Waste Management 128 (2021) 122–131

Contents lists available at ScienceDirect

Waste Management

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

The implications of facility design and enabling policies on the economics of dry anaerobic digestion

Sarah Josephine Smith [a,b,*], Andrew J. Satchwell [a], Thomas W. Kirchstetter [a,b], Corinne D. Scown [a,c,d]

[a] Energy Technologies Area, Lawrence Berkeley National Laboratory, 1 Cyclotron Road, Berkeley, CA 94720, United States
[b] Department of Civil and Environmental Engineering, University of California, Berkeley, Berkeley, CA 94720, United States
[c] Biosciences Area, Lawrence Berkeley National Laboratory, 1 Cyclotron Road, Berkeley, CA 94720, United States
[d] Joint BioEnergy Institute, 5885 Hollis Street, Emeryville, CA 94720, United States

ARTICLE INFO

Article history:
Received 18 September 2020
Revised 30 March 2021
Accepted 25 April 2021
Available online 11 May 2021

Keywords:
Dry anaerobic digestion
High-solids anaerobic digestion
Solid-state fermentation
Renewable natural gas
Biofuel
Organic waste management
Landfill diversion

ABSTRACT

Diverting organic waste from landfills provides significant emissions benefits in addition to preserving landfill capacity and creating value-added energy and compost products. Dry anaerobic digestion (AD) is particularly attractive for managing the organic fraction of municipal solid waste because of its high-solids composition and minimal water requirements. This study utilizes empirical data from operational facilities in California in order to explore the key drivers of dry AD facility profitability, impacts of market forces, and the efficacy of policy incentives. The study finds that dry AD facilities can achieve meaningful economies of scale with organic waste intake amounts larger than 75,000 tonnes per year. Materials handling costs, including the disposal of inorganic residuals from contaminated waste streams and post-digester mass (digestate) management, are both the largest and the most uncertain facility costs. Facilities that utilize the biogas for vehicle fueling and earn associated fuel credits collect revenues that are 4-6x greater than those of facilities generating and selling electricity and 10-12x greater than facilities selling natural gas at market prices. The results suggest important facility design elements and enabling policies to support an increased scale of organic waste handling infrastructure.

1. Introduction

Waste management activities, including landfills, wastewater treatment, and composting, were estimated to account for 2% of U.S. greenhouse gas (GHG) emissions in 2016, based on the 100-year global warming potential of the emissions (U.S. EPA, 2018a). The majority of these emissions occur when organic waste decomposes anaerobically in landfills and produces methane (CH_4), a potent short-lived climate pollutant (Duren et al., 2019). While landfill gas capture systems operate at approximately half of U.S. landfills (U.S. EPA, 2020), these systems may only capture 66–88% of gases created over the lifetime of the landfill (Barlaz et al., 2009). Prior life-cycle assessments agree that diverting organic waste from landfills to waste-to-energy systems or composting achieves substantial net GHG emissions reductions (Morris et al., 2013; Nordahl et al., 2020). In addition to the climate benefits, diverting the organic fraction of municipal solid waste (OFMSW) from landfills also reduces the need to expand landfill capacity, improves leachate quality (Jordan et al., 2020), and provides an opportunity to generate renewable energy and recycle the organic material and nutrients back to the soil (Breunig et al., 2019). These advantages of landfill diversion have motivated several large states and cities in the United States (U.S.) to establish aggressive municipal solid waste diversion goals, which will require extensive build-out of organic waste handling infrastructure (Satchwell et al., 2018).

Anaerobic digestion (AD) facilitates the decomposition of organic waste in a controlled, oxygen-limited environment and captures the resulting biogas (an approximately 50/50 mixture of CH_4 and CO_2) to either generate electricity or upgrade to pure methane for use in pipelines or vehicles. AD also recovers nutrients through the management of remaining solid digestate (Breunig et al., 2019). In the U.S., AD has traditionally been used in the treatment of municipal wastewater (U.S. EPA, 2018b) and, increasingly, at dairy farms (AgSTAR, 2018). While a small number of these traditional facilities accept municipal and agricultural organic waste streams to co-digest with the human or animal wastes, new dedicated AD facilities have been built in the U.S. within the last decade with the sole purpose of processing organic waste diverted from landfills (Linville et al., 2015). Standalone AD of OFMSW presents

* Corresponding author at: Energy Technologies Area, Lawrence Berkeley National Laboratory, 1 Cyclotron Road, Berkeley, CA 94720, United States.
E-mail address: sjsmith@lbl.gov (S.J. Smith).

https://doi.org/10.1016/j.wasman.2021.04.048
0956-053X/

Nomenclature

List of Acronyms

AD	Anaerobic digestion
BioMAT	Bioenergy Market Adjusting Tariff
CHP	Combined heat and power
CI	Carbon intensity
CNG	Compressed natural gas
GHG	Greenhouse gas
LCOD	Levelized cost of disposal
LCFS	Low Carbon Fuel Standard
MRWMD	Monterey Regional Waste Management District
O&M	Operations and maintenance
OFMSW	Organic fraction of municipal solid waste
RIN	Renewable Identification Number
RNG	Renewable natural gas
SI	Supplementary Information
SSFSC	South San Francisco Scavenger Company
SSO	Source-separated organics
ZWEDC	Zero Waste Energy Development Company

challenges that are not faced by wastewater treatment and dairy digester facilities, namely a highly variable feedstock, inorganic contamination, and low moisture content relative to liquid wastes. Some of these challenges can be mitigated through the use of dry AD (also known as high-solids or solid-state digestion) facilities that process waste in their existing solid form, making dry AD of OFMSW a promising technology to meeting landfill diversion, GHG emission, and renewable energy goals (Nordahl et al., 2020; Preble et al., 2020; Satchwell et al., 2018). Dry AD is ideal for processing waste with solids content between 20 and 40% (Rocamora et al., 2020). However, experiences from a growing set of dry AD facilities indicate that the success of these facilities hinges on operational risks associated with mixed feedstocks, as well as the cost of handling residual solids and the ability to monetize renewable energy outputs.

These technical and economic complexities make the long-term viability of dry AD facilities difficult to predict. This work employs technoeconomic analysis to provide a comprehensive cost and revenue analysis for dry AD facilities based on real-world data, covering a variety of technology and design options. In a review of AD literature, Rajendran and Murthy (2019) found that environmental life-cycle assessment papers vastly outnumber technoeconomic analysis, and existing technoeconomic analysis has limited focus on OFMSW and no focus on dry or high-solids AD. Nordahl et al. (2020) conducted a deep-dive into the environmental impacts of dry AD of OFMSW, including a comparison of bioenergy utilization and digestate management pathways, but did not assess facility economics. Angelonidi and Smith (2015) present operational and cost data from nine AD facilities in Europe, the majority of which are high-solids or dry AD, but do not examine facility profitability or energy-related revenue streams. Increasingly, systems optimization studies have incorporated wet or dry AD into the portfolio of assessed technologies. Ascher et al. (2019) assessed wet AD with electricity generation and waste heat recovery for management of organic waste in Glasgow, Scotland, and found that in order to be profitable, digestate by-products must be sold or a carbon tax of at least $140/tonne must be implemented. Waste tipping fees were not considered in this study. Finally, Tominac et al (2021) examine the environmental impacts and high-level economics (using a single dollars per tonne cost metric) of managing municipal organic waste in Milwaukee, Wisconsin via landfill, compost, dry AD, and wet AD, with biogas being used to generate electricity and digestate land-applied. Independent of tipping fee and specific facility siting and transportation dynamics, they find that dry AD is preferred for 12% of organic waste. Economic analyses have been conducted for wet AD of municipal solid waste, highlighting the dynamics between tipping fee, energy revenue, and profitability (Rajendran et al., 2014; Sanscartier et al., 2012), but these studies do not consider dry AD in particular, nor do they assess multiple energy utilization pathways, the cost of disposing of post-AD solid digestate, or current bioenergy incentive frameworks in the U.S. Lastly, Khan et al. (2016) found that a tipping fee of $70–100 per tonne of waste was required for a dry AD facility to earn an IRR of 10% in Alberta, Canada. In this analysis, only an electricity generation pathway was examined, digestate management costs were not included, and modeled capital costs were significantly lower than recently reported values.

