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Environmental and Economic Impacts of Managing Nutrients in Digestate Derived from Sewage Sludge and High-Strength Organic Waste

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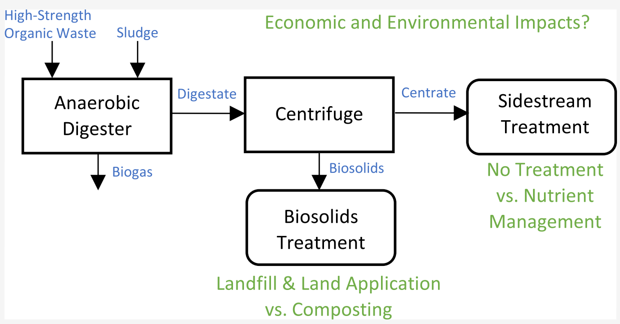
ABSTRACT: Increasingly stringent limits on nutrient discharges are motivating water resource recovery facilities (WRRFs) to consider the implementation of sidestream nutrient removal or recovery technologies. To further increase biogas production and reduce landfilled waste, WRRFs with excess anaerobic digestion capacity can accept other high-strength organic waste (HSOW) streams. The goal of this study was to characterize and evaluate the life-cycle global warming potential (GWP), eutrophication potential, and economic costs and benefits of sidestream nutrient management and biosolid management strategies following digestion of sewage sludge augmented by HSOW. Five sidestream nutrient management strategies were analyzed using environmental life-cycle assessment (LCA) and life-cycle cost analysis (LCCA) for codigestion of municipal sewage sludge with and without HSOW. As expected, thermal stripping and ammonia stripping were characterized by a much lower eutrophication potential than no sidestream treatment; significantly higher fertilizer prices would be needed for this revenue stream to cover the capital and chemical costs. Composting all biosolids dramatically reduced the GWP relative to the baseline biosolid option but had slightly higher eutrophication potential. These complex environmental and economic tradeoffs require utilities to consider their social, environmental, and economic values in addition to present or upcoming nutrient discharge limits prior to making decisions in sidestream and biosolids management.



KEYWORDS: *anaerobic digestion, nutrient recovery, energy recovery, resource recovery, wastewater, food waste, life-cycle assessment, cost analysis*

1. INTRODUCTION

Limits on nutrient discharges from many water resource recovery facilities (WRRFs) are likely to become more stringent.[1] Wastewater agencies can reduce nutrient discharge at WRRFs via mainstream or sidestream nutrient management, or upstream via building- or community-scale nutrient management.[2] Conventional approaches to mainstream nutrient management, such as nitrification/denitrification and enhanced biological phosphorus removal, can be energy- and chemical-intensive; therefore, technologies such as mainstream deammonification are gaining attention but have not reached full-scale implementation.[3] Building-scale nutrient management approaches, such as urine source separation and treatment, are promising but would require separate pipe networks for urine or routine household pick-up.[4,5]

The opportunity for sidestream nutrient management is vast considering that over 1200 WRRFs in the United States and over 10,000 WRRFs in Europe have anaerobic digesters to treat their municipal sewage sludge, thereby recovering biogas and reducing the quantity of sludge that would otherwise be landfilled.[6] In an effort to further increase biogas production and reduce landfilled waste, WRRFs with excess anaerobic digestion (AD) capacity can accept other high-strength organic waste (HSOW) streams, such as food waste.[7] Key policy drivers include ambitious landfill diversion programs for food waste, such as California Senate Bill 1383. In past studies, codigestion was shown to have reduced global warming potential (GWP) and eutrophication potential when compared to single-substrate digestion,[8] composting,[9] and landfilling.[10] While the addition of HSOW increases energy recovery,[11] it also increases nutrient concentrations in the digestate. Additional nutrients in the digestate may provide benefits if the digestate is land-applied; however, issues may arise if these

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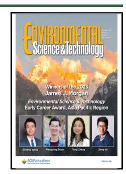

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nutrients enter surface water that is at risk of eutrophication and/or subject to stringent nutrient regulations.

