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Energy production from waste: Evaluation of anaerobic digestion and bioelectrochemical systems based on energy efficiency and economic factors☆

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ABSTRACT

Anaerobic digesters (AD) and bioelectrochemical systems (BES) are becoming increasingly popular technologies for the generation of renewable energy from wastes. Synergies between these technologies exist, however, configurations to couple them have been insufficiently investigated. This study compares the theoretical energy efficiencies of converting waste directly into electricity, using AD and BES alone and in various combinations. This study reviews the experimentally demonstrated energy efficiencies reported in the literature with comparisons to the maximum theoretical efficiencies, considering thermodynamic limits. Acetate is used as an ideal substrate for theoretical calculations, whereas complex wastes are used for extended analyses of practical efficiencies. In addition, to evaluate the economic potential of this technology, a brief case study was conducted using the Oak Ridge National Laboratory (ORNL) water resource recovery facility (WRRF). Sensitivity analysis was performed on several parameters in the economic model. The results of this study indicate the combined Anaerobic Digester/Microbial Electrolysis Cell (ADMEC) process may be the best path forward due to the high energy efficiency, combined with potential economic benefits, but is not at commercial readiness. We estimate energy efficiencies of 52.9% and 45.6% for the ADMEC process, using current state-of-the-technology, for converting food waste and sewage sludge to a CH_4/H_2 mix, respectively. This study concludes with a discussion of new strategies to improve the energy efficiency of AD and BES processes.

Significance: The analysis performed in this study supports the implementation of anaerobic digestion with bioelectrochemical systems for the production of energy from complex wastes. The energy efficiency analysis alludes to research areas that should be pursued to maximize the performance of these technologies in large-scale installation, based on the performance gaps between theoretical and practical energy efficiencies determined in previous studies.

Abbreviations: AD, Anaerobic Digestion; ADMEC, Anaerobic Digestion & Microbial Electrolysis Cell; ADMFC, Anaerobic Digestion & Microbial Fuel Cell; AOP, Advanced Oxidation Process; BES, Bioelectrochemical System; BOD, Biochemical Oxygen Demand; CE, Coulombic Efficiency; CCE, Cathode Conversion Efficiency; COD, Chemical Oxygen Demand; E_x, Energy, for a particular process x; η_x, Eta, energy efficiency for a particular process x; HE, Hydrogen Efficiency; HRT, Hydraulic Residence Time; I_x, Current, for a particular process x; MEC, Microbial Electrolysis Cell; MFC, Microbial Fuel Cell; MGD, Million Gallon per Day; MSW, Municipal Solid Waste; NPV, Net Present Value; OLR, Organic Loading Rate; ORNL, Oak Ridge National Laboratory; O&M, Operation and Maintenance cost; PEMFC, Proton exchange membrane fuel cell; TS, Total Solids; V_x, Voltage, for particular process x; WRRF, Water resource recovery facility; WWTP, Wastewater Treatment Plant; Y_{H2}, Hydrogen Yield

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1. Introduction

The strong international dependence on fossil fuels for energy generation and the sensitive relationship between water and energy requires new energy technologies to perform at high standards while utilizing natural resources in an environmentally and socially responsible way [1]. In the US, fossil fuels provide 77.5% of the primary energy supply, however food wastes, sewage sludges and other wastes represent an underutilized renewable feedstock for the production of electricity, hydrogen gas, biomethane, and biochemicals that can be used to offset fossil fuel demand [2–6]. Municipal solid waste (MSW) is one of the potential sources of energy with over 60% containing organic material, including paper and paperboard. The food waste fraction alone (14.6% of total MSW) is produced at a rate of 33.5 billion kilograms (10^{12} g or Tg) per year by individuals, with an additional 27 billion kilograms generated by retailers [7,8]. In terms of chemical oxygen demand (COD), food waste represents a resource of 24.4 billion kilograms of COD per year [9]. Landfilling is the most common method of food waste disposal (~54%) but composting, incineration, anaerobic digestion, gasification, combustion, torrefaction, and pyrolysis are also used [3,7,10–12].

Similarly, wastewater sludge generated during the treatment of domestic and industrial wastewater represents a second potential energy resource. In the US, the effluent standards for secondary wastewater treatment are 30 mg/L Biochemical Oxygen Demand (BOD) [13]. In the US, each person produces approximately 80 g of sewage solids per day which are treated in water resource recovery facilities (WRRFs), leading to a production rate over 9 billion kilograms per year, which is equivalent to 13.40 billion kilograms of COD per year. [14]. Conventional disposal methods for sewage sludges include anaerobic digestion, fermentation, gasification, incineration, and pyrolysis [15–17]. A summary of the energy resources provided by food waste and sewage sludge is provided in Table 1. Many of the disposal methods for food waste and sewage sludges rely on thermochemical processes, but these are typically less energy efficient, due to high moisture content [18].

In contrast, biological processes represent a group of technologies capable of generating energy from waste without the need to reduce moisture content. Anaerobic digestion (AD) represents a mature biological treatment process but more recently, bioelectrochemical systems (BES) have been proposed to treat sewage sludge and other substrates, such as food waste, in addition to anaerobic digestion [19–21]. Furthermore, the biological treatment processes used in this study have the potential to eliminate the need for aerobic treatment, a common component of the conventional water treatment process, which consumes upwards of 1.5% of total electricity demand in developed countries [15,22–24]. AD and BES can be integrated into waste treatment processes to establish net-energy positive treatment facilities [23,25,26].