This study is the first comprehensive technoeconomic analysis of dry AD facilities processing OFMSW, using a new model with empirical data from operational facilities in California. The model is used to determine the key cost and revenues, sources of uncertainty, and potential economic competitiveness with landfills for dry AD across a wide range of facility scales and for three energy utilization pathways. Careful attention is provided to specific policy and regulatory landscapes in California that impact costs and revenues, and to factors that are of particular interest to dry AD facilities such as digestate management.

2. Methods

The dry AD economic model developed in this study calculates the annual costs and revenues of a privately-owned dry AD facility over a 25-year lifetime (see Supplementary Information (SI) for details on annual cash flow and discounting calculations). An overview of the modeling framework is shown in Fig. 1. A set of facility operational modeling equations, described in Section 2.1, determine the mass and energy flows of the waste as it moves through sorting and digestion, and the generated biogas as it moves from creation (in the digester) to the final energy delivery. Key assumptions for this analysis are shown in Table 1. The model assesses three pathways for biogas utilization: (1) on-site combined heat and power (CHP) generation using biogas ("Electricity scenario"), (2) upgrading biogas to pure methane, referred to as renewable natural gas (RNG) or biomethane, and compressing for on-site vehicle fueling ("RNG Vehicle Fuel scenario"), or (3) upgrading biogas to RNG for natural gas pipeline injection with unspecified end-use ("RNG Non-Vehicle scenario"). Cost and revenues are calculated for each step in the model, as denoted by the red and green dollar symbols in Fig. 1 and described in Section 2.2. Capital, operational, and labor costs are calculated for the sorting and digestion facility, the gas upgrading or energy conversion equipment, and the energy delivery infrastructure. Trucking and disposal costs for the inorganic residuals and post-digester digestate are also modeled. Revenues for a given energy product are calculated based on current California market and policy incentive landscapes, while the revenue associated with incoming waste into the facility (i.e., the tipping fee) is not modeled directly but is instead an output of the model. Key cost parameters are shown in Table 2.

Model parameters were developed in part on empirical data from full-scale operational facilities, particularly the Zero Waste Energy Development Company (ZWEDC) facility in San Jose,

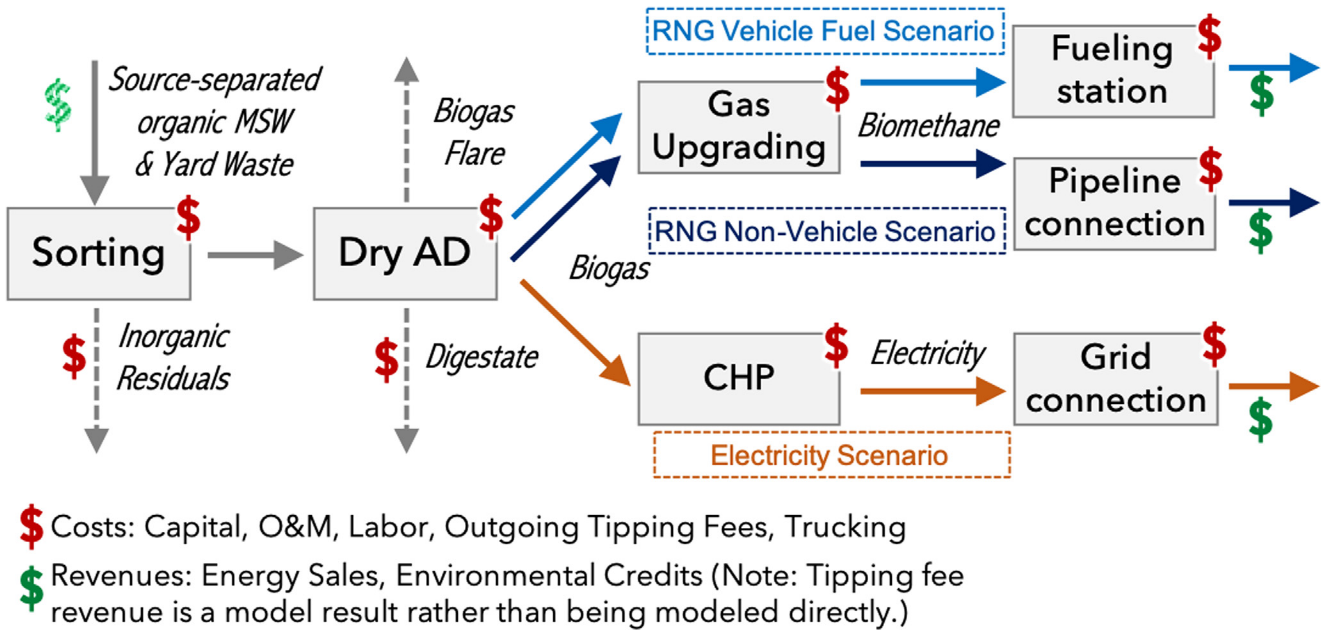

Fig. 1. Model process flow diagram showing the major costs and revenues captured in the model for each of the three energy scenarios.

California. Additional dry AD facilities referenced include the Blue Line Transfer Inc. and South San Francisco Scavenger Co. (SSFSC) digester in South San Francisco, California and the Monterey Regional Waste Management District (MRWMD) digester in Monterey, California (see SI for a description of these facilities). The modeling approach includes an uncertainty analysis to capture variations in potential facility performance (i.e., Low, Base, and High Performance scenarios) and uncertainty in costs and revenue streams (i.e., Low, Base, and High Cost scenarios; note that the Low Cost scenario contains both the lower bound of costs and the upper bound of revenues, and vice versa for the High Cost scenarios). Facilities with rated digestion capacity of 25,000 to 300,000 metric tonnes of wet waste (hereafter noted as tonnes) per year are modeled to assess economies of scale. Though facility operations and cost categories and trends are generalizable, costs were modeled with a focus on California, and therefore other regions within the U.S. and globally may lie beyond our uncertainty ranges if feed-in tariffs, labor costs, and other market prices are dramatically different.

Table 1
Key operational parameter assumptions.

Category	Base Performance	Upper and Lower Bounds Modeled	Units
Biogas yield	85	65–105	scm/tonne waste
Biogas methane content	55	50–60	%
Portion of biogas flared (CHP facilities)	25	15–35	%
Portion of biogas flared (biogas upgrading facilities)	15	10–20	%
Methane loss during biogas upgrading	3	1–10	%
CHP electrical efficiency	40	36–44	%

scm = standard cubic meter, CHP = combined heat and power.

2.1. Facility operations modeling

Table 1 shows the key operational parameters used in the model, including ranges representing uncertainty, and Section 2 of the Supplementary Information provides more detail on param-

Table 2
Key cost assumptions.

