a conservative, low-leak scenario compared to existing measurement data. Additionally, for fraction of leaked gas that is capturable, we assumed a uniform distribution between 0.5 and 0.9. For conversion to electricity, we used the same engine efficiency properties described above. For monetary energy values, we assumed a normal distribution around the average electricity and natural gas price from 2023 for industrial users within a given facility's state.[22] For this dataset, electricity price ranged from $0.06/kWh to $0.19/kWh (mean: $0.11/kWh, median: $0.09/kWh) and natural gas price ranged from $0.002/MJ to $0.013/MJ (mean: $0.009/MJ, median: $0.008/MJ).

Results

Comparison of measurement-based methane leak rates from WRRFs

Methane emission factors are typically calculated using methane leak rates normalized by either facility treated wastewater flowrate or biogas production. To allow us to estimate biogas production where metered data is unavailable, we developed an empirical method for estimating biogas production based on facility flow rate using data from a 1-year period across 42 facilities (**Figure 1a**). We used a linear regression with a fixed y-intercept at the origin (**Equation 1**) and calculated both 95% confidence intervals and 95% predictive intervals. Full statistical results of the linear regression are included in **Supplementary Tables S1** and **S2**.

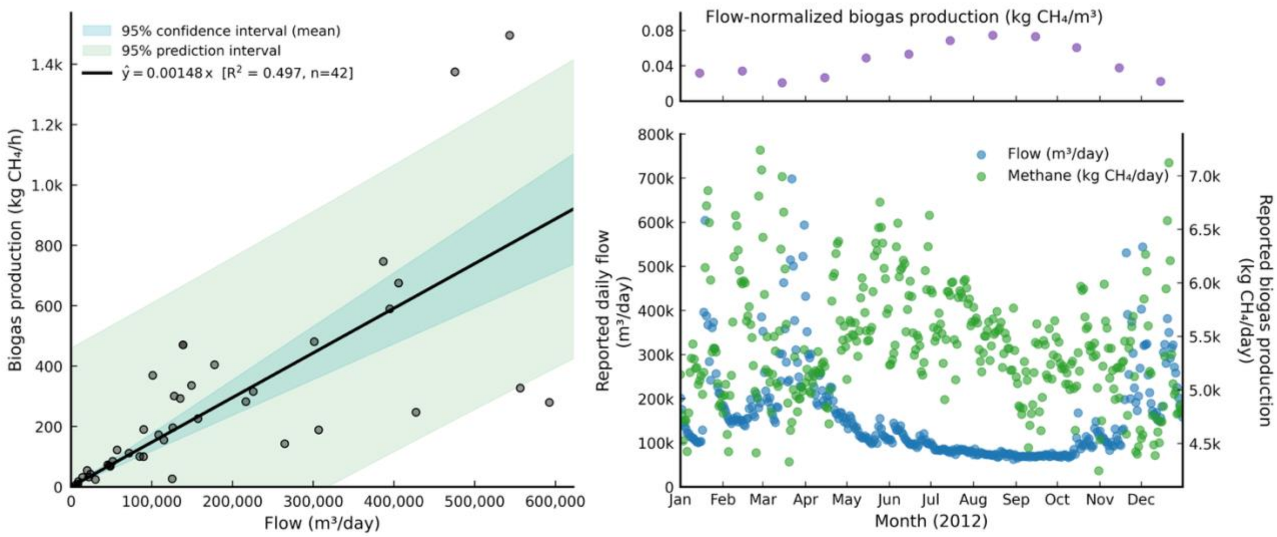

Figure 1: Relationship between biogas production and facility flow rate. A. Linear regression on biogas production (kg CH4/hr) and flow (m3/day) from 1-year measurement data at 43 facilities. B. Measurement data from a facility in Eugene, Oregon (bottom) and calculated flow normalized biogas production (top). Underlying data are those described in Chini and Stillwell (2018), provided to the authors upon request.[13]

Our method predicts mean biogas production rate of 0.0355 kg CH_4 or 1.8 MJ CH_4 per m^3 of treated wastewater, aligning with previously published process models where mean production across different treatment configurations is 1.7 MJ biogas per m^3 of wastewater treated.[24] However, data show a high degree of scatter ($R^2 = 0.5$ and $R_0^2 = 0.7$, see Supplementary Methods for statistical details) and wide 95% predictive intervals. The high degree of scatter reflects the fact that biogas production rates can vary substantially based on facility design and operation. For example, co-digesting wastewater solids with food waste and fats, oils and grease (FOG) can double biogas production at a given facility.[25] Collecting additional data on the composition of influent wastewater and solids streams could inform future work to develop an expanded version of this regression. Additionally, we observed a high degree of variability in both flow and biogas production within the data, as highlighted in **Figure 1b**, which depicts the daily measurement data at one of the facilities included in **Figure 1a**. We aggregated oftentimes daily reported measurements to an annual scale, which could contribute to the uncertainty of our model given the underlying variability. Future work could examine shorter timescales to improve our ability to predict methane production.