Anaerobic digestion (AD) is a robust, mature bioconversion process that can utilize both food waste and sewage sludge as substrate [10,15]. Methane produced during AD can be converted into electricity and heat, which can be used to offset energy use in MSW facilities and WRRFs. To supplement the performance of AD, it has been proposed that bioelectrochemical systems (BES) can be used as a secondary treatment stage [19]. Two BES technologies are considered in this study, microbial fuel cells (MFC) and microbial electrolysis cells (MEC). MFCs produce electricity directly from waste and MECs produce hydrogen gas. However, large differences are observed in the performance of these systems at small vs. large scale due to increases in electrochemical losses with scale, engineering issues like reactor dead space, diffusion limitations, and high internal resistances [30,31]. A review of laboratory and pilot systems was reported in Janicek et al., which states that the performance of milliliter scale systems do not directly translate to larger scales [30]. Interest in MECs has increased significantly in the past few years due to its ability to produce hydrogen and its operational advantages over MFCs [21,31]. A review of small and large-scale MECs was also reported by Escapa et al., which concludes that MECs are an immature technology facing several barriers, such as large capital costs and hydrogen management, but show promise from recent pilot scale studies and offer unique benefits, such as mediating electrical and gas grids and utilizing a wide range of organic substrates [31]. A summary of notable AD and BES studies referenced in this report are shown in Table 2.

Previous publications have reported on the principles that outline AD and BES processes, however the focus is often only on theoretical performance [3,6,32,33–35]. While these reviews are useful for demonstrating fundamental concepts for these technologies, there is a failure to address the expected performance of these technologies with complex substrates, which is required for the planning of these systems in the real world. This report proceeds in four parts: 1) an evaluation of theoretical energy efficiency and performance based on acetate as an ideal substrate, 2) a review of the state-of-the-art technology used for anaerobic digestion (AD), microbial fuel cells (MFCs), and microbial electrolysis cells (MECs) at laboratory and pilot scales, 3) estimation of energy efficiency and performance using complex wastes at large scales, and 4) calculation of potential economic, using ORNL WRRF as a case study. This WWRF has an average daily capacity of 0.2 MGD (757 m^3/d) with an average incoming COD of 300 mg/L. To address the flexibility of these technologies, in part 3, we will investigate AD and BES technologies as standalone and integrated processes (Fig. 1). The goal of this study is to investigate a group of bioconversion systems capable of maximizing the energy recovery from abundant waste streams. The study concludes with a discussion of the energy efficiency losses and current methods available to reduce the gap between theoretical and practical efficiencies.

The energy efficiency results of this study suggest that an integrated ADMEC system has potential to be implemented as an energy-positive water treatment system. However, the economic analysis shows there are several system components that must be improved in order to see positive economic returns, at least within the constraints of this case study. Biodegradability of substrate is the most significant variable, in terms of influencing energy efficiency and economic return. The discussion section includes strategies to improve the biodegradability of substrate. Pretreatment of the organic substrate prior to processing is our recommendation to improve the efficiency and economic potential of a combined anaerobic digester and bioelectrochemical system.

2. Description of system and calculations for efficiency and economic analysis

2.1. Calculation of energy efficiencies

The energy efficiencies of AD and BES systems were evaluated for multiple substrates, including acetate as an ideal substrate and food waste and sewage sludge as complex substrates. Fig. 1 illustrates the configurations used to investigate acetate, food waste, and sewage sludge as energy sources. In order to compare the different processes, a common end-product is necessary. Electricity was chosen as the

Table 1
Summary of food waste and wastewater energy content and energy recovery.

Waste Type	Resource Energy Content				
	Energy Content (kWh/kg COD)	Mass of Resource (billion kg COD)	Energy Resource (billion kWh)	Mass per Person (kg per person)	Energy per Person (kWh per person)
Food Waste	2.95	24.40	71.98	76.25	224.94
Sewage Sludge	4.08	13.40	54.67	41.88	170.84

References: [27–29].

Table 2
Literature review of selected technologies.

Platform	Substrate	Reactor Size (l)	Retention Time (d)	Applied or Maximum Power Voltage	COD removal (%)	Biogas Production (L/L*d)	Hydrogen Production (L/L*day)	Power Density Wmˆ− 2 (Wmˆ−3)	CE (%)	Reference
AD	Waste Activated Sludge	3400,000	16	–	NA	0.18	–	–	–	[45]
AD	Separated Food Waste	900,000	80	–	NA	1.51	–	–	–	[46]
AD	Food Waste	60	20–60	–	NA	2.5−8.0	–	–	–	[47]
AD	OFMSW and Sludge	30	38	–	NA	1.00	–	–	–	[47]
MEC	Domestic WW	0.3	4.5**	0.2–0.6	90	–	NA	–	26	[49]
MEC	Domestic WW	4	0.17	1	85	–	0.05	–	719*	[50]
MEC	Domestic WW	4	0.17	0.6 and 1.0	80	–	0.02	–	190*	[51]
MEC	Domestic WW	3	8**	0.7	92	–	0.02	–	238*	[52]
MEC	Domestic WW	3	0.28	0.7	75	–	NA	–	344*	[52]
MEC	Domestic WW	100	1.00	0.6–1.1	33.7	–	0.015	–	55	[53]
MEC	Domestic WW	100	1.00	0.6–1.1	65.6	–	0.007	–	41	[54]
MFC	Sludge w/ Synthetic feed	1.5	3.8**	~0.45	88	–	–	0.133 (2.02)	–	[41]
MFC	Synthetic WW	5	0.0006	0.475	NA	–	–	2 (200)	–	[42]
MFC	Synthetic WW	7.5	0.26	0.213–0.300	69–97	–	–	(2–10)	–	[55]
MFC	Synthetic WW	20	0.0049	0.25	NA	–	–	1.44 (144)	–	[43]
MFC	Acetate***	0.02	0.0016	~0.5	NA	–	–	3.650 (345)	–	[44]

