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Environmental Science & Technology
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Policy Analysis

Life-Cycle Greenhouse Gas Emissions and Human Health Trade-Offs of Organic Waste Management Strategies

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Cite This: *Environ. Sci. Technol.* 2020, 54, 9200−9209

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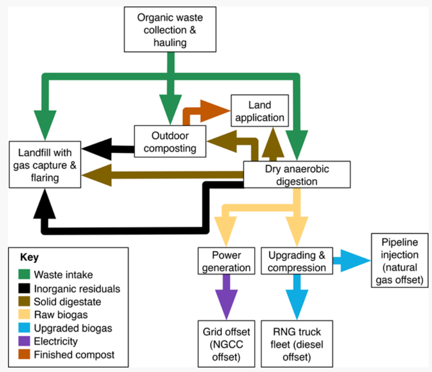
ABSTRACT: Waste-to-energy systems can play an important role in diverting organic waste from landfills. However, real-world waste management can differ from idealized practices, and emissions driven by microbial communities and complex chemical processes are poorly understood. This study presents a comprehensive life-cycle assessment, using reported and measured data, of competing management alternatives for organic municipal solid waste including landfilling, composting, dry anaerobic digestion (AD) for the production of renewable natural gas (RNG), and dry AD with electricity generation. Landfilling is the most greenhouse gas (GHG)-intensive option, emitting nearly 400 kg CO_{2e} per tonne of organic waste. Composting raw organics resulted in the lowest GHG emissions, at −41 kg CO_{2e} per tonne of waste, while upgrading biogas to RNG after dry AD resulted in −36 to −2 kg CO_{2e} per tonne. Monetizing the results based on social costs of carbon and other air pollutant emissions highlights the importance of ground-level NH_3 emissions from composting nitrogen-rich organic waste or post-AD solids. However, better characterization of material-specific NH_3 emissions from landfills and land-application of digestate is essential to fully understand the trade-offs between alternatives.



■ INTRODUCTION

Local and state governments are pursuing ambitious “zero waste” policies with the goal of reducing methane emissions to the atmosphere and minimizing the quantity of waste sent to landfills. For example, California’s strategy for reducing short-lived climate pollutant emissions (Senate Bill 1383) includes a goal to reduce the fraction of organic waste sent to landfills by 75% in 2025 relative to 2014 levels.[1] The highest-emitting point sources of methane in California are a subset of the state’s landfills.[2] Dedicated facilities capable of processing mixed solid organic waste streams will be critical to meeting ambitious organics diversion and renewable energy goals.[3,4] Organic waste anaerobic digestion projects can also earn valuable credits for producing low-carbon fuel when biogas is sold for use in transportation applications. As of 2019, the only net negative carbon-intensity fuel pathways approved as part of the California Low Carbon Fuel Standard (LCFS) are based on landfill gas utilization, anaerobic digestion (AD) of manure, and AD of mixed organic solid waste.[5] Previous literature, as reviewed by Morris et al.,[6] overwhelmingly agrees that the GHG footprint of landfilling organic waste is higher relative to composting or waste-to-energy by as much as a factor of 9, even when landfill gas is captured and utilized.[7,8] However, there is less consensus around the GHG footprints of specific waste-to-energy and composting options, and limited research is available on non-GHG emissions.

Cities hoping to reduce landfill disposal of organic waste must weigh a complex set of competing options across a range of environmental metrics including GHG emissions, air quality and human health burdens, public nuisances such as odor impacts, and environmental justice implications. In this study, we conduct a rigorous life-cycle assessment (LCA) that integrates the best available estimates across the scientific literature and newly collected empirical data to explore the climate and human health trade-offs between landfilling, composting, and dry AD of mixed municipal organic waste. Our choice to focus on dry AD (solids loading 22−40% versus <16% for wet AD[9]) for waste-to-energy stems from its usefulness in processing solid organic waste streams, particularly those with appreciable inorganic contamination, in dedicated facilities and its potential to reduce costs.[10−13] This study also explores variations in the management of solid digestate (residual solids remaining after AD), including landfilling, raw digestate application to land, and composting, including estimated net GHG impacts and fertilizer offset

Received: January 30, 2020
Revised: July 2, 2020
Accepted: July 6, 2020
Published: July 6, 2020

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credits after the material is applied to working lands. By establishing a system boundary that extends from waste collection through application of residual solids/compost to soils, this study provides a comprehensive analysis of life-cycle GHG emissions (carbon dioxide (CO_2), methane (CH_4), nitrous oxide (N_2O)), air pollutant emissions (nitrogen oxides (NO_x), non-methane volatile organic compounds (NMVOC), sulfur dioxide (SO_2), carbon monoxide (CO), ammonia (NH_3), and particulate matter ($PM_{2.5}$)) and monetized climate and human health damages associated with organic waste management options.

METHODS AND DATA

Clearly defined and sufficiently expansive system boundaries are essential to understanding the trade-offs between different organic waste management and utilization strategies, along with input data that is as robust and representative as possible. Trucking distances, landfill emissions, composting emissions, and net emissions after land application are all closely tied to the specifics of a location, waste composition, and detailed management strategy. Attempting to quantify a broadly applicable set of average values is of limited usefulness. We have chosen to begin with an existing set of operations in San Jose, CA. Specific mass and energy balances, emission rates, and transportation distances are tied to a dry AD facility built and operated by Zero Waste Energy Development Company (ZWEDC), referred to simply as ZWEDC in the following sections (see Figure S2 for an aerial photo). In addition to the ZWEDC case, we evaluate alternative management options for the same material as variations on this scenario (see Figure 1).