Next, we evaluated the relationship between measured methane leaks and facility size, in terms of flow of wastewater treated (m^3/day) and biogas production (kg CH_4/hr) (**Figure 2**). We found measured leak rate scales with flow according to a power law (linear on a log-log scale, **Figure 2a**). We fit power-law equations across the full dataset, and separately for facilities with and without anaerobic digestion. Facilities with anaerobic digestion have higher median flow-normalized emissions than those without (0.0082 vs 0.0037 kg CH_4/m^3), although mean values (with AD: 0.0121 [95% CI: 0.0099–0.0143] kg CH_4/m^3, without AD: 0.0134 [95% CI: 0.0097–0.0172] kg CH_4/m^3) are not significantly different according to Welch's t-test (p=0.55), reflecting the skewed distribution of the data. Within this dataset, facilities with AD have an average flow of 0.15 Mm^3/day (40 million gallons per day, MGD), larger than those without AD which have a mean flow rate of 0.067 Mm^3/day (18 MGD) (p=0.0022 with Welch's t-test). Additional research is needed to further characterize methane emissions a function of facility size, and to identify key underlying drivers.

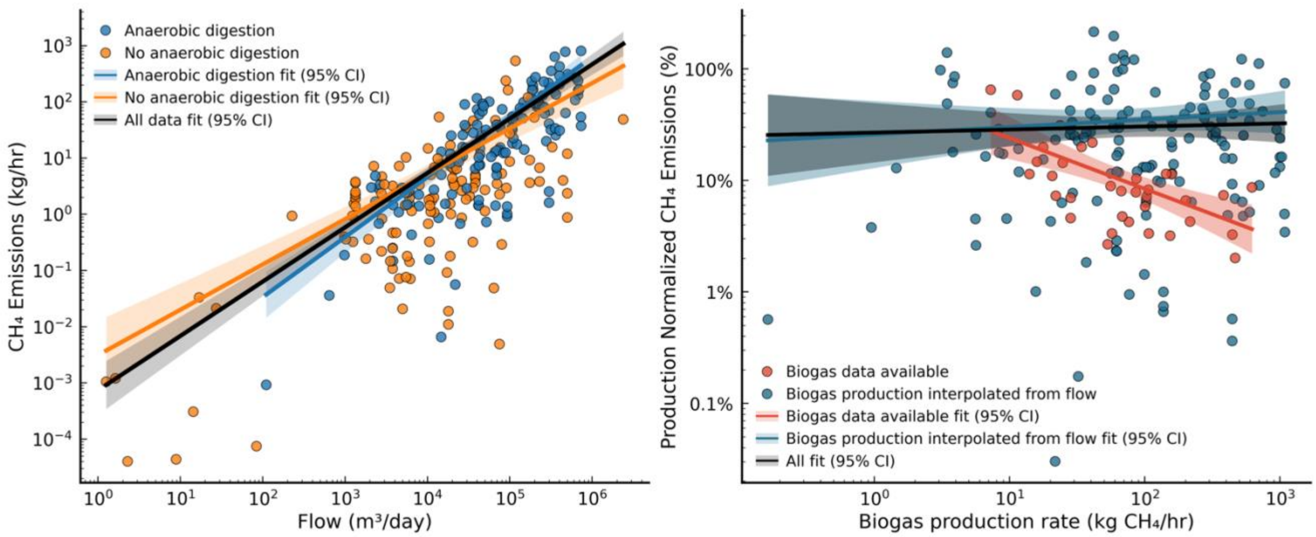

Figure 2: Facility-level methane emissions, absolute rates (a) and production-normalized (b). (a) only include facilities with reported flow rates. For (b), we calculated biogas production rate for facilities that did not report it, as indicated by color. Lines represent equations of best fit: (a) black, all data: $y = 2.63e\text{-}04 \cdot x^{0.97}$ ($R^2 = 0.593$); orange, no anaerobic digestion onsite: $7.72e\text{-}04 \cdot x^{0.81}$ ($R^2 = 0.489$); blue, anaerobic digestion onsite: $1.22e\text{-}04 \cdot x^{1.07}$ ($R^2 = 0.596$)(b) black, all data: $y = 1.33e{+}01 \cdot x^{0.03}$, ($R^2 = 0.001$); red, biogas data available: $y = 5.68e{+}01 \cdot x^{-0.46}$, ($R^2 = 0.401$); blue, biogas production interpolated from flow: $y = 1.27e{+}01 \cdot x^{0.07}$ ($R^2 = 0.006$).

Figure 2b depicts production-normalized emissions vs biogas production rate for all AD facilities in our dataset, using **Equation 1** to estimate biogas where it was not reported in original studies. Estimated emission rates display a high degree of scatter and range from 0.03% – 215%. Loss rates above 100% imply methane leak rate exceeds biogas production, a technically possible scenario (because methane can be produced from unit processes other than AD) that is also highly improbably. These instances, which all occurred at facilities for which we estimated biogas production using **Equation 1**, thus likely represent cases in which we underestimated biogas production.