Installing nutrient removal or recovery technologies to treat the high-strength ammonium and phosphate sidestream flows at WRRFs could promote environmental and economic sustainability by reducing nutrient fluxes back to the mainstream, and nutrient recovery technologies could produce fertilizer that offsets the energy-intensive production and transport of synthetic fertilizer.[12,13] Recovered nitrogen can substitute for the production of nitrogen through the energy-intensive Haber–Bosch process,[14] while recovered phosphorus substitutes for the mining of finite phosphorus reserves largely located in politically sensitive regions.[15] In this study, we evaluated sidestream nutrient management technologies that have been implemented at full scale,[16] such as struvite precipitation,[17] ammonia stripping,[18] and thermal stripping.[19] Additional nutrients exist in the biosolids, which can be composted for agricultural or landscaping purposes to improve both the nutrition and the carbon content of soils or can be directly land-applied under certain conditions.[20]

Existing studies have investigated the environmental and economic sustainability of individual nutrient recovery technologies following AD of only sewage sludge or only food waste using environmental life-cycle assessment (LCA) and life-cycle cost analysis (LCCA). Existing LCA studies on nutrient management technologies, such as SHARON-Anammox,[21,22] struvite precipitation,[16,23−25] ammonia stripping,[23,24,26,27] and composting[7,28,29] typically found greater environmental benefits at the expense of higher economic costs for chemicals and energy. However, the environmental and economic benefits and tradeoffs of the codigestion of multiple organic waste streams integrated with nutrient recovery at WRRFs are not well understood. Becker et al. determined that codigestion at WRRFs in the United States can provide lower global warming impacts than AD with only one substrate but did not analyze sidestream nutrient recovery technologies.[7] Lee et al. found similar results for high-solids codigestion in Hillsborough County, Florida, when compared to alternatives, such as landfilling, incineration, and composting.[28] We have found no studies that determine the environmental and economic sustainability of AD at WRRFs with inputs from multiple organic waste streams integrated with nutrient recovery technologies, such as struvite precipitation, ammonia stripping, thermal stripping, and composting. We hypothesize that integrating nutrient management technologies and codigestion would reduce environmental impacts with improved economic performance.

The overall goal of this study was to characterize and evaluate the life-cycle GWP, eutrophication potential, and economic costs and benefits of five sidestream nutrient management strategies and two biosolids management scenarios following digestion of sewage sludge augmented by HSOW. The questions that drive this research include the following: what are the nutrient management implications of accepting HSOW? Would recovering nutrients from digestate make sense at a WRRF that currently codigests food waste (or other HSOW streams) in addition to municipal sewage sludge? What are the tradeoffs of nutrient recovery versus nutrient removal? How sensitive are nutrient management costs to current fertilizer market prices and other factors?

2. MATERIALS AND METHODS

2.1. Site Description. The East Bay Municipal Utility District (EBMUD) serves approximately 740,000 people at a WRRF on the eastern edge of San Francisco Bay with a permitted dry weather capacity of 546,000 m^3 per day (120 million gallons per day, MGD) (and a typical average flow of 50−55 MGD) (Figures 1 and S1). The current analysis is

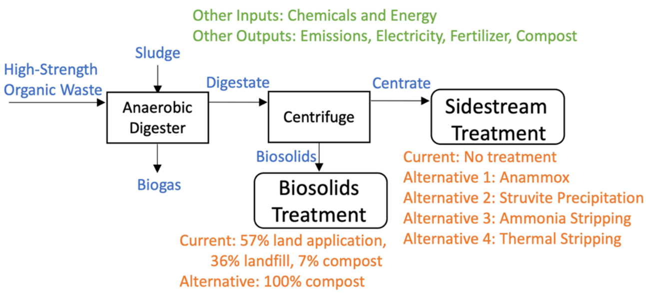

Figure 1. Process diagram for the study based on treatment of primary sludge, waste activated sludge, and high-strength organic waste at East Bay Municipal Utility District's Water Resource Recovery Facility in Oakland, California.

likely generalizable to other WRRFs as the EBMUD WRRF contains a typical process configuration of primary and secondary treatment for liquids and AD and centrifugation of solids.