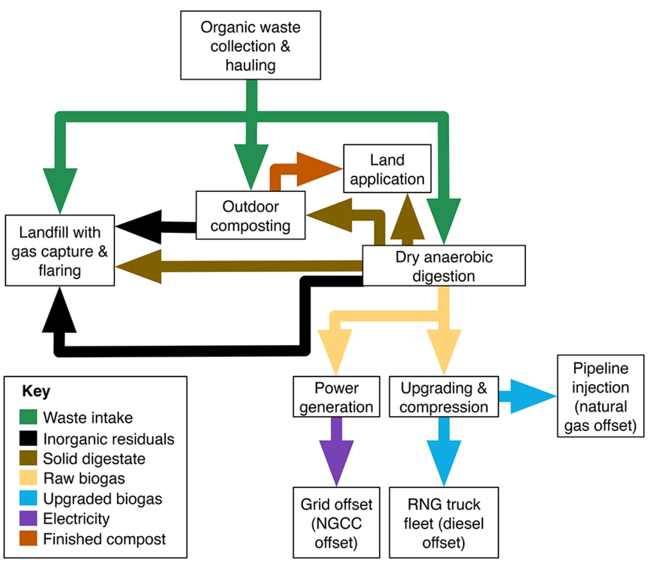

Figure 1. System boundary for life-cycle assessment.

In the existing ZWEDC operations, mixed municipal organic waste (largely dominated by food waste and food-soiled paper) is sent to a dry AD facility, and raw biogas is combusted to generate electricity for on-site use and export to the grid. The solid digestate is sent to a composting facility before it is ultimately applied to land as finished compost (Figure S7 shows detailed ZWEDC operations). For the purposes of this study, we will refer to the ZWEDC waste stream as "mixed organics", which are approximated as food waste. The additional hypothetical alternatives include the following: landfilling all mixed organics, composting all mixed organics, variations on the ZWEDC configuration in which digestate is either directly land-applied or landfilled, dry AD with biogas upgrading for pipeline injection to offset natural gas, and dry AD with biogas upgrading to fuel an otherwise diesel-powered truck fleet. Key details of these scenarios are discussed in following subsections.

To compare these scenarios on a common basis, we express all emissions in terms of one wet tonne of mixed organic waste processed. The question we seek to answer is given a unit mass of organic waste, what management strategy results in the most favorable net GHG and human health impacts? The results are dependent on the waste composition, and for this analysis, the mixed commercial organics processed at ZWEDC are approximated as food waste. Visual inspection at ZWEDC indicated that the organics received by ZWEDC are, in large part, food and food-soiled paper products (Figure S1), although the exact composition varies day-to-day and is not characterized on a regular basis. For the landfilling and composting scenarios, as well as for hypothetical variations on ZWEDC operations such as biogas upgrading to RNG, the best-available literature and industry values form the basis of our analysis. We expect these results to be generalizable in the U.S. national and international context for similar waste mixtures and technologies, with the exception of possible variations in composting and land application emissions. Landfill emissions will also be higher in states and countries that do not tightly regulate fugitive emissions.

Landfilling Organic Waste. The most common basis for comparison in organic waste management is landfilling. This reflects "business as usual" practices for 76% of food waste and other collected MSW organics across the U.S.[14] In the specific ZWEDC case, waste would be transported for disposal at the Newby Island Landfill. Large commercial waste streams (e.g., grocery stores and company cafeterias) are hauled directly, whereas municipal streams are sent first to processing facilities for initial sorting. In places like California where there is a marketable need for isolated organic waste streams, sorting/processing facilities may conduct sorting for organics in addition to plastic or paper sorting for recycling. Emissions sources in this scenario include diesel trucks hauling waste from commercial facilities and waste sorting/processing facilities to the landfill, fugitive emissions from waste decomposition in the landfill not captured by the gas capture system, and emissions from the landfill gas flare. We account only for emissions that occur within 100 years of disposal. As mentioned above, we approximate digested organics at ZWEDC as food waste. Fugitive landfill gas emissions are based on food waste-specific data in the literature.[15,16] Food waste decays relatively quickly and to some extent before individual landfill cells can be capped and connected to the gas capture system, so a significant proportion of total methane emitted over their lifetime is emitted to the atmosphere.[15] The emissions of NO_x, NH_3, SO_2, CO, NMVOCs, and $PM_{2.5}$ from the landfill operation and flaring are estimated using data from Ecoinvent (Table S1). We do not account for landfills' potential to sequester biogenic carbon, because the global warming potential offset is fairly small compared to methane emitted and uncertainty surrounding the fugitive methane emissions and related sequestration offset is captured in our sensitivity analysis.[15]