Notably, we also find diverging trends in production normalized emissions based on whether biogas production was reported in the source data (likely from a plant biogas flow meter) or calculated based on flow rate. For facilities with empirical biogas data, production normalized emissions display a decreasing trend with increasing production rate (R^2=0.4). Physically, this could be explained because the sources of leaks (likely unscrewed flanges or pressure gauges) may maintain a similar physical size across different facilities sizes, while pipes and tanks would increase in size at larger facilities, thus making leaks a smaller proportion of total gas flow. Similarly, for digesters, treatment volume increases at a much greater rate than exposed annular spaces. This finding parallels the oil and gas sector, where low producing well sites disproportionately contribute to overall emissions.[26,27]

However, for facilities where we estimated biogas production from flow rate in the absence of reported biogas data, production-normalized emissions do not decrease with increasing production rate, and display a high degree of scatter and poor fit to the trendline (R^2=0.006 for the power-law equation of best fit). Mean and median production normalized leak rate for these facilities (mean: 34% [95% CI: 28–41%]; median: 23%) is higher than for those where biogas production data was available (mean: 12% [95% CI: 8–17%]; median: 8%). We also observed differences based on data source, potentially indicative of the influence of measurement approach: Moore et al. (2023 and 2025) show no trend between production normalized emissions and our calculated biogas production rate while Song et al. 2023 data display a trend of increasing leak rate with biogas production (**Supplementary Figure S1**). The diverging trends across measurement techniques underscores the importance of validation, ideally through independent single-blind controlled release studies, to prioritize data used in subsequent analysis.

For all facilities without biogas data, our analysis of production normalized emissions rate is dependent on our ability to estimate biogas production from flow rate. This approach does not consider other factors that impact biogas production, such as AD capacity, digester type, implementation of co-digestion, or any other operational parameters (temperature, pH, retention time and loading rate).[16,25,28] While our method aligns with existing process models, a recent study validating WRRF electricity generation models found that many methods may underestimate power generation, although data availability limited drawing any robust conclusions.[29] Nevertheless, the discrepancy in our calculations indicate the importance of consistent data collection across studies, and the need to document biogas data production where possible.

There are additional potential sources of the divergent trends we observed. All facilities reporting biogas data to Fredenslund et al. (2023) and Gålfalk and Bastviken (2025) were in Europe (Denmark and Sweden, respectively). In contrast, Moore et al. (2023 and 2025) conducted measurements in the United States, and Song et al. compiled measurement data globally. Regional differences in treatment, monitoring, maintenance and repair practices may impact methane emissions, and facilities participating in a study by providing biogas data to researchers may be also predisposed to practices that mitigate methane leaks even prior to the measurement campaign itself. Additionally, methane measurements themselves also can have high uncertainty,[30] which would compound as we combined data collected with different measurement techniques and strategies. As discussed previously in the scientific literature, plant-wide methane emissions estimates are meaningfully influenced by measurement technique and study duration.[16]

By synthesizing measurement studies on WRRFs to date, **Figure 2** highlights the importance of further investigating the mechanisms of methane emissions during wastewater treatment. Facilities without anaerobic digestors can be high methane emitters, thus whole-facility measurement studies may be detecting methane produced across the plant, and not just from anaerobic digestion (if present) or solids handling. The high variability across facilities with the same flow rate or biogas production rate indicates the potential role of facility design and operation, not reflected in current emission factors. For example, wastewater industry experts understand that anaerobic digestors with floating covers leak at rates much higher than fixed cover digesters, a key design factor not accounted for in existing measurement studies and inventories. Additionally, whole-facility measurement studies might also detect methane produced from across the plant, not just at anaerobic digestion and solids handling.

Economically finding, capturing, and using currently emitted biogas will require mechanistic insight into specific leak sources within a WRRF. To better understand leak sources, we examined images selected from leak detection surveys conducted by environmental consulting company Brown and Caldwell. All images in **Figure 3** were collected using the Konica Minolta GMP02 infrared camera. False color overlays on images **Figure 3a, b, c** were added for visual clarity given the more complex visual background and generated automatically through Konica Minolta's native software. These images document methane leaks from incinerators (**Figure 3b, c**) and at the influent junction of plants (**3d, e**). Methane generated in the sewer system and released at the headworks of a WRRF could contribute to high measured emissions rates in surveys while facilities themselves may not observe abnormalities in biogas capture rate or overall facility carbon balance.