* High CE values from hydrogen recycling.
** Batch Operation.
*** Fed with Ferricyanide.

standard end-product for direct comparison of viable energy yield from each process. In this study, proton exchange membrane fuel cells (PEMFCs) were used to convert hydrogen gas produced by MECs into electricity and gas turbines were used to convert biomethane to electricity. The theoretical maximum energy yields for each technology were estimated using chemical oxygen demand (COD), an indirect measure of the organic content, which can also be used to evaluate organic wastes. Acetate was used as a primary substrate for these evaluations. To evaluate energy production from food waste and sewage sludge, a loading rate of one kg COD equivalent/day of food waste and wastewater was added to each process, respectively. The differences in performance from food waste and wastewater yields compared to theoretical yields with acetate is used to illustrate the efficiency losses due to the complexity of substrates and overpotentials in waste-to-energy processes.

2.2. Calculation of theoretical energy efficiencies using acetate

The theoretical energy efficiency of AD and BES was estimated using acetate as a substrate. It is assumed that all the electrons contained in

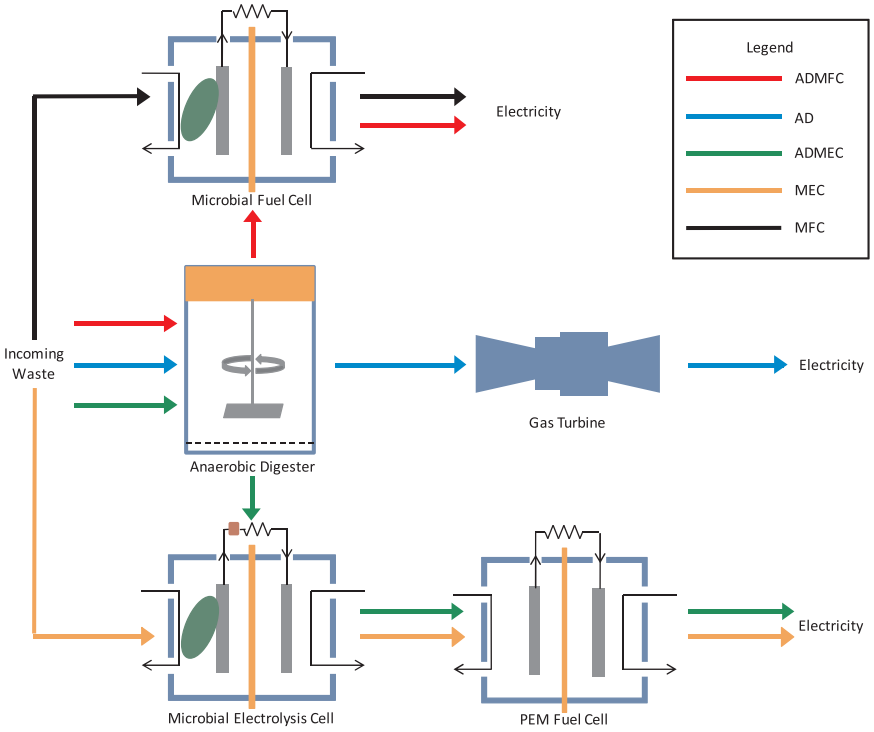

Fig. 1. Configurations of waste to energy systems. ADMFC (Red) combines an AD with a downstream MFC. AD (Blue) uses only digestion. ADMEC (Green) combines an AD with a downstream MEC. MEC (Orange) and MFC (Black) use only microbial fuel cells or microbial electrolysis, respectively. All systems ultimately generate electricity. (For interpretation of the references to color in this figure legend, the reader is referred to the web version of this article)

Table 3
Net present value of case study.

	CAPEX	Revenue	O&M	20-yr NPV
AD	$22,712.46	$1142.48	$3979.22	-$73,903.01
MFC	$35,216.00	$1563.40	$4145.02	-$81,802.88
MEC	$35,216.00	$4008.71	$9119.05	-$78,723.22
ADMFC	$25,213.17	$1443.14	$4012.38	-$47,086.62
ADMEC	$25,213.17	$1904.14	$5007.19	-$51,631.19

an acetate molecule are oxidized and utilized by the microbial communities. Our theoretical calculations include metabolic losses reported by Foley et al., which divert electrons destined for current production to microbial growth [36]. In this analysis, we assume that 12% of incoming COD is lost to microbial growth in AD and 15% of incoming COD is lost in BES [36,37]. The section below briefly summarizes the calculations for each technology and more detailed equations are shown in the Supplemental Information.