Cost Category	Base Cost	Upper and Lower Bounds Modeled	Cost Units
Residuals landfilling fee	50	27–138	\$/tonne residuals
Digester capital cost	$15,900 * (\text{facility capacity in tonnes/year})^{0.7}$	±20%	\$
Upgrading Capital + O&M	$444,000 + 0.18 * (\text{biogas treated annually in scm})$	±20%	\$/year
Labor (total cost to employer)	92,300	±20%	\$/FTE
Digestate trucking	11	0 to + 50%	\$/tonne digestate
Digestate disposal	50	±33%	\$/tonne digestate
CHP capital	$13,150 * (\text{engine capacity in kW})^{0.75}$	±30%	\$/engine
Electrical interconnection	200,000	100,000–500,000	\$/5 MW capacity
Electricity selling price	100	60–127	\$/MWh
CNG fueling station	7	±25%	\$/MJ/d
CNG selling price	0.015	−10%; +30%	\$/MJ
LCFS credit value	150	100–200	\$/tonne CO_{2e}
RIN (D5) value	0.6	±50%	\$/RIN
Natural gas interconnection	1	0.5–2	\$M
Natural gas selling price	0.003	±20%	\$/MJ

CHP = combined heat and power; CNG = compressed natural gas; LCFS = Low Carbon Fuel Standard; RIN = Renewable Identification Number, FTE = full-time equivalent.

eters not described here. The facility modeled in this study accepts source-separated organics (SSO) from municipal residential and commercial sources and municipal yard waste. The level of inorganic contamination in SSO was assumed to be 40%, as reported by currently operating facilities; the facility employs manual sorting to remove 85% of the contamination, then mixes the SSO with yard waste in a 3-to-1 ratio (see SI). We assumed an average biogas yield of 85 standard cubic meters of biogas per wet metric tonne of waste (scm per tonne) and a range of 65 to 105 scm per tonne for the combined SSO and yard waste in the digester. Our base performance and bounded assumptions are consistent with assumptions used in recent literature (Tominac et al., 2021) as well as empirical data on dry AD system biogas yields: the SSFSC and ZWEDC facilities in California reported 94–100 and 62–78 scm per tonne, respectively, while Angelonidi and Smith (2015) report 78–90 scm per tonne for facilities in Europe that process mixed food and green waste and 50–106 scm per tonne for facilities with dry continuous systems. These biogas yield ranges depend on various digester design parameters such as feedstock moisture content, operating temperature, and retention time. We also assumed the generated biogas has a methane content of 55% (range 50–60%) by volume based on empirical facility data from ZWEDC and others (Angelonidi and Smith, 2015; Ong et al., 2014). The remaining portion is carbon dioxide (CO_2) with trace contaminant compounds.

2.2. Facility cost modeling

The residuals (i.e., inorganic contamination) separated from the waste once it reaches the AD facility must be landfilled; we estimated the cost of disposing residuals based on typical landfill tipping fees in California (see Table 2). California tipping fees are similar to the average tipping fees across the U.S. in terms of both the average and the range of values (CalRecycle, 2015; Environmental Research & Education Foundation, 2018). We assumed the cost of transporting the residuals to the landfill are negligible, as the total tonnage of inorganic contamination hauled out is small compared to other inbound and outbound tonnages, and digestion facilities are often co-located with landfills or other waste transfer operations (e.g., as is the case for the observed facilities). A small portion of residuals will be recyclable, but the monetary value of these materials, if any, is negligible.

Digester capital costs were modeled using a standard exponential equation (see Table 2). The capital cost equation for the base scenario was calibrated to the ZWEDC facility capital cost of $43.5 M (see SI for details, including comparison to other reported facility costs). A scaling factor of 0.7 was assumed; though wet AD facilities exhibit a scaling factor of 0.6 (Sanscartier et al., 2012), previously reported costs of dry batch AD facilities appear to scale more linearly with facility capacity (Angelonidi and Smith, 2015). This is likely due to the fact that dry AD facilities, particularly those with batch processes, operate multiple identical reactors in parallel and therefore do not experience the same economies of scale as simply increasing the size of a wet AD tank. We assumed annual operating costs (not including labor) of 5% of capital cost based on ZWEDC facility data. Gas upgrading costs (for removal of CO_2) for the RNG energy utilization scenarios take a linear form, based on combined capital and operational costs from European data collected from 2007 to 2008 (Ong et al., 2014). Labor was separated into three categories: overhead, operations, and sorting, that scale differently with facility size and operational parameters (see SI for assumptions). Annual employment costs, including employer-paid benefits, were based on California-specific wage data (see Table 2; see SI for details).

The AD process and subsequent short-term in-vessel composting of the digestate cause a reduction in solid waste mass (30%, according to ZWEDC tonnage reports) due to the transformation of solid mass into biogas as well as loss of moisture. We modeled an arrangement where the facility pays to haul and dispose of digestate at a third-party facility. For example, ZWEDC sends digestate to be composted off-site, and other known California AD facilities (i.e., SSFSC and MRWMD) also send their digestate for off-site management. Base digestate trucking costs and tipping fees are based on ZWEDC's actual costs. The low-cost scenario assumes that facilities are co-located with compost operations, so trucking costs are negligible, and tipping fees were modeled as the California statewide median for yard waste at compost facilities. This is a reasonable lower bound given that digestate from facilities processing mixed organic waste will be more heterogeneous and contaminated than yard waste, so fees would likely be higher to reflect the management challenges posed by moisture, odor, and inorganic contamination (Cotton, 2019).

2.2.1. Electricity generation

Assumptions regarding the combined heat and power (CHP) equipment efficiency and costs are described in the SI. There are various arrangements under which dry AD facilities can sell electricity in California for a premium (see compensation assumptions in Table 2). The highest price typically achievable is through the state's Bioenergy Market Adjusting Tariff (BioMAT) program, which for municipal waste digesters is currently $127 per MWh (CPUC, 2018). This price is multiplied by a seasonal- and time-of-day-varying factor that represents the value of the electricity to the grid, but we assumed facilities do not have long-term biogas storage or excess generator capacity and therefore generate roughly constant power output throughout the year. If unable to enroll in the BioMAT program, a facility could at a minimum obtain a wholesale price for electricity, which generally ranges from $40–60 per MWh in California (U.S. Energy Information Administration, 2019), plus any value from renewable energy credits. A third option is to enter a power purchase agreement with an individual energy consumer or aggregator; examples of this have been seen at wastewater treatment plant digesters in California. For example, East Bay Municipal Utility District sells power directly to the neighboring port of Oakland for $58 per MWh (Hake et al., 2017), while Central Marin Sanitation Agency has a power purchase agreement with Marin Clean Energy, a municipal energy aggregator, for approximately $105 per MWh (CMSA, 2018).

2.2.2. RNG Vehicle fuel

For the RNG Vehicle Fuel scenario, the facility pays to construct and maintain a fueling station to sell renewable compressed natural gas (sometimes referred to as R-CNG, where the vehicle fuel more broadly, including from fossil resources, is referred to as CNG) (see SI for fueling station costs). RNG is modeled as sold (or self-consumed) at a value equal to current California fossil CNG fuel prices (see Table 2), and the facility is eligible for both state and federal renewable fuel incentives.