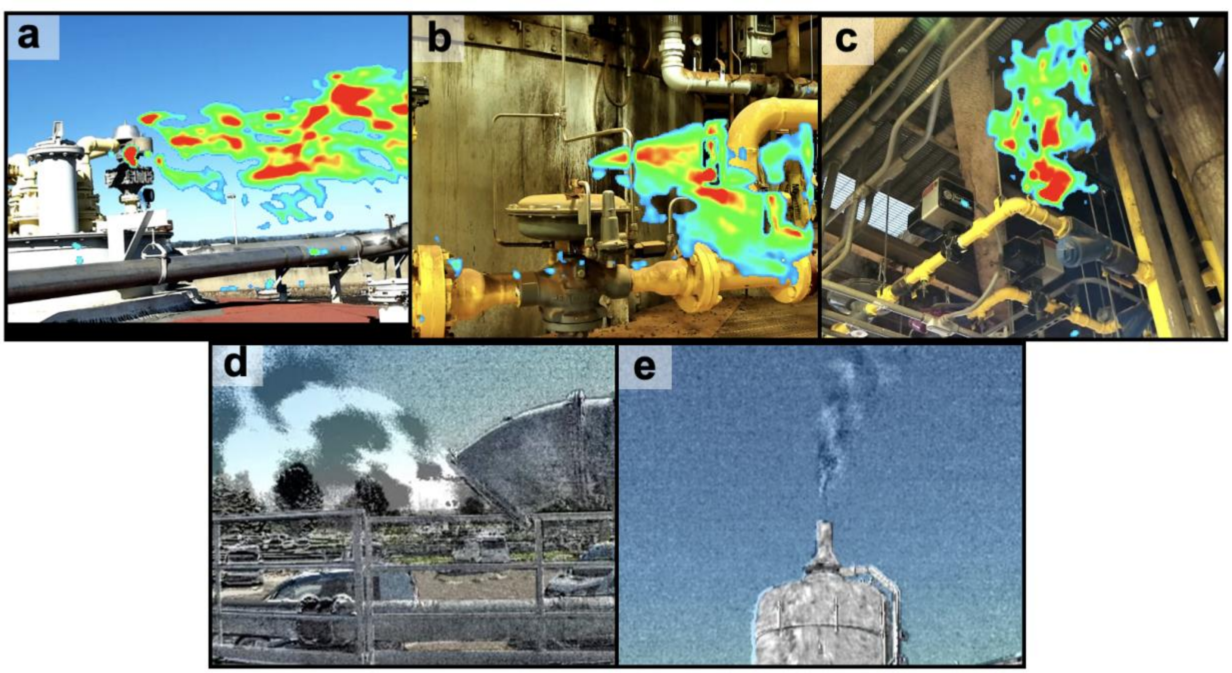

*Figure 3: Methane leaks detected using optical gas imaging at anonymous WRRFs. (**a**) digester pressure release valve (**b**) incinerator natural gas main header (**c**) Incinerator piping (**d**) Raw influent junction chamber vent (**e**) Raw influent junction chamber odor control. Note false color overlays in (a)-(c) were added to improve visual clarity.*

Economic opportunities from leak repairs

We evaluated the economic opportunities that would be available to WRRFs if currently leaked methane were captured for onsite power and heat generation, offsetting purchased electricity and natural gas (**Figure 4**). We consider facilities with flows up to 1.6 Mm^3/day (420 MGD), inclusive of all facilities in the U.S. with CHP, whose size range is represented by the box and whisker plots on the bottom panel of **Figure 4**.