2.2.1. Microbial fuel cell efficiency

The energy efficiency for MFCs is calculated by comparing the electrical energy output to the chemical energy of the substrate. The theoretical current is determined by converting the incoming COD of the substrate into electrons (Eq. S1). The Nernst equation is used to calculate the half-cell voltage of the anode and cathode, ultimately leading to the whole cell voltage. Eq. (S2) illustrates the use of the Nernst equation for anode half-cell voltage, where E_{anode}° is standard half-cell voltage, R is gas rate constant, T is temperature, F is Faraday's constant, n is number of electrons in half-cell reaction, and Q is the quotient of products to reactants. The product of cell voltage and current yields the electrical energy output which is compared to the chemical energy of acetate, 3.778 kWh/kgCOD [38]. The calculations for determining the theoretical current and voltage of an MFC are presented in Eqs. (S1–S8). Eqs. (S9–S11) demonstrate how the energy produced by the MFC and the chemical energy of acetate are combined to calculate energy efficiency. Note that the theoretical efficiency is not 100%, due to losses from microbial growth and the calculations using the Nernst equation in non-standard conditions.

2.2.2. Microbial electrolysis cell efficiency

Similar to the MFC calculations, the theoretical current generated from acetate was calculated and used to determine the theoretical hydrogen production in MECs. The theoretical energy requirement for this system is calculated using the Nernst equation, Eqs. (S12–S23) in the Supplemental Information. Equation S13 shows the theoretical hydrogen yield from 1 kg of COD, assuming 15% biomass losses, and Equation S21 shows the electrical energy produced after conversion in a 83% efficient PEMFC [39]. However, MECs require an input of energy to overcome the thermodynamic limitations to enable the production of hydrogen. The theoretical energy requirement for this system is also calculated using the Nernst equation. The energy efficiency of the system is calculated as a ratio of the electrical energy equivalent produced by the MEC/PEMFC system to the combined chemical energy plus electrical energy consumed (Eq. S23).

2.2.3. Anaerobic digester efficiency

Acetate is an essential metabolite in the anaerobic digestion process, where it is generated as an intermediate from fermentation and is consumed by methanogens for methane production. From the literature, the reported theoretical yield of methane is 0.35 m^3/kgCOD, assuming no metabolic losses [29]. This occurs when 64 gCOD converts to 1 mol of methane. In our analysis, we included losses for microbial growth, estimated at 12% of COD [36]. In addition, the methane generated from AD, with a heating value of 10.35 kWh/m^3, is converted to electricity at an efficiency of 38% [40]. The details of these calculations are shown in Supplemental information (Eq. S24-S26).

2.2.4. Combined anaerobic digester and bioelectrochemical systems

Two combinations of the AD and BES were used to investigate economic and environmental effectiveness and to develop a more robust waste treatment design. In both of these configurations, the AD is assumed to remove 80% of the available COD. The remaining 20% COD present in the AD effluent is fed into either a MFC or MEC, resulting in two configurations: ADMFC and ADMEC. The value of 80% COD removal was chosen for the digester because it is a conservative estimate for current digester technology, compared to values used in a recent Life Cycle Analysis (LCA) study [36]. A summary of these results is shown in Equations S27–30 for the ADMFC process and Equations S31–35 for the ADMEC process.

2.3. Calculation of practical energy efficiencies using acetate

In this section, the performance parameters from a range of anaerobic digestion, microbial fuel cell, and microbial electrolysis cell studies were compared. A summary of this literature review is presented in Table 3. The analysis focused on the performance of large-scale studies to illustrate the barriers to scaling up this technology. The results from these studies were used to estimate the expected performance of AD and BES in real lab and/or pilot-scale conditions. For this analysis, it was assumed that the acetate fed into the system is readily biodegradable by microorganisms and the only losses in the system, in addition to losses to microbial growth, are those created by thermodynamic constraints and design inefficiencies. The efficiencies for gas turbines and PEMFCs are the same as above for this analysis.

2.3.1. Large-scale microbial fuel cells

Several pilot-scale studies have been carried out using MFCs. The substrates used in these studies vary but many utilize synthetic wastewater. From the literature survey, the voltage observed at the maximum power point was between 0.2 and 0.5 V. In this study, an MFC voltage of 0.3 V was used to forecast the expected improvements in large-scale MFC designs [41–44]

2.3.2. Large-scale microbial electrolysis cells

Like MFCs, several pilot-scale studies have been carried out using MECs. However, many of these studies have used domestic wastewater as a substrate. From our literature survey, we looked at the applied voltages required and the Coulombic Efficiency (CE), a percentage that estimates the electrons recovered in hydrogen from substrate, in each study and used these values in our energy efficiency calculations. Hydrogen production was estimated by multiplying the theoretical hydrogen production by the Coulombic Efficiency. The applied voltage was estimated at 0.6 V and the practical CE for pilot-scale systems was assumed to be 50%. The low CE is due to factors experienced in real systems, such as electron scavengers in the consortium, limited biodegradability of substrate, and reduced mass and charge transfer rates [30,31,56].