Composting Organic Waste. The most conventional alternative to landfilling is composting of raw organic waste. In the U.S., 61% of yard trimmings are composted, and only 5% of mixed organics/food waste is composted.[14] Composting can be a useful alternative for diverting either raw organic waste or further processing solid digestate to make it more suitable for land application. That said, even well-managed compost can release NH_3, N_2O, CH_4, SO_2, CO, and odor. These emissions are not well-studied across a range of starting materials, management techniques, and local climates.[17−20] In the raw organics composting scenario, we model direct transportation of all raw organic waste to the Z-Best composting facility near the City of Gilroy (approximately 70 km from ZWEDC), which is an outdoor composting operation capable of handling up to 1,200 tonnes of organic waste per day. This longer driving distance will likely be representative of large-scale composting options for cities across the U.S., given odor and emissions concerns associated with such operations. In the scenario where all organic waste is shipped directly to Z-Best for composting rather than ZWEDC for digestion, we assume it is bagged and composted for 14 weeks as per typical practice at Z-Best. We assume finished compost is applied to cropland as a soil amendment and partial fertilizer replacement.[21−23] This compost ultimately displaces the need for urea fertilizer (46% nitrogen by mass), and the offset credit is calculated on the basis of nitrogen, using an assumption of 1.7% nitrogen content in the compost.[24,25] The life-cycle of urea production is modeled assuming electricity, transportation (truck and rail), and natural gas production in the United States (see Table S2).

Dry Anaerobic Digestion of Organics with On-Site Electricity Generation. Dry AD for conversion of solid organic waste to biogas is understudied relative to wet AD, whereas the life-cycle impacts of wet AD systems used to process the municipal organic waste, manure, and biosolids have been explored in numerous papers.[26−30] To populate our model, we were able to obtain operating data over multiple years from the ZWEDC dry AD facility in San Jose, CA. Detailed operations are laid out in Figure S7. The facility is designed to accept approximately 81,650 tonnes (90,000 short tons) of waste annually. Waste intake at ZWEDC is dominated by mixed organics including food and food-soiled paper products, often accompanied by a substantial quantity of inorganic contamination that must be separated and landfilled. Our model relies on delivery logs that include the origin of each truckload of waste, some of which is delivered from waste sorting/processing facilities while other loads are hauled directly from commercial sources including grocery stores and office parks. Assumptions for the origins and driving distances of inbound waste, based on these logs, are described in the SI.

At the ZWEDC facility, sorted organic waste is dewatered using an extruder and loaded into one of 16 digester bays for a typical residence time of 21 days. Produced biogas is first sent to storage bladders located on the facility roof, which provide storage for a few hours' worth of biogas production. In overpressure events, raw biogas can be vented from these bladders. Stored biogas is then treated to reduce H_2S concentrations using an iron sponge and fed to an on-site combined heat and power (CHP) facility comprised of two 800 kW generators, for a combined nameplate capacity of 1.6 MW. Approximately 30% of the biogas is flared due to gas storage limitations as well as the nature of batch digestion, which produces low-methane content (referred to as lean) gas at the start and end of each cycle that cannot be sent to CHP units (see Figure S7). Daily electricity consumption at ZWEDC averages 3,700 kWh/day (translating to an average load of 156 kW), including operation of the extruder, lighting, and fans. We assume net electricity exports offset generation from natural gas combined cycle (NGCC) power plants which often satisfy the marginal demand on California's grid.[31] The solid digestate generated at ZWEDC (4,040 tonnes per month on average, as shown in Figure S4) is aerated in four in-vessel composting tunnels on-site for 4−5 days before being sent to the Z-Best composting facility (72 km from ZWEDC).

After being trucked down to the Z-Best facility, solid digestate from ZWEDC is placed into commercial composting bags that are approximately 100 m × 6 m × 3 m when filled. Each encased windrow is filled with approximately 635 tonnes of material and undergoes a 14-week composting cycle, during which piles are force aerated but not turned. The finished compost is ultimately sold for agricultural and landscaping applications. Emission rates of CO_2, CH_4, N_2O, and NH_3 were determined from in situ measurements at the Z-Best facility. As described in Kirchstetter et al.,[32] concentrations of emitted gas were measured from nine windrows that captured different stages in the 14-week composting cycle. Bag samples were collected at ∼35 locations across each windrow pile surface and later analyzed in the laboratory with three cavity ring-down spectrometers (Los Gatos Research, models 915-0011, N2OCM-919, and 915-0039; San Jose, CA). While bag samples were drawn, the aeration flow into each windrow was continuously measured using pairs of integrating pitot tubes (Dwyer Instruments, series PAFS-1005; Michigan City, IN). Emission rates with units of pollutant mass emitted per mass of digestate composted over the 14-week cycle were determined from the windrow-average emitted concentrations, average aeration flow, average mass of digestate per windrow, and average mass of digestate trucked from ZWEDC to Z-Best.