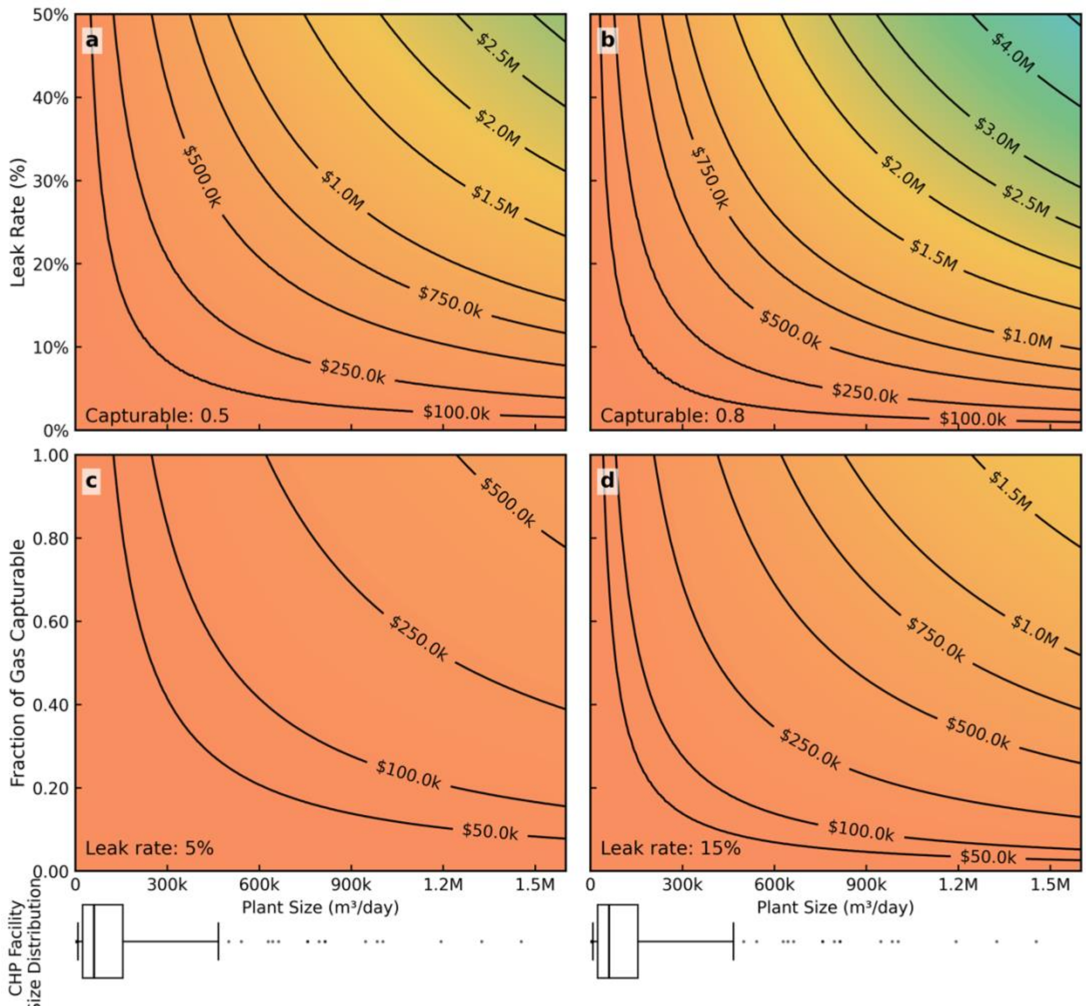

Figure 4: Potential annual revenue stream if methane emissions are captured and used onsite for heat and power, assuming it offsets electricity ($0.09/kWh) and natural gas for heating ($0.008/MJ). The top row fixes the fraction of gas capturable at 0.5 (**a**) and 0.8 (**b**), while varying plant-wide leak rate (y-axis) across facilities of different sizes (x-axis). The bottom row fixes leak rate at 5% (**c**) and 15% (**d**), while varying the fraction of gas that is capturable (y-axis). Box and whisker plots at the bottom represent the size of facilities in the United States with onsite CHP, and use the same x-axis as top panels. Boxes represent 25th and 75th percentiles, with midlines indicating median, and whiskers extend to the 5th and 95th percentile.

In **Figures 4a, b** we varied production-normalized leak rate from 0 to 50%, informed by the range of production normalized emissions from facilities where both biogas and emissions data are available (range: 2 – 65%, mean: 12.32%, median: 7.92%, standard deviation: 16%).[8] We fixed the fraction of fugitive emissions recoverable for power generation at 0.5 and 0.8, reflecting improvements that can be made with relatively minor repairs[8] or more substantial investments, respectively. The largest 5% of facilities by flow may accrue over $100,000 in less than one year with initial leak rates as low as 5% (fraction gas capturable: 0.5) and 3% (fraction gas capturable: 0.8). Notably, these rates are both below the median value of 7.9% for facilities that reported biogas production. Additionally, while leak rates may not often exceed 25%, when this occurs capturing lost methane could increase revenue generation equivalent to several million dollars in both electricity and heat.

For **Figures 4c,d** we fixed methane leak rates at values at 5% and 15%, and varied fraction of leaked gas that can be captured from 0.0 (no capture) to 1.0 (complete capture), although both extremes of this distribution are unlikely. If the fraction of gas capturable exceeds 0.6 with a 5% leak rate, we found that the largest 5% of facilities could potentially increase revenue by $100,000 or more, a threshold that can be reached by the largest 25% of facilities if leak rates reach 15%. Similarly, across all scenarios depicted in **Figure 4**, the smallest 50% of facilities often may accrue less than $100,000 per year from gas capture. For these facilities, economic benefits from gas capture may require other drivers for leak detection and repair which, while important, may not directly translate to improved energy efficiency or cost reductions. These may include industry concerns regarding worker safety, health, odors, and climate impact.

To determine the national impact of methane leaks, we applied this economic analysis to U.S. WRRFs with energy recovery. Monte Carlo results vary substantially across the different scenarios we used for leak rate distribution, with mean values ranging from $20.7M [95% CI: $20.3M – $21.1M] under the conservative heavy-tail distribution to $72.4 [95% CI: $71.0M – $73.8M] when bootstrapping leak rates from the entire dataset of production normalized emissions. The differences in mean and median results across these simulations reflects the importance of improving available data on both leaks and biogas production. However, across all scenarios, millions of dollars are lost annually to methane leaks, demonstrating that regardless of the economics at an individual facility, the cumulative impact nationwide can be substantial.

Table 1. National opportunity cost of fugitive methane leaks, estimated using a Monte Carlo simulation with three sampling scenarios for facility biogas leak rate: bootstrapping from all biogas leak data (including where biogas production rate was interpolated from flow rate), bootstrapping from only facilities with reported biogas production rate and leak rate, and assuming a lognormal (heavy-tail) distribution with a median leak rate of 5%.

Leak Rate Distribution	*Median [2.5%–97.5%]*	*Mean* [95% CI]
Bootstrap – all data	$46.9M [$1.67M – $276M]	$72.4M [$71.0M – $73.8M]
Bootstrap – reported biogas production	$23.9M [$6.55M – $191M]	$36.9M [$36.1 – $37.7]
Heavy tail distribution (median: 5%)	$14.8M [$2.96M – $73.5M]	$20.7M [$20.3 – $21.1]

Discussion

Economics of fugitive methane in the United States will vary substantially depending on the facility size and nature of the leaks. By compiling recent methane leak measurement studies at WRRFs, we highlight the overall trends observed to date, as well as current limitations in measurement strategy and data collection. To the best of our knowledge, this is the first data synthesis for WRRFs estimating production normalized emissions across site-level measurements. Our results highlight the need for uniform data collection in this field to more readily allow for cross-comparison. Specifically, we recommend future studies report facility flow rate and biogas production during the measurement period, where possible, and otherwise provide annual averages.

We also observed differing trends in production normalized emissions across measurement technologies. Independent verification across a range of release rates and mimicking the conditions of WRRF emissions could advance technology development and facilitate interpretation of results, as has been the case for the oil and gas sector.[31] A recent single-blind landfill controlled-release study found disparities in quantification performance between vehicle- and drone-based platforms. Vehicles using Gaussian plume dispersion model underestimated compared to the drone-based flux plane method, which had reduced scatter and no downward bias.[32] However, without additional data, these results are difficult to reconcile with the findings of our study, where we calculated higher production normalized emissions from studies using vehicle-based methods. Higher emissions rates, including those >100%, could be the result of either measurement inaccuracy or uncertainty in estimating biogas production. Improving data access and availability on both biogas production and technology validation would benefit future analyses.