2.3.3. Large-scale anaerobic digestion

From our literature survey, the highest efficiency obtained in an AD study was reported by Wei et al., which demonstrated a methane yield of 0.32 m^3/kgCOD [57]. Using this methane yield, the revised energy efficiency for AD is 33.3%.

2.4. Calculation of practical energy efficiencies using complex wastes

In this final section on energy efficiency, the effects of utilizing recalcitrant substrates was evaluated. Food waste and sewage sludge could be used to generate energy via anaerobic digestion and bioelectrochemical systems. However, there are significant energy efficiency losses based on the literature survey of laboratory and pilot-scale studies. To investigate energy production from more realistic feedstocks, we reviewed the conversion of complex substrates in the literature and

estimated the energy efficiencies of these processes. A biodegradability factor was included in the calculations. The biodegradability of food waste and sewage sludge was found to be 53% and 63%, respectively, based on the literature [58–60]. In addition, the energy content of these substrates is different from acetate. As such, the energy content of food waste and sewage sludge was estimated at 2.95 kWh/kgCOD and 4.08 kWh/kgCOD, respectively [27,29,47].

2.5. Economic analysis

2.5.1. Economic model development

A case study was developed to investigate the potential economic impacts of implementing an AD/BES treatment system at the ORNL WRRF, based on the practical energy efficiencies for sewage sludge. The intent of this analysis was to identify the treatment platform with the greatest economic potential. The ORNL facility has an average daily capacity of 757 m^3/d with an average incoming COD of 300 mg/L. The economic model calculated the expected capital costs, operational costs, product revenue, and the net present value (NPV) for the system after 20 years. The annual revenue was discounted at a rate of 1% to estimate the 20-year net present value (NPV), in accordance with the US federal discount rate in 2016. Cash flow and NPV calculations are shown in the Supplemental information, Eqs. (S36–S37), where discounted cash flow is calculated as the difference between revenue and O&M costs, divided by the discount rate, and NPV is calculated as the sum of discounted cash flow, minus the initial capital costs.

In this analysis, each configuration was evaluated on the primary products created by the process; the systems were not standardized to electricity production. This change is to reflect the variable economic value of products like hydrogen, electricity, and biogas. This also opens the door for other products to be evaluated which may have greater economic value, i.e. hydrogen peroxide. For capital cost, reported values of \$100,000/ton COD*day and \$3110/m_a^3 were used for AD and BES, respectively [19,61]. The operational costs were estimated at \$0.05/kgCOD, \$0.11/kgCOD, and \$0.048/kgCOD, respectively for MFC, MEC, and AD [62,63]. The respective volumes for MFCs and MECs were estimated based on a 20 gCOD/L*day organic loading rate (OLR) which corresponds to a 11.35 m^3 anode reactor. Lastly, the revenue values for hydrogen gas, biogas, and electricity in Tennessee were \$0.10/kWh, \$0.03/kWh, and \$0.06/kWh, respectively [64–66]. To gain a more robust insight of the factors in this system, a sensitivity analysis was performed on each system indicated in the economic model, Table S7.

2.5.2. Sensitivity analysis

After establishing the economic model, a sensitivity analysis was performed to evaluate key variables in the model, including: BES Capital Costs, O&M Costs, Energy Efficiency, Sludge Biodegradability, Organic Loading Rate, and Sale Price. Each system was evaluated based on the 20-yr NPV, as mentioned above. While the primary goal of the sensitivity analysis was to observe the effects of improvements specific to BES, the sensitivity of sales price and efficiency were addressed for all systems. A summary of the 20-yr NPV's for each system are presented in Tables S8-S12. Tornado diagrams were created to provide a visual representation of the sensitivity analysis, where the vertical axis is located at the average 20-yr NPV.

3. Results and discussion

3.1. Energy efficiencies

The analysis performed in this report investigated the theoretical and practical energy efficiencies of ideal and complex substrates. The methods described above were used to compare the expected energy efficiencies of AD and BES processes with practical energy efficiencies observed in laboratory studies reported in the literature. The results are also given in Tables S7–S10 in the Supplemental Information. The bar graph in Fig. 2 represents this information so that the difference between theoretical and practical efficiencies is clearer. It is important to note that the differences in efficiency are largest in the bioelectrochemical systems. The efficiencies are quite low given the current state-of-the-art, so there is room for improvement. Increasing the efficiencies of BES technology will lead to significant economic and environmental benefits, as discussed below.

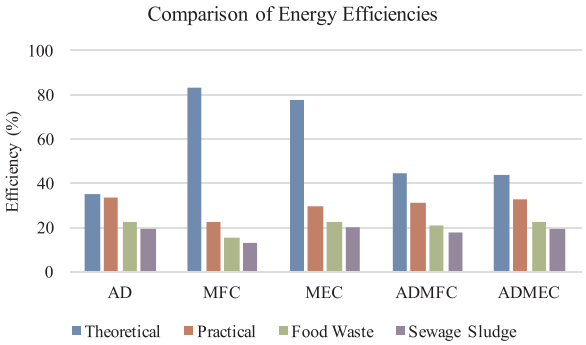

Fig. 2. Comparison of energy efficiencies for waste conversion processes.