California's Low Carbon Fuel Standard (LCFS) awards credits based on the carbon intensity (CI) of the renewable fuel, calculated on a facility-by-facility basis with a limit on the GHG credit a facility can incorporate based on avoided landfill methane emissions (California Air Resources Board, 2018; Schwarzenegger, 2007). Two dry AD facilities are currently certified through the LCFS system; the SSFSC facility, which processes a mix of food and yard waste, has a CI of −22.93 grams of CO_2 equivalents (g CO_{2e}) per MJ, while a southern California facility digesting only yard waste is certified at a CI of 0.34 g CO_{2e} per MJ (California Air Resources Board, 2019a). Assuming a conservative CI of −10 g CO_{2e} per MJ for the modeled AD facility, the resulting fuel would earn credits at a rate of 104 g CO_{2e} per MJ of fuel (California Air Resources Board, 2019b). Historically, credit prices have varied from $20 to

120 per tonne CO_{2e} mitigated, but from July 2018 to August 2020 they have consistently hovered near the ceiling price of $200 per tonne CO_{2e} (California Air Resources Board, 2020). Future credit amounts and market conditions leave significant uncertainty in the long-term price and, therefore, we model LCFS credit prices of $100–200 per tonne CO_{2e}.

Federal Renewable Fuel Standard credits, called Renewable Identification Numbers (RINs), are earned for every 77,000 Btu (81 MJ) of fuel produced. We conservatively used "D5" market prices in this analysis, which is the credit type currently available for dry AD facilities processing OFMSW, though there has been much contention surrounding the qualifying category of credits (Greene, 2017; Pleima, 2019). The regulatory floor and ceiling price for D5 RINs is $0.05 and $2.00 per RIN, respectively, and while prices have varied over the last 10 years between $0.15–1.15 per RIN, they have remained below $0.50 per RIN since late 2018 (U.S. EPA, 2019).

2.2.3. RNG Non-Vehicle (Pipeline Injection)

The RNG Non-Vehicle scenario assesses the economics of injecting RNG into the natural gas grid for generic, untracked usage. In this case, the facility must invest in a pipeline interconnection station and will receive revenues in line with wholesale natural gas prices, which are typically $0.003–0.006 per MJ; in 2017 the California average selling price was $0.003 per MJ (U.S. Energy Information Administration, 2019). There is currently no option for monetizing the environmental benefits of pipeline-injected, end-use-agnostic RNG at a state or federal level. In late 2018, the California legislature passed a bill authorizing the state utilities commission to develop a biomethane procurement program, which in theory will raise the market value of pipeline-injected biomethane (California Senate, 2018). However, the scope and timeline of this bill is unknown, and therefore we did not include any above-market revenues for biomethane in this scenario.

3. Results and discussion

Our results are reported on the basis of the levelized cost of disposal (LCOD), which can be thought of as analogous to the often-used levelized cost of energy metric (Ayres et al., 2004). LCOD represents the per-tonne tipping fee the facility would need to receive to achieve a net present value of zero over its lifetime (i.e., the facility earns a rate of return equal to the discount rate). Fig. 2 shows the key costs and drivers for an AD facility under the Electricity scenario. Costs are reported per tonne of wet waste delivered to the facility, levelized over the 25-year lifetime of the facility. The net of these costs therefore represents the LCOD (black line in Fig. 2). The dashed lines in Fig. 2 represent the upper and lower bounds of LCOD across cost and performance scenarios. Results demonstrate economies of scale, as the LCOD for the smallest facility analyzed here (25,000 tonnes of waste intake per year) is approximately $160 per tonne (range $110–223), more than 50% higher than the largest facility (300,000 tonnes of waste intake per year; LCOD $105 per tonne with range $60–160). These economies of scale primarily occur in the capital, labor, and O&M categories, and half of the potential economies of scale shown are realized at a facility size of 75,000 tonnes per year, with decreasing cost reductions at larger sizes. There is a tradeoff in transportation costs associated with sourcing waste from a larger area to achieve these economies of scale. However, the role that waste hauling costs plays in the LCOD is complicated by the fact that waste collection and hauling services may be provided by one or more separate entities (as is the case for the ZWEDC facility), each with their own contracts and cost structures.

Baseline costs for a 100,000 tonne-per-year AD facility under three energy production scenarios are shown in Fig. 3. LCOD is highest for the RNG Non-Vehicle scenario at $140 per tonne (range $102–195), due to the low value of the RNG as a replacement for natural gas, the absence of policy incentives, and the biogas upgrading costs incurred by the facility. Electricity-generating facilities have a lower LCOD than RNG Non-Vehicle facilities at $123 per tonne (range $78–181). The RNG Vehicle Fuel scenario has a baseline LCOD of $105 per tonne. The RNG Vehicle Fuel scenario's lower bound is estimated at $32 per tonne, significantly lower than the other scenarios, due largely to potential revenue from California state and U.S. Federal incentives for renewable transportation fuels (LCFS and RINs), although the upper end of the LCOD range is similar to other energy scenarios at around $178 per tonne.

In all cases, the largest cost on a per-tonne basis is the management of the digestate (26–29% of facility costs), which includes costs for third-party digestate transportation and management. Of the organic waste loaded into the digester, approximately 70% of the original mass remains in the solid digestate. Combined with the cost of disposing of inorganic residuals, (11–12% of costs), the final disposal of waste streams from the facility accounts for 40% of operating costs. This outweighs even the capital costs of building the digester (21–23% of costs) and upgrading the gas to methane in the RNG scenarios (10%). The role of solid waste management and disposal in facility economics is a motivator for recent efforts to develop alternative treatments for digestate, such as pyrolysis, which can produce additional energy (syngas), soil amendments (biochar), and/or bio-oil that can be recycled into the digester (Liu et al., 2020; Monlau et al., 2016).

Energy revenues vary across the three scenarios. In the RNG Non-Vehicle scenario, gas is sold at wholesale prices earning ~ $2–6 per tonne of waste. This revenue is insufficient to offset the cost of upgrading the biogas to biomethane (~$14 per tonne waste). At the low end of power output and feed-in tariffs, revenue from electricity generation is similarly low at $3 per tonne of waste (as modeled in the Low Performance-High Cost scenario). At the upper end of potential power generation and feed-in tariffs, revenue from electricity sales can reach $21 per tonne waste, while the Base Performance-Base Cost scenario yields approximately $10 per tonne waste. The comparative costs and revenues across energy utilization pathways are applicable beyond dry AD, to any biogas-generating facility including wastewater treatment plants and stand-alone wet AD, though metrics per tonne of waste will vary due to the impact of waste composition, moisture, residence time, and other operational parameters on biogas yields.

3.1. Competitiveness with landfills

Landfilling is the most prevalent OFMSW disposal alternative to AD, as compost facilities do not commonly handle OFMSW due to contamination levels and concerns about odors and pests (Cotton, 2019). Hence, the LCOD results in this study can be contextualized through a comparison with landfill tipping fees adjusted to represent a lifetime average comparable to LCOD (denoted as LCOD-equivalent; see SI for adjustment calculations). Tipping fees across California vary from $0-184 per tonne of waste (LCOD-equivalent); zero values arise when landfills are county-owned and paid for through non-tipping-fee revenue mechanisms such as property taxes (see Fig. S2) (CalRecycle, 2015). Landfill tipping fees vary across the United States, with average LCOD-equivalents exceeding $100 per tonne in small, land-constrained states in the Northeast (e.g. Rhode Island, Delaware), Hawaii, and Alaska (see Fig. S2) (Environmental Research & Education Foundation, 2018). Fig. 4 compares the California median landfill LCOD-equivalent of $66 per tonne and the maximum of $184 per tonne to modeled LCOD