Solid Digestate Landfilling and Land Application. An alternative to the current ZWEDC operations, as described above, is a system in which all on-site operations are identical, but solid digestate is not sent to a composting facility. The first option is to landfill the digestate. At landfills, digestate can be handled as traditional waste or possibly used as alternative daily cover (ADC) to control insects, rodents, odors, and fire. In both cases the same material is being placed in the landfill (and ultimately covered as more waste is placed in the landfill), hence we do not expect that the use of digestate as ADC would result in substantial differences in the GHG footprint or other emissions relative to traditional landfilling. For this case, we modeled outbound trucking of raw digestate to the Newby Island Landfill nearby, which captures and flares its landfill gas. Emissions associated with the landfilling of digestate, or using digestate as ADC, are highly uncertain, and empirical data in the literature is inadequate although it is intuitive that the fugitive methane emissions will be reduced for waste that has undergone AD. Thus, we scale the emission factors for landfilled digestate based on the volatile solids content reduction that occurs during AD. For food waste, AD reduces volatile solids content by about 80%.[33] Another alternative fate for raw digestate is direct land application. In this case, we assume the raw digestate can offset the use of inorganic fertilizers like urea (as with compost) but achieves negligible net long-term carbon sequestration.[34−36] We conservatively use the same nitrogen content of 1.7% for dried digestate as food waste-derived compost because the range of digestate nitrogen contents reported in the literature is comparable to that of compost. Although uncertain, the urea offsets are a

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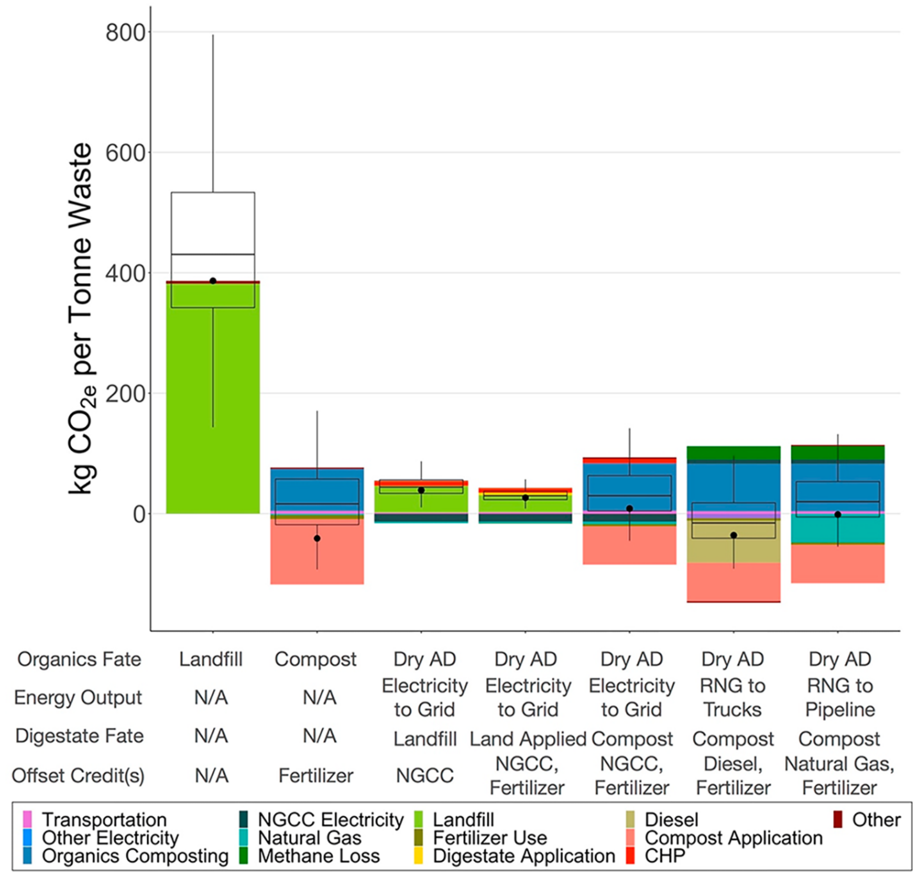

Figure 2. Life-cycle greenhouse gas emissions for all scenarios. Contributors totaling less than 1% are categorized as "Other". "Other Electricity" category refers to avoided electricity consumption from reduced urea fertilizer consumption.

relatively small contributor to the overall results, as discussed in the Results section. Another factor incorporated in our analysis are fugitive CH_4 and N_2O emissions from land application of biosolids, estimated at 65 g CO_{2e} per tonne of dry digestate.[37,38] Because of nutrient runoff concerns, land application of digestate only occurs for half of the year, with digestate being sent to landfills during the winter rainy season.[39,40]

Dry Anaerobic Digestion of Organics and Renewable Natural Gas Use for On-Site Truck Fleet. Understanding the trade-offs between on-site combustion versus RNG applications is important for owners and operators of anaerobic digestion facilities, particularly when building new facilities or expanding existing ones. Without additional cleaning (removal of H_2S and water) and upgrading (where CO_2 is removed to increase the heating value), raw biogas cannot be compressed for use in pipelines or vehicles. This means raw biogas must either be flared or combusted for on-site heat and electricity generation, as is the case at ZWEDC. In the RNG for trucks scenario, we explore a hypothetical alternative scenario in which ZWEDC utilizes its biogas to fuel a retrofitted fleet of trucks rather than combusting it for electricity generation. Conversion of biogas to RNG is energy-intensive, and reported mass/energy balances vary across the literature. Removing moisture, particles, contaminants, and other gases (such as CO_2, O_2, N_2, H_2S, and VOCs) increases the biogas methane content to 90% or more, depending on the upgrading technology. Commonly used biogas upgrading technologies include water scrubbing, pressure swing adsorption, and membrane separation. Some studies estimate membrane separation energy requirements around 0.3 kWh/m^3,[41,42] but the energy demand estimates can be as high as 0.5 kWh/m^3.[43] Pressure swing adsorption and water scrubbing require around 0.2 kWh/m^3 and 0.27 kWh/m^3, respectively.[43] We use an approximate value of 0.32 kWh/m^3 with a 0.6% loss factor and methane content of upgraded biogas of 96%. Because the biogas is being compressed and thus longer-term storage (beyond a few hours' worth of production) is more feasible, we conservatively approximate that venting events can be cut by 50% relative to the base case and flaring is also reduced by 50%. Flaring is not reduced by more than half because some rich gas will still be required as supplemental fuel when lean gas is flared. We assume produced RNG displaces diesel use in trucks that would be fueled on-site (Figure 1).