Our economic analysis indicates the largest facilities in the country will be able to recover substantial economic value from leaked methane, additional revenue which may cover the costs of conducting surveys and repairs. Implementing leak detection and repair programs at the largest 15 WRRFs in the United States (the outliers in the box and whisker plots at the bottom of **Figure 4**) could accrue these facilities economic benefits while also providing valuable data on the nature of leaks to inform methane mitigation strategies at smaller plants where economics may be less favorable.

While we consider revenue lost by methane leaks, and do not account for costs of leak surveys and repairs, which can vary widely and are poorly characterized in the scientific literature. For example, based on the authors' familiarity with the industry in the United States, environmental consulting firms charge \$30,000 – \$60,000 for a leak detection survey with optical gas imaging (OGI). In contrast, and highlighting the complexity of wastewater treatment plants, OGI surveys of oil and gas sites were assumed to be \$600/site, where each site contained on average 2 wellheads.[10] Note that a recent techno-economic analysis of fugitive methane in Europe reported leak detection surveys cost €400 to €1200 (\$432 – \$1,300) per day depending on the technology,[9] rates unlikely in the United States given typical hourly consultant fees.

Repair costs are similarly variable, and data is primarily from Europe. One study from Denmark reported that the cost of relatively minor repairs in 2021 ranged from 0.1 – 22.5 million DKK[8], equivalent \$18,000 – \$4M in \$2023. Another study interviewed European industry stakeholders, and found relevant repair costs can range from relatively minor fixes to flanges (\$30–\$1,000) and connections (\$10–\$300) to more substantial repairs to digester domes (\$32,000–\$38,000) and membrane storage repairs (\$16,000–\$27,000).[9] However, these estimates were for agricultural digester facilities, which may have different design standards than municipal ones. In contrast, one U.S.-based wastewater treatment utility we spoke with indicated replacing a pressure release valve costs around \$10,000. Based on our knowledge of the U.S. industry, the costs of major leak remediation upgrades, such as replacing a floating cover with a fixed cover, may reach \$2 – \$7 million per digester. Given the wide range in available data, and lack of information on how repair costs relate to leak sizes, additional data collection is needed before repair cost can be factored into economic studies.

Our analysis considers the economic impacts of methane leaks over 1 year, and future analysis should consider the temporal aspects of leak detection and repair (LDAR). In one study in the oil and gas sector, over 90% of leaks identified in an initial survey were not present in a follow-up survey 0.5–2 years later.[34] Another study evaluated the impact of a California regulation requiring quarterly LDAR inspections at oil and gas facilities, and found the ratio of leaks identified to components surveyed dropped from ~90% to under 20% over a two-year period.[11] However, a recent study comparing different strategies for detecting methane leaks in the Canadian oil and gas sector found that multiple strategies (aerial surveys alongside OGI) may be necessary to mitigate total emissions.[33] Nonetheless, investments to fix leaks at WRRFs will likely provide economic benefits beyond the year of the initial investment. Additionally, facilities need not hire external service providers to conduct repeated surveys: the cost of an OGI camera can be around \$200,000, corresponding to an annualized cost of \$28,500/year over a 10-year lifetime (calculated with a 7% discount rate). Utilities or local governments may purchase this equipment for shared use across multiple facilities, further reducing costs.

There are several other limitations of this work and opportunities for future refinements. Our analysis only considers facilities with existing anaerobic digestion and energy recovery infrastructure. Moderately sized facilities may find favorable economics through other revenue streams, such as by upgrading biogas to renewable natural gas or vehicle fuel, which can be profitable when considering federal and state-level incentives.[35,36] Alternative high-value bioproducts are currently only economical at large scale, but research and development may drive down costs.[37] However, these pathways will require upfront capital investment and are beyond the scope of the current study. Additionally, electricity and natural gas prices vary widely across the United States, and in some states is over double the median value used in this study (**Supplementary Figure S2**), further incentivizing economics of energy recovery.

Capturing fugitive methane leaks is key for reducing the climate impact of WRRFs, and this work evaluates current knowledge gaps and the economic landscape for methane leak detection and repair. Economic favorability relies heavily on biogas leak rates, which vary widely across current published literature and depend on poorly characterized biogas production rate. The proportion of gas that can be captured for electricity production also impacts economics, underscoring the importance of establishing component-level emission factors for WRRFs and further characterizing the underlying mechanisms causing methane emissions. With current infrastructure, economics appear favorable for the largest facilities with onsite energy recovery capabilities. However, for moderately sized or small facilities, climate or safety considerations may be a more salient factor in motivating methane leak detection and repair programs and should be considered.