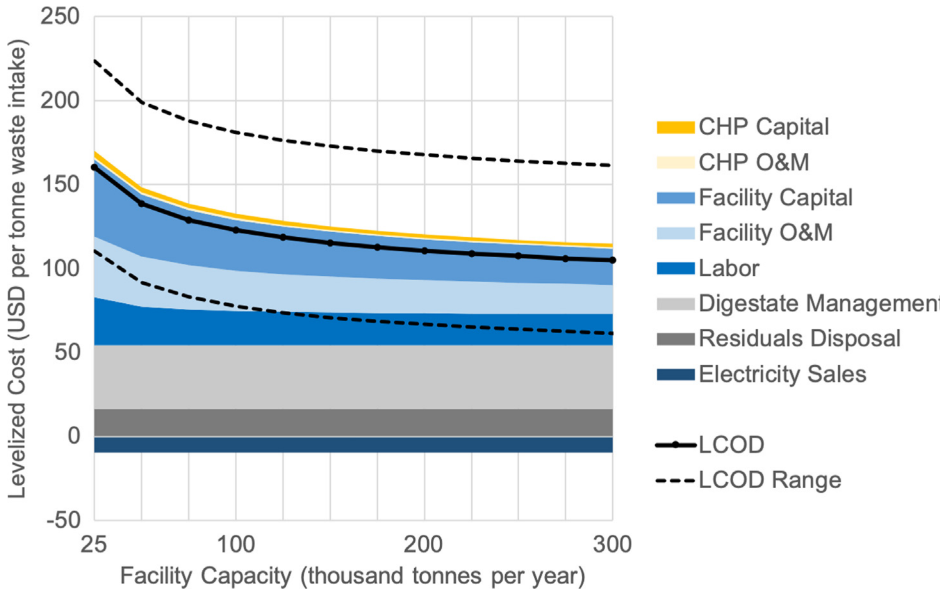

Fig. 2. Levelized costs and revenues for a facility over a range of sizes operating under the Electricity scenario. Lines show the net costs, or the levelized cost of disposal (LCOD). Cost and revenue areas and the solid LCOD line represent the Base Performance-Base Cost scenario, while dashed lines show the range of LCOD values under all performance and cost scenarios.

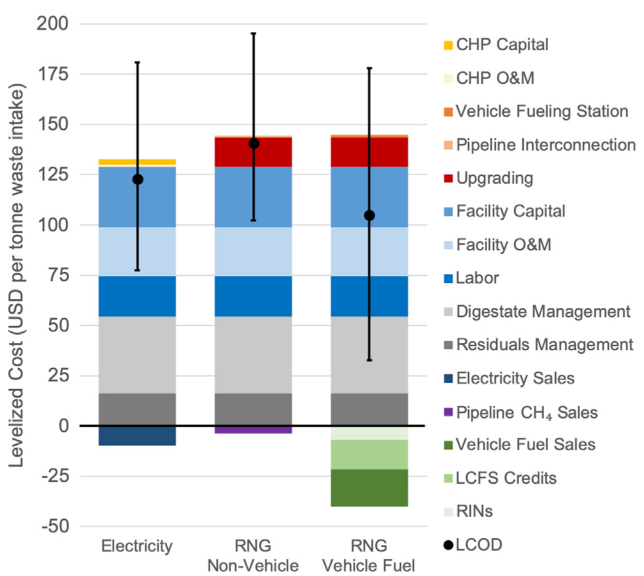

Fig. 3. Per-tonne costs and revenues by category for a 100,000 tonne-per-year facility under three energy production scenarios. The stacked bars and dots represent the costs and revenues for the Base Performance-Base Cost scenario, while the error bars represent the range of LCOD values across all performance and cost scenario combinations.

for dry AD. A facility with LCOD in the orange zone of Fig. 4 would not be competitive with any landfill in California, while the yellow zone represents competitiveness with the more expensive landfills in the state and the green zone corresponds to competitive costs relative to a majority of state landfills. Compared to a $66 per tonne LCOD-equivalent, the RNG Vehicle Fuel and Electricity scenarios could be competitive if facilities achieve lower-than-expected costs, but the scale required to make the Electricity scenario competitive (225,000 tonnes per year) is beyond the largest dry AD facilities currently in operation. If compared to $184 per tonne, which is the highest tipping fee in California and near the upper end in the U.S., AD facilities are far more likely to be competitive. Specifically, at this $184 per tonne landfill tipping fee equivalent, assuming waste intake greater than 100,000 tonnes per year ensures that the Electricity and RNG Vehicle Fuel scenarios will be economically attractive. In more populated areas, reaching this scale will be possible (even if waste is hauled no more than 20 km), while smaller cities and rural areas will need to aggregate their wastes at centralized facilities (Scown et al., 2019).

Although outside of the scope of this analysis, it is worth noting that the source-separation and/or pre-processing required to reduce contamination to levels acceptable for dry AD can result in additional costs. These costs may be in the form of separate bins and collection routes to facilitate increased source-separation or in processing at materials recovery facilities that are designed for high organics recovery (e.g., using de-packaging machines). Some of the costs and benefits of diverting organic waste from landfills are not monetized in this analysis, including the conservation of existing landfill capacity, reduction in landfill biogas generation, improvement in leachate quality, potential recovery of organic matter and nutrients, and changes in GHG and other air pollutant emissions (Chen et al., 2021; Morris et al., 2013; Nordahl et al., 2020). However, estimated social costs of landfilling wet organic waste based on the impacts of air pollutant and greenhouse gas emissions are $25–40 per tonne according to results from a recent life-cycle assessment of dry AD (Nordahl et al., 2020). If these costs were incorporated into landfill tipping fees, it would increase the relevant point of comparison and make AD facilities competitive at even more sizes and configurations.

3.2. Uncertainty

The LCOD can vary by more than $60 per tonne within the range of cost and performance scenarios we analyzed (see Fig. 5). Unit costs drive the majority of the uncertainty, while the variations in facility performance do not have as large an impact on LCOD. In all cases, the cost of managing inorganic residuals and solid digestate is the largest source of uncertainty, accounting for a -$25 to $42 per tonne variation in LCOD. If residuals and digestate costs were held constant at the Base Cost values, the overall LCOD uncertainty range for 100,000 tonne-per-year facilities would

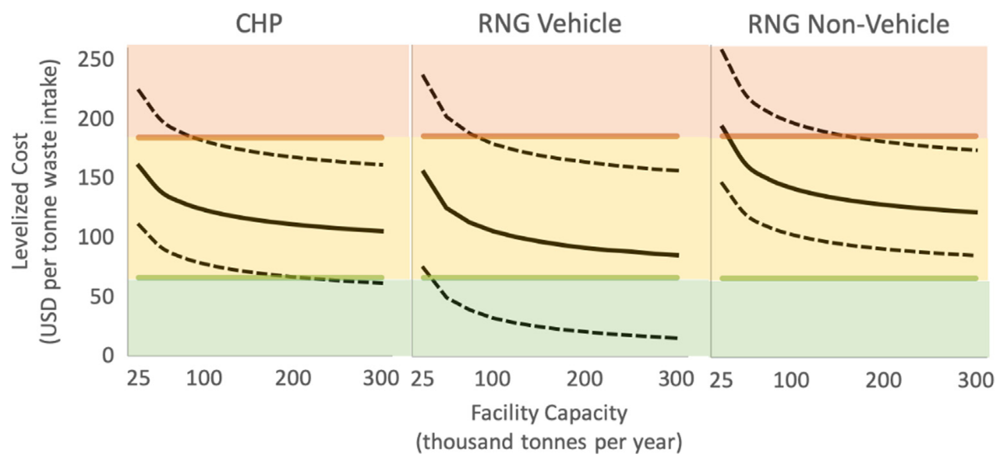

Fig. 4. Economic competitiveness of AD facilities with landfills across a range of capacities (x-axis) and energy scenarios (panels). LCOD of AD for the base case (solid black line) and uncertainty range (dashed lines) are compared to the median (green line) and maximum (orange line) LCOD-equivalent landfill tipping fee in California. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