Dry Anaerobic Digestion of Organics and Renewable Natural Gas Pipeline Injection. Upgraded biogas with methane content more than 96% can also be used as renewable pipeline-injected natural gas. The upgrading process and associated energy demand is identical to the case described above for on-site RNG use in trucks. However, the offset credit is different because we assume the RNG displaces fossil natural gas (as opposed to offsetting diesel in the on-site truck fleet scenario) in unspecified end-uses and that the facility transports biogas via an interconnecting pipeline to an existing commercial pipeline located one mile away. In other words, end-use emissions are assumed to remain unchanged relative to a base case in which fossil natural gas is used. Emissions associated with the construction of the one-mile pipeline

https://dx.doi.org/10.1021/acs.est.0c00364
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interconnection are assumed negligible when amortized over its lifetime and thus are excluded.

Life-Cycle Emissions Inventory. The life-cycle inventory includes the following emissions: CO_2, CH_4, N_2O, NO_x, NH_3, NMVOC, SO_2, CO, and $PM_{2.5}$. These are all evaluated across a common functional unit of one wet tonne of organic waste processed (Figure 1). To construct a life-cycle inventory for each scenario, we collected direct mass and energy flow data, using as much measured and facility-logged data as possible from the ZWEDC facility's four years of operation. This is particularly important given the lack of data on dry AD and solid digestate composting in the existing literature, as well as the gap between best practices in an idealized scenario and what is typical at organic waste management facilities that handle highly contaminated waste streams. Through a collaboration with the ZWEDC facility owners and operators, we accessed inbound and outbound logs, including organics by type, residuals (trash for landfilling), and solid digestate. The facility also provided total biogas production, biogas flared, and electricity production; venting frequency and duration at the storage bladders were measured by the coauthors on-site.[32] As described in Kirchstetter et al.,[32] venting volume of biogas released to the atmosphere was determined with measurements of CO_2 (LI-COR, model LI-820; Lincoln, NE), gas temperature (Onset, HOBO model UX120 with Type T thermocouple; Bourne, MA), and gas velocity (The Energy Conservatory, model DG-700; Minneapolis, MN) within the pressure relief valve chimney for one of the two ZWEDC biogas storage bladders. Emission factors for digestate composting, biogas flaring, and biogas venting are all based on measured values at Z-Best and ZWEDC. Values that could not be or were not directly measured are assembled from literature sources, including peer-reviewed articles, GREET, and the Ecoinvent database (Table S1).

Direct mass and energy flows from the waste sources to final product(s) were incorporated into a physical units-based input-output life-cycle inventory model, Agile-Cradle-to-Grave (Agile-C2G), which has been documented extensively in previous literature.[44−47] This model was used to calculate indirect emissions associated with electricity generation, fertilizer production, diesel fuel production, and other minor material/energy inputs. California-based sources were considered wherever appropriate. To account for net CO_2, CH_4, and N_2O emissions after land application of composted organics, raw digestate, and composted digestate, we use GHG emission and sequestration factors documented in Breunig et al.[44] Details are also provided in Table S1. Other non-GHG air pollutant emission factors after land application are assumed to be negligible relative to the emissions during waste management, AD, and composting.

To capture parameter uncertainty, we established probability distributions for key parameters based on previous literature and used these in a Monte Carlo analysis (see Table S3). The model was run for 10,000 trials drawing from these distributions to develop the box and whisker plots shown in the results. Although the distributions were developed based on wide-ranging literature values from both inside and outside California, the expected values (denoted by black dots in Figure 2 and Figure S8) indicate values specific to the ZWEDC case study. At times, the specific study result may lie beyond the upper or lower quartile because the measured values at ZWEDC or Z-Best are not in the middle of the ranges published in previous literature. This text will focus its discussion on the expected-value results for ZWEDC/Z-Best.

Social Cost and Public Health Damage Cost. Although monetized externality estimates are an imperfect measure of environmental impacts, converting GHG emissions and air pollutant impacts into social costs is useful. First, these estimates provide a means of comparing different inventory metrics based on their relative importance to one another. Second, monetizing human health damages allows for differentiation between emissions that occur within or outside densely populated areas and thus the expected impact on the population. Third, the dollar values provide some guidance as to what governments may wish to pay in order to avoid undesirable externalities. To account for the human health damages associated with air pollutant emissions, we compare two common integrated assessment models: Air Pollution Emission Experiments and Policy analysis (APEEP, specifically version 3, hereafter referred to as AP3) and Estimating Air Pollution Social Impact Using Regression (EASIUR).[48−50] Multipliers to convert emissions to social costs are provided in Table S4 of the SI. In these cases, we include only pollutants that occur locally, either at the ZWEDC facility, Z-Best compost facility, or nearby transportation routes, assuming ground-level emissions values. The damage factor most difficult to refine on a scientific basis is the social cost per tonne of CO_{2e} emitted, and the cost of carbon used in regulations can be highly politicized. We use a relatively conservative social cost of carbon of \$42 per tonne CO_{2e}, which was established by the Interagency Working Group on the Social Cost of Greenhouse Gases for use in regulatory analyses.[51]

RESULTS

The results of our analysis are presented in three sections. First, we show life-cycle GHG emissions results, followed by results for all air pollutants (NO_x, NH_3, NMVOC, SO_2, CO, and $PM_{2.5}$). Last, we convert these life-cycle inventory results into monetized damages using the multipliers discussed in the Methods and Data section and provided in Table S4.