Besides the low, practical energy efficiencies of BES, these technologies are also not adequate to handle the high solid content of complex waste streams, such as food waste and sludge. As such, BES can be used as secondary processes to AD, which are capable of processing high solid content. The combined systems also have a smaller energy efficiency gap to overcome for commercialization. To evaluate the feasibility of these configurations, a brief economic and environmental analysis was performed for all configurations at the ORNL Water Resource Recovery Facility (WRRF).

3.2. Economic benefit

The Oak Ridge National Laboratory WRRF has a capacity of 757 m^3/d and has an average influent COD of approximately 300 mg/L. Using the determined energy efficiencies for sewage sludge in Section 3.1, the 20-yr net present value for each treatment system was calculated (Table 3). The capital costs, revenue, and operational costs were listed above. The annual energy resource available for the ORNL WRRF was estimated at 200,435 kWh, based on the incoming COD concentration and the chemical energy potential of sewage COD, as determined by Heidrich et al. [29]. From this analysis, no system showed a positive return, however the ADMEC and ADMFC systems were significantly closer than AD or BES alone. The negative NPV is largely due to the low annual revenues and relatively high O&M costs for each system, which were both calculated as a function of influent COD. Installation of AD at WRRFs is typically supported by funding from government agencies such as EPA, USDA (REAP grant), etc. Thus, the negative NPV can be overcome using funds to support the capital costs of the ADBES systems, which as shown in this analysis will increase NPV. The key strategies to improve NPV are to maintain high product yield while decreasing capital and O&M costs. Some strategies are addressed below in the discussion. Improving the electrochemical energy efficiencies of BES is a key parameter for improving NPV. A value of BES that was not explored in this study is the production of non-energy products, like hydrogen peroxide. These products have greater value than electricity and could be used to supplement the revenue of the proposed configurations. Hydrogen peroxide could also be used to mitigate chemical expenses for disinfection at some WRRFs. In addition, the capital costs for BES systems are very high, relative to AD. The scaling factor of \$1220/m^3-anode is optimistic for the current state of the art and real systems will likely have much higher capital costs. This is expected because the technology is relatively new but it also requires expensive electrical components and membranes to support high conversion efficiencies.

New and novel designs without membranes can potentially reduce these costs. Overall, the combination of relatively moderate capital costs, high revenue potential, and moderate O&M costs suggests that the ADMEC or ADMFC platform are better for wastewater treatment, compared to AD or BES alone.

3.3. Sensitivity analysis

The sensitivity analysis adjusted six parameters in the economic model to search for the parameters that have the greatest impact on economic return. The six parameters were: BES Capital Costs, BES O&M Costs, Energy Efficiency, Biodegradability, Organic Loading Rate, and Sales Price. The Sales Price, Biodegradability, and Energy Efficiency had the most significant impacts on 20-yr NPV, whereas Capital Costs, O&M Costs, and Organic Loading Rate had lesser effects. Figs. 3 and 4 illustrate the Tornado diagrams for the ADMFC and ADMEC systems, respectively. The variables in this analysis resulted in a wide range of potential 20-yr NPVs for the ADMFC system, although several of these scenarios result in conditions with favorable outcomes, compared to 20-yr NPVs provided in Table 3.

In contrast, the ADMEC system showed a narrower range of potential outcomes, however the 20-yr NPVs in many scenarios were more favorable than those presented in Table 3. As mentioned above, Sales Price, Energy Efficiency, and Biodegradability show the greatest impact on 20-yr NPV. This model indicates that future work in this area should focus on areas that will improve these 3 variables.

3.4. Implications for future work

The integration of anaerobic digestion and bioelectrochemical system has the potential to recover nutrients and organic compounds from complex waste streams and transform them into energy and other valuable products. In this study, the energy efficiencies of these systems were evaluated to illustrate the technology gap between theoretical and practical efficiencies. Furthermore, the economic analysis showed that most of the systems studied in this paper would not likely have a favorable financial return based on current technology and costs. Progress toward increasing the biodegradability, reducing capital and operating costs, increase in product price and improving energy efficiencies via pretreatment of complex waste streams are all strategies that can improve the feasibility of AD and BES technologies.

As indicated in Fig. 2, there is a large energy efficiency gap between theoretical and practical performance for BES, which is a barrier to commercialization for this technology. The inefficiencies in BES can be attributed to four factors, called overpotentials: (1) ohmic losses, (2) activation losses, (3) concentration losses, and (4) metabolic losses [6,32,35,56]. Ohmic losses are characterized by the resistance of the system to the transport of protons through the electrolyte and electrons in external circuits. The spacing between electrodes and the conductivity of the medium are factors that affect ohmic losses [6]. Activation losses occur due to the thermodynamic limits of redox reactions taking place at the surface of electrodes and bacteria. These overpotentials are more prevalent at low current densities. Concentration losses are the result of mass transfer limitations at the anode and cathode and are prominent at high current densities. Lastly, metabolic losses are generated by the allocation of electrons to microbial growth and by the divergence of electrons to undesirable metabolic pathways in the microbial communities.