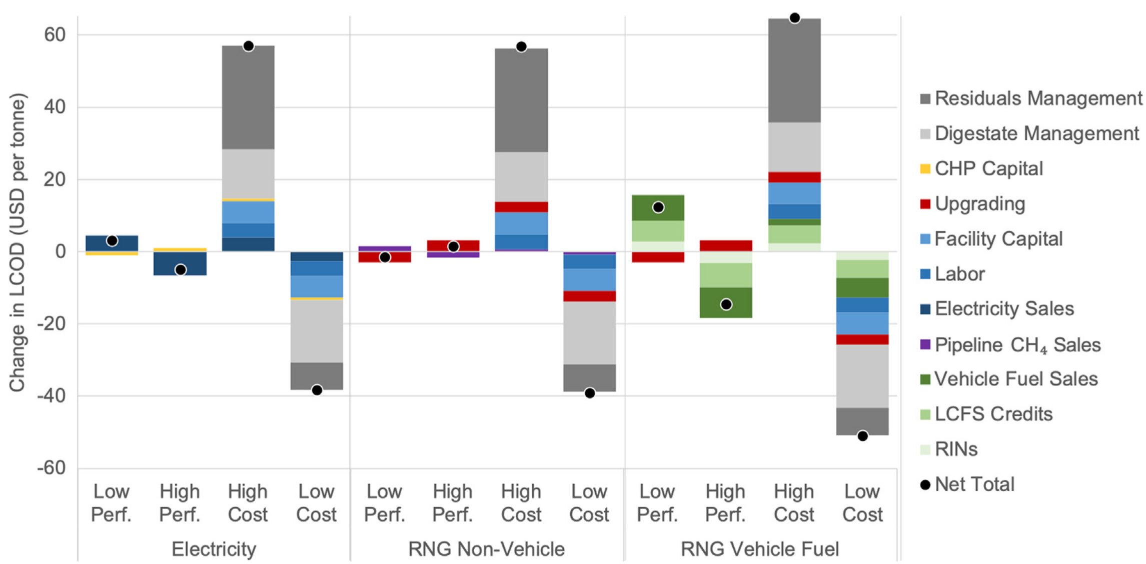

Fig. 5. Variation in LCOD caused by individual factors for a 100,000 tonne-per-year facility under various scenarios as compared to the Base Performance-Base Cost scenario for the given fuel pathway. The Low and High Performance (Perf.) scenarios retain the Base Cost assumptions, while the Low and High Cost scenarios retain the Base Performance assumptions. Results for combinations of these scenarios (e.g. High Performance-High Cost) would be additive, but not linearly so. Cost categories with less than a 1% impact on LCOD across all scenarios have been removed from the figure for clarity.

shrink by 65%, 73%, and 46% for the Electricity, RNG Non-Vehicle, and RNG Vehicle Fuel scenarios, respectively (see Fig. S3 for comparison). In the High Performance scenarios, costs increase due to the need to process (i.e., combust or upgrade) additional biogas. In the Electricity and RNG Vehicle scenarios, the revenues from increased energy production outweigh this cost, but in the RNG Non-Vehicle-High Performance scenario the additional market gas revenues are smaller than the additional gas upgrading costs, resulting in a net increase in LCOD.

Although a wide range of costs are covered in our uncertainty analysis, facilities outside of California may have costs outside of that range, particularly labor costs, which in much of the U.S. would likely be lower than our modeled range. (Bureau of Labor Statistics, 2019). Digestate management costs may also be lower in areas with inexpensive land, where odor is not a concern, and with different regulations governing year-round composting or land application. It should also be noted that, while the LCFS is specific to California (a number of other states do have variations of this policy in place), facilities outside of California are eligible to earn LCFS credits if they sell into the California market.

3.3. Energy prices and policy impacts

The wholesale market value of energy outputs from AD facilities, absent any policy incentives, is a minor source of revenue relative to what is required to offset facility costs, as highlighted in the RNG Non-Vehicle scenario where gas pipeline revenues offset 1–7% of facility costs. If the facility can earn retail energy prices instead of wholesale prices, as is the case for the RNG Vehicle Fuel scenario, these energy revenues can offset 4–37% of costs. However, participation in the vehicle fuel market is predicated upon the ability to find reliable customers, ideally a fleet of medium- or heavy-duty vehicles, which may be difficult for facility owners who do not own related businesses such as a waste hauling fleet, or facility locations that are not close to a major road network. Adjacent investment decisions such as whether to convert a trucking fleet from diesel to CNG are outside the scope of the model, but it is worthwhile to note that the availability of CNG customers can factor into a facility's decision to pursue this option. In the Electricity scenario, policies such as renewable feed-in tariffs or power purchase agreements valued above wholesale prices can increase energy revenues, though the total value relative to the costs is at most 25%. The combination of the facility's ability to capture retail revenues paired with substantial state and federal incentive programs make the RNG Vehicle Fuel scenario the lowest-cost option from an LCOD perspective. However, these increased revenues are uncertain, varying by a factor of 5 across the cost scenarios modeled, because fuel credits trade on an open market and therefore future prices will fluctuate.

The less money a facility earns through energy sales and energy-related incentives, the more they must earn through tipping fees in order to be financially viable. The relative importance of these two revenue streams (waste intake vs. energy output) may impact the way the facility is built and operated. Fig. 6 shows the share of total revenues that comes from energy sales and energy-related incentives in each scenario assuming the facility earns a tipping fee required to break-even (i.e., commensurate with the LCOD). Energy is responsible for at most 7% of revenues in the RNG Non-Vehicle scenario, while the Electricity scenario earns 15–25% of revenues from energy at the high end, depending on facility size. In these cases, the facility would be less motivated to invest in improving energy generation processes such as optimizing gas yield or reducing flaring, as their money would primarily come from waste intake. Conversely, if energy is the dominant revenue source (up to 83% in the RNG Vehicle Fuel scenario), the facility may be motivated to maximize energy output and become selective in the waste they accept in an effort to generate as much gas as possible. This could limit the diversion opportunities for more contaminated or difficult-to-handle streams such as mixed municipal solid waste. Energy and waste policy planners should carefully consider the prices being offered from various sources and what it will mean for both waste disposal costs as well as drivers for facility operation and investment. Additionally, regulations on the way facilities operate could be put in place (e.g., minimum retention times, best practices for percolate circulation) and acceptable waste streams in order to ensure that facilities being supported by public policies and money are operating in a way that maximizes their social benefits.

New energy-related value streams could also be considered to ensure that AD facilities are incentivized to maximize their energy and emissions benefits. For example, electricity-generating facilities offer a dispatchable form of renewable energy and therefore could be incentivized to follow specific dispatch schedules that help meet peak demand and ramping needs of the grid. The current BioMAT program in California accomplishes this to a limited extent through time-of-day modifying factors. The same grid benefits could be achieved by injecting RNG into natural gas pipelines for use at off-site generation facilities, but this scenario is currently the least economically viable. New monetization mechanisms for RNG used in non-vehicle fuel purposes could open up opportunities to reduce the carbon footprint of existing natural gas power plants and decarbonize industrial processes that are too difficult or expensive to electrify, either through co-location with an AD facility for direct biogas combustion and/or heat recovery from CHP units, or through direct sales via the natural gas grid. These added revenues would also lower the LCOD of AD facilities and make them economically viable at lower tipping fees.