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Supplementary Information: Evaluating economic opportunities and challenges for energy recovery from methane leaks during wastewater treatment

Supplementary Methods

Statistical analysis for linear regression

We conducted a linear regression on the biogas and flow data included in **Figure 1a** to obtain **Equation 1** in the main text. We used a least squares regression with the y-intercept fixed at the origin (**Equation S1**). We fixed the y-intercept at the origin because if facility flow rate is zero, biogas production must also be zero.

$$\beta = \frac{\sum x_i y_i}{\sum x_i^2} \qquad \textbf{Equation S1}$$

Where:

x_i	Reported facility flow rate (m^3/day)
y_i	Reported facility biogas production (kg CH_4/hr)
$\hat{y}_i$	Fitted value for biogas production (kg CH_4/hr)
$\bar{y}$	Mean of reported facility biogas production (kg CH_4/hr)

Table S1 summarizes the results of the linear regression and key statistical parameters, described further below.

Table S1 Linear regression results for biogas production at WRRFs based on flow rate

Symbol	Description	Value
n	Sample size	42 facilities
β	Slope of equation of best fit	$0.0015 \frac{kg\ CH_4/hr}{m^3\ wastewater/day}$ $= 0.0355 \frac{kg\ CH_4}{m^3\ wastewater}$ $= 1.7756 \frac{MJ\ biogas}{m^3\ wastewater}$
R^2	Coefficient of determination - centered	0.4968
R_0^2	Coefficient of determination - uncentered	0.7158
ν	Degrees of freedom	1
$\hat{\sigma}$	Standard error of the regression	228 kg CH_4/hr
$se(\beta)$	Standard error of the slope β	0.00021

We calculated standard error of the regression $\hat{\sigma}$ (**Equation S2**) and of the standard error of the slope (**Equation S3)**. For $\hat{\sigma}^2$, we used the standard equation used for linear regressions with a y-intercept,[1] and assumed one degree of freedom instead of two to account for the fixed y-intercept.

$$\hat{\sigma}^2 = \frac{\sum_{i=1}^{n}(\hat{y}_i - \bar{y})^2}{n - \nu} \qquad \textbf{Equation S2}$$

$$se(\beta) = \frac{\hat{\sigma}}{\left(\sum_{i=1}^{n}(x_i - \bar{x})^2\right)^{1/2}} \qquad \textbf{Equation S3}$$

For coefficient of determination (R^2) values reported in the main text, we used **Equation S4** for centered R^2. Note that many statistical packages, including Excel, calculate the coefficient of determination for a linear regression with a fixed y-intercept using an uncentered R_0^2 value (**Equation S5**), in which the mean is not subtracted from y_i in the denominator of the equation. As noted elsewhere, R^2 and R_0^2 are different values and not directly comparable. Based on statistical best-practice guidelines[1,2], we chose to report R^2 in the main text but also report R_0^2 in **Table S1**.

$$R^2 = 1 - \frac{\sum_{i=1}^{n}(\hat{y}_i - \bar{y})^2}{\sum_{i=1}^{n}(y_i - \bar{y})^2} \qquad \textbf{Equation S4}$$

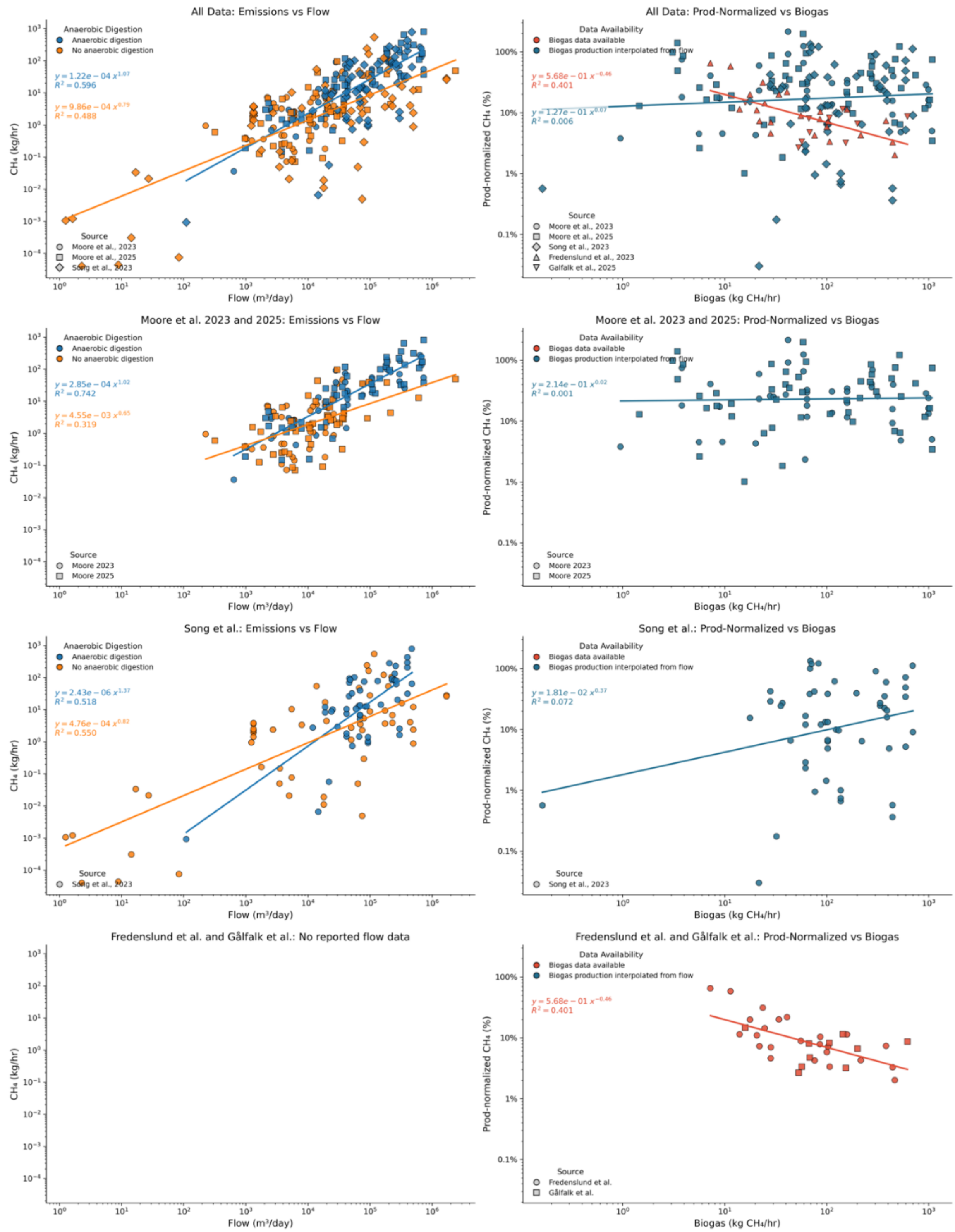
$$R_0^2 = 1 - \frac{\sum_{i=1}^{n}(\hat{y}_i - \bar{y})^2}{\sum_{i=1}^{n} {y_i}^2} \qquad \textbf{Equation S5}$$