The optimization and improvement of BES designs and operation are a key research area to improve overall energy efficiency. For both MEC and MFC experiments, the most common reactor types are tubular and flat-plate designs. MFC studies tend to favor tubular reactors because they maintain plug-flow-like conditions with stable flow regimes [67]. While MEC studies also favor tubular and flat-plate designs, some alternative designs have been proposed, notably an MEC using the walls of an anaerobic digester as a cathode, a tubular reactor using a conductive nickel-based hollow fiber membrane as a cathode, and a reactor consisting of granular activated carbon as a fluidized anode [68–70]. Important factors for BES design are electrode spacing, electrode surface area, and materials. A review by Janicek et al. illustrates that a reduction in electrode spacing from 1 cm to 1 mm increases the power density in an MFC by over 150% from 0.907 W/m^2 to 2.34 W/m^2 [30]. Electrode spacing is influenced by reactor design (tubular vs. flat-plate), electrode material (brush, felt, granules, etc.), and the presence of a membrane separator [35]. In general, the addition of a membrane separator facilitates the reduction of electrode spacing for MFCs and MECs. However, using a membrane will add to the capital costs of the system and may reduce performance due to the formation of pH gradients between anode and cathode and increased internal resistances [52]. The effect of pH on overpotentials can be determined by the Nernst equation; a change in voltage of 0.06 V occurs per unit pH change [71]. Increasing anode surface area is another strategy to improve current density in BES. Brush anodes have been used to increase the anode surface area, leading to increases in current densities, but during scale-up the electrode spacing increases, leading to overall losses [72,73]. For scale-up, carbon cloths, fibers, and foams could be used successfully as anode materials due to their high surface area, surface properties, and conductivity, provided they can be produced at a low cost. Carbon felt has been shown to be an effective material in combination with carbon or stainless steel rod as the current collector and has potential for use in pilot studies [79], [83,84].

The delivery of substrate into a BES reactor and the electrochemical balance in the anode and cathode also needs to be addressed. Without adequate flow in the anode, pH gradients and some mass transfer limitations can occur, reducing current densities [74,75]. The method of substrate addition, batch vs. continuous, also impacts current density, as well as consortia composition [76]. In batch systems, high current densities are feasible for short periods but there are several drawbacks, such as the growth of undesired microorganisms, like methanogens [77]. Minimizing methane production and maximizing Coulombic efficiency has been reported via integration of engineering and biological control in MFCs [78,79]. In general, higher substrate concentration and loading rates will increase current production because substrate limitations are reduced. However, at very high substrate

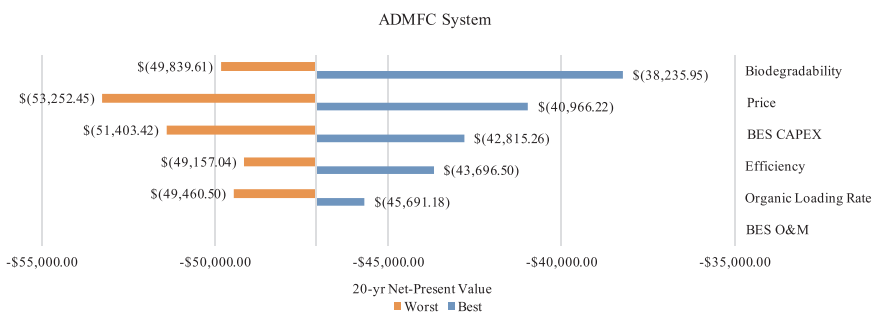

Fig. 3. Sensitivity analysis of the ADMFC system.

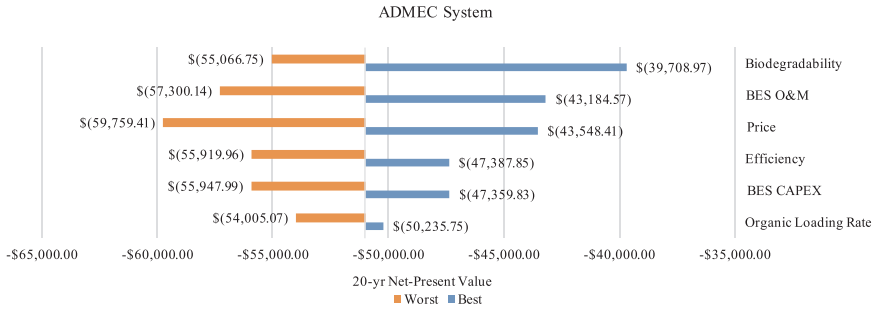

Fig. 4. Sensitivity analysis of the ADMEC system.

conditions, the excess substrate can be consumed in alternative metabolic pathways, such as methane production via methanogenesis, which will reduce the yield of current [76]. The electrons diverted to alternative metabolisms result in lower Coulombic efficiencies. In continuous systems, the flow of liquid can result in the development of shear force along the electrode surface. Shear rates can have profound effects on BES performance. Biofilms grown at higher shear rates were found to be 5-times thicker and generated current densities 2–3 fold higher than lower shear rate biofilms [80]. During the startup of a BES reactor, the external resistance needs to be monitored closely [81]. High external resistances, compared to the internal resistance of the reactor, can select for methanogens. Thus, it is important to identify the internal resistance of the system, which will change during operation. The effect of external resistances has been investigated in MFCs. Borole et al. employed a method of gradually decreasing the external resistance in an MFC over time and found an increase in current densities reaching as high as 800 A/m^3 [44]. However, lower external resistance can lead to thinner biofilms. McLean et al., observed that lower external resistances (100 Ω) led to the formation of biofilms that were 10-fold thinner than biofilms developed at higher resistances (1 MΩ) [82]. Interestingly, in this same study, the current production per cell was higher in the biofilm formed at lower external resistance. This suggests that conditions like external resistance can influence biofilm growth and structure to balance metabolism and electrode-respiration [56]. Strategies to allow growth of electroactive biofilm, while keeping external resistance low can help develop systems with high performance [56,83,84].