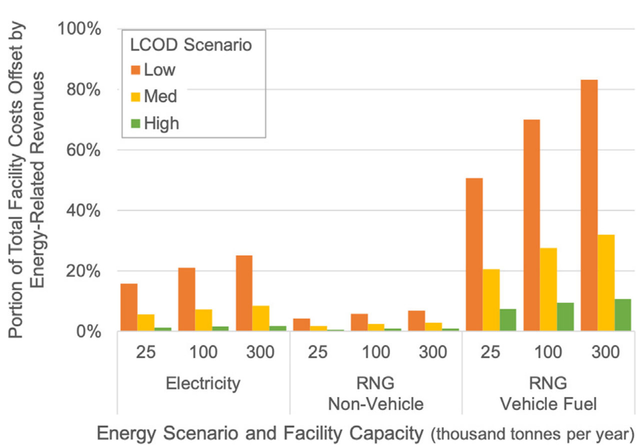

Fig. 6. Share of revenues coming from energy sales and incentives under a breakeven scenario, where the facility is earning a tipping fee equal to the levelized cost of disposal (LCOD), for three energy scenarios and three facility sizes over the range of LCOD uncertainty (low, medium, high).

4. Conclusions

Dry AD facilities are a financially viable approach to help meet landfill diversion and renewable energy goals, though organic and inorganic waste management costs are significant and uncertain (+/- 30–70% of baseline LCOD depending on scenario). Depending on alternative disposal options and local landfill tipping fees, dry AD may be cost-competitive under existing policies and energy prices or may require additional support through guaranteed higher tipping fees, subsidized materials handling costs, or new and increased energy-related monetary incentives. Each of these strategies incentivize different investments and behavior by owners and operators, and should, therefore, be considered carefully.

Economies of scale are important to the overall LCOD. The largest facilities modeled (i.e., 300,000 tonnes per year) generally had LCOD values that were 55–70% those of the smallest facilities modeled (i.e., 25,000 tonnes per year). However, larger facilities may face barriers not explicitly considered in this study such as difficulty obtaining feedstocks from local sources, odor and emissions management issues, and resistance from neighboring communities. Materials handling costs, namely the disposal of inorganic residuals that come into the facility and the management of post-AD digestate by third-party composters, vary considerably and have the potential to be well over half of the total per-tonne costs incurred by a facility. However, some of the uncertainty can be mitigated through materials contracts and holistic waste management policy support.

RNG for use as a vehicle fuel is currently the most lucrative energy utilization pathway for AD facilities due to existing U.S. Federal and state policy incentives. Upgrading the biogas to RNG for non-vehicle fuel uses is not economically attractive, as the upgrading costs alone outweigh wholesale natural gas revenues and no economic incentives currently in place to support the production of RNG for non-vehicle applications. Lastly, electricity generation is an economically viable pathway in cases where the alternative landfill tipping fees are high, and may be attractive for facilities that do not operate truck fleets capable of utilizing RNG and cannot easily connect to natural gas pipelines. Advanced electricity dispatch strategies and monetization of thermal energy from combined heat and power units could help make this scenario more attractive.

Limitations of the study include significant focus on the physical, financial, and operational characteristics of a specific facility in California and use of deterministic inputs as opposed to an operational framework that determines facility size, operations and costs to achieve a specific financial or operational objective (e.g., least cost, specific return on investment). Dry AD technology is still quite nascent in the U.S. and we have limited empirical data to draw from for inputs and assumptions. While this study used bounding assumptions to represent uncertainty, public data on dry AD facility financial, operating, and production characteristics would generate more precise results. Opportunities for future technoeconomic analysis of dry AD facilities include additional consideration of byproduct management pathways such as novel composting methods, land application of raw material, or pyrolysis of digestate, as well as any associated revenues from the sale of finished compost or biochar (a byproduct of pyrolysis). Future studies could also quantify the benefits of various levels of non-vehicle RNG- and thermal energy recovery-related economic incentives and opportunities for co-location with other industrial facilities that can utilize a range of energy byproducts. As more dry AD facilities are built and more data becomes available, research should further explore the impact of facility design parameters such as digester residence times, gas storage capacity, and feedstock composition on the costs and benefits, as well as the societal impacts, of dry AD.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Acknowledgements

The research for this paper was financially supported by the California Energy Commission under agreement number EPC-14-044 and EPC-14-030. We thank Greg Ryan, John Pena, Amelin Norzamini, Osvaldo Cordero, and Prab Sethi for their input. This work was also part of the DOE Joint BioEnergy Institute (http://www.jbei.org) supported by the U.S. Department of Energy, Office of Science, Office of Biological and Environmental Research, through contract DE-AC02-05CH11231 with Lawrence Berkeley National Laboratory. This work was supported by the National Science Foundation under Grant No. 1739676. Any opinions, findings, and conclusions or recommendations expressed in this material are those of the authors and do not necessarily reflect the views of the sponsors.

Appendix A. Supplementary material

Supplementary data to this article can be found online at https://doi.org/10.1016/j.wasman.2021.04.048.

References

AgSTAR, 2018. Livestock Anaerobic Digester Database [WWW Document]. URL https://www.epa.gov/agstar/livestock-anaerobic-digester-database (accessed 7.26.19).

Angelonidi, E., Smith, S.R., 2015. A comparison of wet and dry anaerobic digestion processes for the treatment of municipal solid waste and food waste. Water Environ. J. 29, 549–557. https://doi.org/10.1111/wej.12130.

Ascher, S., Watson, I., Wang, X., You, S., 2019. Township-based bioenergy systems for distributed energy supply and efficient household waste re-utilisation: Techno-economic and environmental feasibility. Energy 181, 455–467. https://doi.org/10.1016/j.energy.2019.05.191.

Ayres, M., MacRae, M., Stogran, M., 2004. Levelised Unit Electricity Cost Comparison of Alternate Technologies for Baseload Generation in Ontario (Prepared for the Canadian Nuclear Association). Canadian Energy Research Institute.

Barlaz, M.A., Chanton, J.P., Green, R.B., 2009. Controls on landfill gas collection efficiency: instantaneous and lifetime performance. J Air Waste Manag Assoc 59, 1399–1404. https://doi.org/10.3155/1047-3289.59.12.1399.

Breunig, H.M., Amirebrahimi, J., Smith, S., Scown, C.D., 2019. Role of Digestate and Biochar in Carbon-Negative Bioenergy. Environ. Sci. Technol. 53, 12989–12998. https://doi.org/10.1021/acs.est.9b03763.

Bureau of Labor Statistics, 2019. Employer Costs for Employee Compensation for the Regions [WWW Document]. URL https://www.bls.gov/regions/southwest/news-release/employercostsforemployeecompensation_regions.htm (accessed 10.10.19).

California Air Resources Board, 2018. Low Carbon Fuel Standard [WWW Document]. URL https://www.arb.ca.gov/fuels/lcfs/lcfs.htm (accessed 8.15.19).

California Air Resources Board, 2019a. LCFS Pathway Certified Carbon Intensities [WWW Document]. URL https://ww3.arb.ca.gov/fuels/lcfs/fuelpathways/pathwaytable.htm (accessed 8.15.19).

California Air Resources Board, 2019b. LCFS Credit Value Calculator [WWW Document]. URL https://ww3.arb.ca.gov/fuels/lcfs/dashboard/dashboard.htm (accessed 8.13.20).

California Air Resources Board, 2020. Weekly LCFS Credit Trading Activity Reports [WWW Document]. URL https://ww3.arb.ca.gov/fuels/lcfs/credit/lrtweeklycreditreports.htm (accessed 8.12.20).