Life-Cycle Greenhouse Gas Inventory. The life-cycle GHG results (see Figure 2), which include nonbiogenic CO_2 as well as all CH_4 and N_2O emissions, normalized using 100-year global warming potentials (298 and 25, respectively), indicate that landfilling organic waste is the most GHG-intensive option on a per-tonne basis, with a GHG footprint of almost 400 kg CO_{2e} per tonne of organic waste. Any option for diverting organic waste, particularly higher-moisture material such as food waste that releases substantial fugitive methane, provides GHG benefits. The footprint will be roughly doubled if organics are sent to a landfill without a functioning gas capture system in place. The next most GHG-intensive options are the dry AD configurations in which some or all of the solid digestate must be landfilled. If all digestate is landfilled, the GHG footprint is 40 kg CO_{2e} per tonne of organic waste. As mentioned in the Methods and Data section, solid digestate can only be land applied for a portion of the year in California because of water quality/runoff concerns during the rainy season, so the land application scenario still results in large landfill emissions. Thus, the land application scenario reduces, but does not eliminate, landfill methane emissions, resulting in a net GHG footprint of 27 kg CO_{2e} per tonne of organic waste. Each of these scenarios is dominated by landfill methane emissions. Some facilities may choose to avoid this seasonal

https://dx.doi.org/10.1021/acs.est.0c00364
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limitation by trucking digestate long distances to locations that do not regulate digestate land application in the winter. In that case, the avoided landfill GHG emissions are likely to be larger than the increased trucking emissions. However, depending on the local climate where digestate is land-applied, there may be other concerns such as increased nitrogen runoff and N_2O emissions.[44]

The GHG footprints of composting raw organics and the three dry AD scenarios that do not require any landfilling of solid digestate all have much lower GHG footprints than scenarios that involve landfilling. The scenario that combines dry AD, electricity generation, and composting digestate (ZWEDC current operations) results in a net GHG footprint of 9 kg CO_{2e} per tonne of organic waste. The composting scenario and the two AD with RNG scenarios all resulted in net negative GHG emissions. These results are reflective of the specific conditions defined in the model and cannot be directly applied to future conditions. Offsets and negative emissions are dependent on the avoidance of current emission-intensive processes (e.g., carbon-intensive electricity generation, fertilizer use). The factors driving the differences between these three net negative scenarios, such as the net soil carbon impacts of compost application, are nuanced and uncertain. This finding is consistent with previous literature, as shown in the meta-analysis by Morris et al.[6] Composting results in the lowest GHG footprint, totaling −41 kg CO_{2e} per tonne of organic waste. A large GHG sequestration credit and a more limited fertilizer offset credit are both based on expected benefits from land application of the compost. If biogas is upgraded to RNG and used to offset diesel fuel use in a fleet of new or retrofitted trucks, the net GHG footprint is −36 kg CO_{2e} per tonne of organic waste (in this scenario, digestate is sent to be composted). This demonstrates that offsetting diesel can avoid a larger quantity of fossil CO_{2e} emissions than offsetting NGCC electricity, as is assumed in the biogas-to-electricity scenarios. Upgrading biogas to RNG and injecting it into the pipeline for use in place of fossil natural gas results in reduced GHG mitigation (−2 kg CO_{2e} per tonne of organic waste) relative to the scenario in which RNG offsets diesel use.

A point of confusion may be the fact that cleaning up biogas and injecting it into the pipeline to be combusted in place of fossil natural gas (at a power plant or otherwise) is preferable to combusting raw biogas on-site for electricity and heat. The process of cleaning and upgrading biogas does, after all, involve energy inputs and methane losses. ZWEDC operates two 800 kW engines at approximately 40% efficiency, not accounting for rich biogas that must be flared or vented when units are down for maintenance or are otherwise not able to utilize all available biogas. Aside from heat losses during electricity generation, 30% of rich biogas is flared at ZWEDC, and a negligible fraction is vented. By comparison, NGCC plants are able to use waste heat in a secondary steam cycle to generate additional electricity, resulting in an average NGCC plant efficiency across California of 47%.[52] We also assume that, once the facility invests in a gas cleanup/upgrading system and pressurized storage, flaring and venting will be cut in half, resulting in only a 15% loss. Thus, even after accounting for beneficial waste heat recovery for use in the digesters, the choice to clean and upgrade the biogas for use as a drop-in replacement for natural gas results in greater GHG reductions. If instead power exports to the electricity grid displace the average California grid mix, given its high share of renewable energy, the disparity is likely to become more pronounced. Furthermore, if pipeline-injected RNG is used for vehicles in place of diesel, the GHG advantage will grow. In short, the benefits of biogas upgrading to RNG are likely to be greater, no matter the end use of the RNG, relative to using biogas for on-site, small-scale electricity generation and export to the electricity grid in California or any other location in which the grid is relatively clean.