Strategies to reduce capital and operational costs have largely focused on finding lower-cost materials when constructing BES reactors. As mentioned above, there are alternative electrode materials for BES that could offer a low-cost alternative to conventional electrode materials in laboratory studies. Perhaps the largest single material cost for a BES reactor is the membrane. Nafion 117, a common proton exchange membrane, costs approximately $1100/$m^2$ [85]. In several BES studies, several groups have investigated membrane-less systems to reduce capital costs and mitigate pH gradients between anode and cathode [52,86,87]. Additionally, conventional anaerobic digestion systems with long HRTs (~20 days) require large footprints, increasing the capital and operational costs. The development of membrane bioreactors (MBRs) and anaerobic membrane bioreactors (AnMBRs) could significantly reduce the HRT (< 8 h) and footprint of AD reactors. A study by Smith et al. compared the life cycle costs between an AnMBR, a high-rate activated sludge process with an AD system, a conventional activated sludge process with an AD system, and an aerobic MBR with AD system. Although the overall capital costs of an AnMBR were higher than the other systems using medium-strength wastewater, the reduced sludge production greatly reduced the life cycle costs [88]. It was also noted that if the hydraulic flux of MBR membranes could be doubled, the capital costs of an AnMBR system would be reduced by 46%. There are additional benefits and applications that support the adoption of AnMBRs, such as potable and non-potable wastewater reuse applications, mitigating the release of antibiotic-resistant bacteria, and decentralized water treatment.

One of the primary cost components of the O&M in an MEC is the electricity. The difference in the O&M costs shown in Table 3 for MEC and MFC related to the electricity costs. As it is seen from the Table, this amounts to more than half of the O&M cost. Renewable electricity is becoming increasingly abundant due to the surge in wind and solar power in some parts of the country. The diurnal trend in this form of energy has resulted in a change in the peak/off-peak hours of electricity availability on the grid [89]. In some cases, this creates an excess of electricity resulting a lower than usual cost of electricity [90]. This offers a potential opportunity for MECs to be deployed for use particularly when the electricity costs are low. The cost of hydrogen production can drop under these circumstances, making it economical in those parts of the country. The hydrogen thus generated can be used as an energy storage medium and used to balance baseload power during peak hours.

Lastly, inefficiencies of substrate bioconversion or biodegradability can also lead to energy recovery losses in all microbial systems. With complex substrates, such as food waste, lignocellulosic biomass and municipal wastewater, significant fractions of the organic material are not easy broken down by electroactive microbes and cannot be converted into useful products or intermediates. Studies using model substrates typically report better performance but studies need to utilize real feedstocks to address existing concerns regarding commercialization. Several pretreatment processes can be implemented to further improve the energy recovery of these processes. Mechanical, chemical, biological, and thermal pretreatment processes have been shown to increase biogas production by 30–50%, increase methane composition in biogas and reduce solids by 20–60%, when using activated sludge [91,92]. Pyrolysis as a potential pretreatment for lignocellulosic biomass has been shown to yield high productivities of hydrogen in MEC, while generating a higher value material in the form of bio-oil for fuel production [79,93]. More innovative pretreatment procedures have been developed to address more recalcitrant organic compounds, like pharmaceuticals, pesticides, and endocrine-disrupting compounds. These include advanced oxidation processes (AOPs) like photocatalysis, ozonation, UV, hydrogen peroxide, and combined treatment processes [94–96]. Similar procedures and results have been produced using the organic fraction of MSW [97]. These processes could be applied before anaerobic digestion to increase the biodegradable fraction of organic material. Although many of these studies focus on anaerobic digestion, it is possible that pretreatment of waste before MFC or MEC reactors could also lead to improved efficiencies.

4. Conclusions

Microbe-based bioconversion processes represent a potential strategy to produce valuable energy products from waste sources. Implementing these processes could lead to economic and environmental benefits. Although the practical efficiencies of these processes are low, there are many strategies available to overcome these barriers and create more efficient systems. This study investigated the use of

bioconversion processes to improve the overall energy recovery from waste substrates. By implementing MFCs and MECs downstream of an anaerobic digester, the energy content in useful products was increased and led to overall energy efficiency improvements. The results in this study indicate that a combined ADMEC process could operate with a relatively high energy efficiency and robust treatment efficacy with the ability to produce high value energy products, like renewable hydrogen gas. While the economic costs of an ADMEC system were not positive in the ORNL case study, several strategies are available to improve the feasibility of these systems in the future. The analysis performed in this study provides a foundation for other biomass sources to be evaluated. Energy crops, agricultural waste, animal waste, and the organic fraction of MSW are all viable substrates that can be co-digested together to produce energy, leading to additional economic and environmental benefits.

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Appendix A. Supporting information

Supplementary data associated with this article can be found in the online version at doi:10.1016/j.rser.2018.07.057.

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