Summary of methane leak measurement data sources

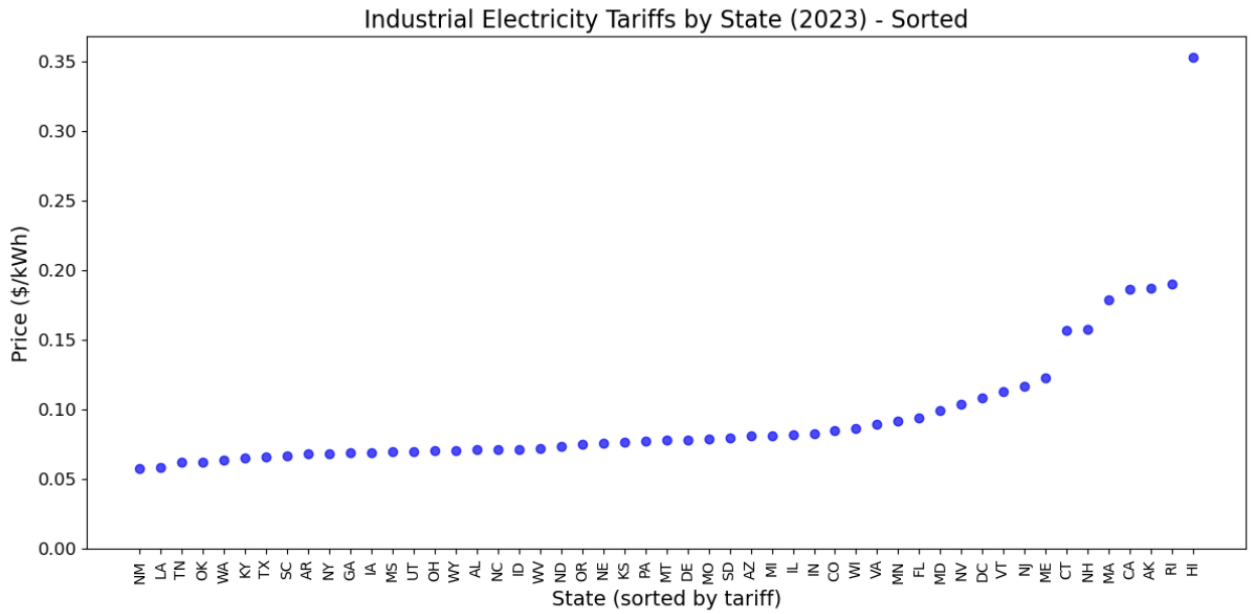
Table S2 Summary of data sources for methane leak rates at WRRFs and key reported parameters

Source	Measurement Approach	Sample Size	Reported Biogas Production?	Reported Facility Flow Rate?
Song et al., 2023[3]	Various: literature review	112	No	Yes
Fredenslund et al., 2023[4]	Tracer gas dispersion method	25	Yes	No
Moore et al., 2023[5]	Vehicle-mounted sensor	83	No	Yes
Gålfalk and Bastviken, 2025[6]	Done-mounted sensor	13	Provided upon request	No
Moore et al., 2025[7]	Vehicle-mounted sensor	109	No	Yes

Supplementary Results



Supplementary Figure S1: biogas measurement data by source. Row 1 (top): reproduction of Figure 2 in the main text; Row 2:data from Moore et al. 2023 and Moore et al., 2025; Row 3: Data from Song et al., 2023; Row 4 (bottom): data from Fredenslund et al., 2023 and Gålfalk et al., 2025.



Supplementary Figure S2: Average industrial price of electricity at the state level (including Washington DC) in 2023, according to the U.S. Energy Information Administration. Mean: $0.0958/kWh, median: $0.08/kWh.

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