Life-Cycle Air Pollutant Emissions Inventory. The life-cycle air pollutant emissions vary dramatically across the different scenarios, as shown in Figure S8. $PM_{2.5}$ is recognized as the air pollutant primarily responsible for human health damages.[53] $PM_{2.5}$ can be emitted directly (primary $PM_{2.5}$) or formed in the atmosphere as the product of chemical reactions of precursors including NO_X, SO_2, VOCs, and NH_3 (referred to as secondary $PM_{2.5}$). Landfills are estimated to release the greatest primary $PM_{2.5}$ per tonne of organic waste across all options, and these emissions are dominated by the on-site flaring of landfill gas. Flares generally do not have emissions control technology, and given varying methane concentrations and imperfect mixing, they tend to emit more PM than biogas-fired power generators. The two dry AD cases in which some or all solid digestate must be landfilled are the next highest-emitting options in terms of primary $PM_{2.5}$. In contrast, the dry AD case in which RNG is used in place of diesel fuel for trucks is a net-negative because of the avoided $PM_{2.5}$ associated with operating diesel trucks (and the relatively negligible $PM_{2.5}$ emissions from RNG trucks). We do not account for potential noncombustion sources of $PM_{2.5}$ because they are expected to emit particles larger than 2.5 μm in diameter, such as dust.

Nitrogen oxides (NO_x), accounted for as mass of NO_2, and SO_2 are respiratory irritants for humans and precursors to secondary PM and ozone. Both pollutants are the product of combustion. Flares at landfills and AD facilities are the dominant source of SO_2. Flaring also emits NO_x, and because flares do not have NO_x emissions controls, the emissions per unit of fuel input are higher than for biogas and natural gas combustion in power generators, as reflected in Figure S8. Composting can result in net-negative NO_x and SO_2 emissions because direct emissions are negligible and applying finished compost to soil can reduce the need for nitrogenous fertilizer, which is energy- and emissions-intensive to produce. In the case where biogas is cleaned and upgraded to RNG, the distinction between RNG used to offset diesel in a fleet of trucks versus use in pipelines to offset fossil natural gas is critical. NO_x and SO_2 emissions are net negative if RNG is used in vehicles because of the avoided tailpipe emissions from diesel combustion. If RNG is used to offset fossil natural gas (through pipeline injection), net emissions are positive but still lower than most other scenarios.

NMVOC and NH_3 emissions are both challenging to quantify because of limited data. However, the life-cycle emissions for both are likely dominated by emissions from composting operations. NH_3 is produced during microbial decomposition, which occurs in both the digesters and during the composting process, as a way to discard excess nitrogen not required as a nutrient for the microbes. Thus, NH_3 is present in rich and lean biogas at the facility as well but is largely removed by the acid scrubber or oxidized to NO_x through combustion. In the case of NMVOCs, small negative values are owed to the fact that offsetting fossil natural gas use reduces fugitive emissions (a small fraction of which are non-methane compounds such as ethane and propane). CO emissions are also dominated by composting operations, although incom-

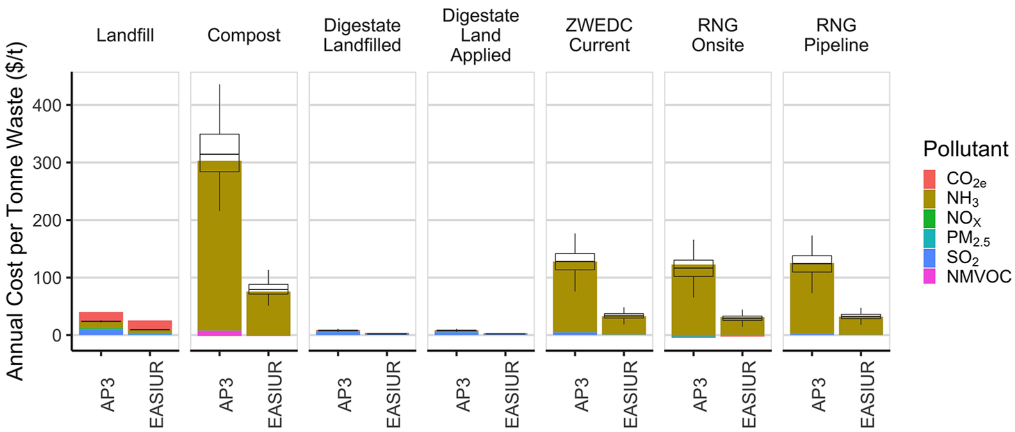

Figure 3. Life-cycle social costs of different organic waste management options as calculated in two different reduced-form public health cost tools (EASIUR and AP3), using shortened titles for scenarios outlined previously.

plete combustion during flaring and biogas-fired electricity generation also contribute to the total emissions.