California Senate, 2018. Senate Bill No. 1440 [WWW Document]. URL http://leginfo.legislature.ca.gov/faces/billNavClient.xhtml?bill_id=201720180SB1440 (accessed 9.3.20).

CalRecycle, 2015. Landfill Tipping Fees in California (No. #DRRR-2015-1520). CalRecycle.

Chen, T., Qiu, X., Feng, H., Yin, J., Shen, D., 2021. Solid digestate disposal strategies to reduce the environmental impact and energy consumption of food waste-based biogas systems. Bioresour. Technol. 325,. https://doi.org/10.1016/j.biortech.2021.124706 124706.

CMSA, 2018. Comprehensive Annual Financial Report for the Fiscal Year 2017–2018. Central Marin Sanitation Agency.

Cotton, M., 2019. SB 1383 Infrastructure and Market Analysis (Contractor's Report No. DRRR-2019-1652). CalRecycle.

CPUC, 2018. Bioenergy Market Adjusting Tariff (BioMAT) Program Review and Staff Proposal. California Public Utilities Commission.

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, 180–184. https://doi.org/10.1038/s41586-019-1720-3.

Environmental Research & Education Foundation, 2018. Analysis of MSW Landfill Tipping Fees, April 2018 (Rev. ed.). www.erefdn.org.

Greene, P., 2017. 101 For RINs. Biocycle.

Hake, J., Zipkin, J., Grow, P., 2017. Key Factors to Enable the Anaerobic Digestion of Food Waste at WWTPs.

Jordan, P., Krause, M.J., Chickering, G., Carson, D., Tolaymat, T., 2020. Impact of Food Waste Diversion on Landfill Emissions.

Khan, M.M.-U.-H., Jain, S., Vaezi, M., Kumar, A., 2016. Development of a decision model for the techno-economic assessment of municipal solid waste utilization pathways. Waste Manag. 48, 548–564. https://doi.org/10.1016/j.wasman.2015.10.016.

Linville, J.L., Shen, Y., Wu, M.M., Urgun-Demirtas, M., 2015. Current state of anaerobic digestion of organic wastes in North America. Curr. Sustain. Renew. Energy Rep. 2, 136–144. https://doi.org/10.1007/s40518-015-0039-4.

Liu, J., Huang, S., Chen, K., Wang, T., Mei, M., Li, J., 2020. Preparation of biochar from food waste digestate: Pyrolysis behavior and product properties. Bioresour. Technol. 302,. https://doi.org/10.1016/j.biortech.2020.122841 122841.

Monlau, F., Francavilla, M., Sambusiti, C., Antoniou, N., Solhy, A., Libutti, A., Zabaniotou, A., Barakat, A., Monteleone, M., 2016. Toward a functional integration of anaerobic digestion and pyrolysis for a sustainable resource management. Comparison between solid-digestate and its derived pyrochar as soil amendment. Appl. Energy 169, 652–662. https://doi.org/10.1016/j.apenergy.2016.02.084.

Morris, J., Scott Matthews, H., Morawski, C., 2013. Review and meta-analysis of 82 studies on end-of-life management methods for source separated organics. Waste Manag. 33, 545–551. https://doi.org/10.1016/j.wasman.2012.08.004.

Nordahl, S.L., Devkota, J.P., Amirebrahimi, J., Smith, S.J., Breunig, H.M., Preble, C.V., Satchwell, A.J., Jin, L., Brown, N.J., Kirchstetter, T.W., Scown, C.D., 2020. Life-Cycle Greenhouse Gas Emissions and Human Health Trade-Offs of Organic Waste Management Strategies. Environ. Sci. Technol. 54, 9200–9209. https://doi.org/10.1021/acs.est.0c00364.

Ong, M.D., Williams, R.B., Kaffka, S.R., 2014. Comparative Assessment of Technology Options for Biogas Clean-Up (Contractor Report No. CEC-500-2017-007-APH). California Energy Commission.

Pleima, B., 2019. Biogas To RNG Projects: What. Why And How. Biocycle 60, 38.

Preble, C.V., Chen, S.S., Hotchi, T., Sohn, M.D., Maddalena, R.L., Russell, M.L., Brown, N.J., Scown, C.D., Kirchstetter, T.W., 2020. Air pollutant emission rates for dry anaerobic digestion and composting of organic municipal solid waste. Environ. Sci. Technol. 54, 16097–16107. https://doi.org/10.1021/acs.est.0c03953.

Rajendran, K., Kankanala, H.R., Martinsson, R., Taherzadeh, M.J., 2014. Uncertainty over techno-economic potentials of biogas from municipal solid waste (MSW): A case study on an industrial process. Appl. Energy 125, 84–92. https://doi.org/10.1016/j.apenergy.2014.03.041.

Rajendran, K., Murthy, G.S., 2019. Techno-economic and life cycle assessments of anaerobic digestion – A review. Biocatal. Agric. Biotechnol. 20,. https://doi.org/10.1016/j.bcab.2019.101207 101207.

Rocamora, I., Wagland, S.T., Villa, R., Simpson, E.W., Fernández, O., Bajón-Fernández, Y., 2020. Dry anaerobic digestion of organic waste: A review of operational parameters and their impact on process performance. Bioresour. Technol. 299,. https://doi.org/10.1016/j.biortech.2019.122681 122681.

Sanscartier, D., Maclean, H.L., Saville, B., 2012. Electricity production from anaerobic digestion of household organic waste in Ontario: techno-economic and GHG emission analyses. Environ. Sci. Technol. 46, 1233–1242. https://doi.org/10.1021/es2016268.
Satchwell, A.J., Scown, C.D., Smith, S.J., Amirebrahimi, J., Jin, L., Kirchstetter, T.W., Brown, N.J., Preble, C.V., 2018. Accelerating the deployment of anaerobic digestion to meet zero waste goals. Environ. Sci. Technol. 52, 13663–13669. https://doi.org/10.1021/acs.est.8b04481.
Schwarzenegger, A., 2007. Executive Order S-01-07.
Scown, C., Breunig, H., Kavvada, O., Huntington, T., 2019. Biositing Webtool, v1. Lawrence Berkeley National Laboratory (LBNL), Berkeley, CA (United States). doi:10.11578/dc.20191029.4
Tominac, P., Aguirre-Villegas, H., Sanford, J., Larson, R., Zavala, V., 2021. Evaluating landfill diversion strategies for municipal organic waste management using environmental and economic factors. ACS Sustain. Chem. Eng. 9, 489–498. https://doi.org/10.1021/acssuschemeng.0c07784.
U.S. Energy Information Administration, 2019. California Natural Gas Prices [WWW Document]. URL https://www.eia.gov/dnav/ng/ng_pri_sum_dcu_SCA_a.htm (accessed 8.15.19).
U.S. EPA, 2018a. Inventory of U.S. Greenhouse Gas Emissions and Sinks 1990-2016 (No. EPA 430-R-18-003). United States Environmental Protection Agency.
U.S. EPA, 2018b. Types of Anaerobic Digesters [WWW Document]. URL https://www.epa.gov/anaerobic-digestion/types-anaerobic-digesters (accessed 7.25.19).
U.S. EPA, 2019. RIN Trades and Price Information [WWW Document]. URL https://www.epa.gov/fuels-registration-reporting-and-compliance-help/rin-trades-and-price-information (accessed 8.14.19).
U.S. EPA, 2020. Landfill Methane Outreach Program: Landfill Technical Data [WWW Document]. URL https://www.epa.gov/lmop/landfill-technical-data (accessed 7.22.20).