To compare the social cost of primary and secondary $PM_{2.5}$ exposure and GHG emissions across waste management scenarios, we used two different integrated assessment models, APEEP and EASIUR, in combination with a \$42 per tonne CO_{2e} social cost for GHG emissions (see Figure 3). The results indicate that the social cost of landfilling wet organic waste is approximately \$25−40 per tonne (this does not include odor externalities or nonemissions related costs such as impacts on local property values). Because GHG-related damages make up the largest fraction of the overall monetized damages for landfilling, this value will change depending on the assumed social cost of carbon. For comparison, the median landfill tipping fee in California, as of 2015, was \$45 per tonne, while countries that landfill very little of their waste, including Germany and Sweden, have tipping fees around \$200 per tonne,[54] suggesting that incorporating even a fraction of the estimated social cost into tipping fees could greatly encourage diversion from landfills.

Both AP3 and EASIUER indicate that NH_3 emissions are the dominant contributor to social costs in every case where some or all organic material is composted. This is because NH_3 emissions per tonne of organic waste processed are at least 2 orders of magnitude greater than any other non-GHG pollutant in each scenario that includes composting (see Figure S8). NH_3 plays an important role in the formation of secondary $PM_{2.5}$ by reacting with nitric acid (HNO_3) and sulfuric acid (H_2SO_4), resulting from NO_x and SO_x emissions, to form ammonium nitrate (NH_4NO_3) and ammonium sulfate ($(NH_4)_2SO_4$) aerosols. However, AP3 and EASIUR disagree in some cases by more than a factor of 3, with AP3 estimating greater NH_3-related damages than EASIUR. An additional caveat is that there is very little known about NH_3 emissions from landfills before cells are capped off and gas capture/flaring systems are in place, particularly for specific waste types, such as nitrogen-rich food wastes. Similarly, very little is known about NH_3 emissions from land application of raw or composted digestate. Section 6 of the Supporting Information provides further details on the challenges of modeling secondary PM formation from NH_3 emissions, particularly in California. Despite these challenges, both EASIUR and AP3, when combined with the social cost of GHG emissions, indicate that composting has a greater social cost than landfilling and that the four scenarios including composting had the highest costs. Both assessments also predict that landfilling and land applying digestate are the least damaging options among all scenarios considered in this study. However, the models yield slightly more contrasting estimates of the relative health impacts of the ZWEDC and two RNG scenarios versus landfilling: AP3 indicates that landfilling is the preferred option cutting emissions nearly in half relative to the other scenarios, while EASIUR indicates that landfilling is the least damaging of these scenarios but only 10% lower in cost than the RNG Onsite scenario.

DISCUSSION

This study reveals the complexity of estimating environmental trade-offs in organic waste management systems and the difficulty of making broadly applicable recommendations for how organic waste should be handled. Previous literature has indicated consistently that landfilling is the least attractive option, even in more tightly regulated states like California that require efficient gas-capture systems.[6] Our GHG emissions results reinforce this conclusion. Fugitive methane emissions are the key driver in the GHG footprint of organic waste, and any scenario in which organics are landfilled will result in higher GHG emissions. The offset credits for electricity, RNG, and finished compost are also important for determining both net GHG emissions and other air pollutant emissions. If, for example, compost application does not cause a net reduction in nitrogenous fertilizer use, the net negative values for NMVOCs, NO_x, SO_2, and GHGs will be eliminated. The question of whether RNG offsets diesel or fossil natural gas will have a substantial impact on net NO_x emissions. However, on a social cost basis, none of these changes to the assumptions would alter the basic conclusions.

Our results suggest that NH_3 emissions resulting from composting nitrogen-rich waste may outweigh any other air pollutant or GHG-related social costs, yet NH_3 emissions are not well documented for organic waste management systems, even relative to other air pollutants such as VOCs. At the very least, these results warrant further study to determine how NH_3 emissions and human health damages will vary based on waste composition, composting practices, and local meteorology. The results also call into question the wisdom of making waste management decisions based solely on GHG emissions, given the potential for unintended human health consequences. If NH_3 emissions are confirmed to be a driving factor in social costs of organic waste management options and are

https://dx.doi.org/10.1021/acs.est.0c00364
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indeed greater on average at composting sites relative to landfills, the larger question is how and to what degree those emissions can be reduced. Because large composting windrows are not well-mixed controlled environments, some pockets of excess nitrogen are inevitable, particularly when nitrogen-rich food waste or digestate serves as the input. However, maximizing microbial activity and thus increasing demand for nitrogen through improved monitoring and control of pH, temperature, and aeration level during the composting process can reduce NH_3 emissions.[55] Another alternative for minimizing total social cost is to locate large composting operations in sparsely populated areas, although this may result in environmental justice/inequity issues if rural populations are socioeconomically disadvantaged relative to urban populations. Further empirical research, exploring a range of material types and composting practices, will be essential to better understanding which options for diverting organic waste from landfills provide the greatest public good.

■ ASSOCIATED CONTENT

Supporting Information

The Supporting Information is available free of charge at https://pubs.acs.org/doi/10.1021/acs.est.0c00364.

Additional methodological details and input data (PDF)

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Notes

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■ ACKNOWLEDGMENTS

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 valuable 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 between Lawrence Berkeley National Laboratory and the U.S. Department of Energy. This study was also supported by the U.S. Department of Energy, Energy Efficiency and Renewable Energy, Bioenergy Technologies Office. The United States Government retains and the publisher, by accepting the article for publication, acknowledges that the United States Government retains a nonexclusive, paid-up, irrevocable, worldwide license to publish or reproduce the published form of this manuscript, or allow others to do so, for United States Government purposes.

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