The analysis was based on EBMUD's full-scale WRRF (located in Oakland) because, in addition to primary sludge and waste activated sludge, the thermophilic anaerobic digesters are fed with an average of 900 m^3 per day (0.24 MGD) of HSOW to promote additional biogas production. HSOW streams include liquid organics (43% of daily mass), protein waste (26%), dairy wastewater (22%), fats, oils, and grease (FOG) (5%), winery wastewater (1%), and other HSOW (5%) (Table S8). These HSOW streams result in typical daily loads of approximately 59,800 kg of total solids, 78,200 kg of chemical oxygen demand, 2970 kg of total nitrogen, and 550 kg of total phosphorus. Five centrifuges separate the digestate into centrate and biosolids, and the centrate is returned to the headworks of the WRRF. Although EBMUD does not currently treat its 0.7 MGD sidestream flow as no nutrient discharge limits exist, EBMUD is considering implementing sidestream nutrient management technologies to meet more stringent discharge limits that will likely be implemented in the next 5 years.[1] Liquid nutrient discharge could also be reduced by decreasing the quantity of HSOW fed to the anaerobic digesters, but biogas production would be reduced and the HSOW would still require treatment. Previously, EBMUD tested a pilot sidestream anammox system that achieved more than 90% ammonia and 80% total nitrogen removal,[1] and anammox was determined to be the leading candidate for future centrate treatment. Biosolids produced at EBMUD's WRRF are currently land-applied (57%), used as landfill daily cover (36%), or composted (7%). In 2023, EBMUD plans to eliminate landfilling of biosolids.

2.2. Scenario Descriptions. Five sidestream nutrient management strategies were analyzed using LCA and LCCA for codigestion of municipal sewage sludge with and without HSOW: (1) no additional sidestream treatment (current operation), (2) anammox, (3) struvite precipitation, (4) ammonia stripping, and (5) thermal stripping. Additionally,

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the current management of biosolids (57% land application, 36% landfill, and 7% composting) was compared to a theoretical scenario where 100% of the biosolids were composted. Combinations of the five sidestream nutrient management technologies and the two biosolids management options resulted in 10 scenarios. Considering only the input of sewage sludge (and excluding HSOW) for AD resulted in a second set of 10 scenarios. Inputs to the LCA included chemicals, energy, primary sludge, waste activated sludge, and HSOW. Outputs included emissions, electricity derived from biogas, sidestream effluent, fertilizer products, and compost. Avoided fertilizer and energy production were calculated to quantify the credits from the coproducts. A detailed inventory for each scenario can be found in the Supplementary Information (SI). Performance data for the full-scale facility were used to characterize current management, and data from the literature were used to characterize the alternate scenarios as described in the following sections.

2.2.1. Anaerobic Digesters and Centrifuges. EBMUD currently operates 11 anaerobic digesters, and each digester is 29 m (95 ft) in diameter and 15 m (50 ft) tall and has a volume of 6800 m^3 (1.8 million gallons). The digesters produce 65 m^3 (2300 standard cubic feet) per minute of biogas, which is sent to a turbine (2015 annual flow of 1.5×10^7 m^3/year, 533,000,000 ft^3/year, 47%), engine (1.3×10^7 m^3/year, 471,000,000 ft^3/year, 41%), boiler (1.4×10^6 m^3/year, 5,000,000 ft^3/year, 0.4%), and flare (3.9×10^6 m^3/year, 137,000,000 ft^3/year, 12%). The turbine, engine, and boiler, which consume 88% of produced biogas, are modeled as combined heat and power (CHP) and generate electricity. The CHP units produce heat and electrical energy, but all heat is consumed on-site and is not included as an output from our system. Some of the electricity that is produced by the CHP units is used on-site, and the remaining energy is sent to the grid and offsets local electricity production. The CHP units and other operations at the AD facility are also associated with fugitive emissions. We take into account biogas flaring and leaks from bladder venting as done in Nordahl et al.[20] Because EBMUD generates more electricity from biogas than it can use (the first WRRF in North America to do so), 14,000 MWh are supplied to the electricity grid, which generates revenue at a rate of \$58/MWh. Consistent with Nordahl et al.,[20] biogenic CO_2 emitted as a result of on-site biogas consumption is not incorporated in the life-cycle GHG footprint because, unlike fossil carbon, this CO_2 will be re-sequestered when plants and other biological material are regenerated. A GHG emissions offset credit is applied only to the portion of electricity that is exported to the grid (as a coproduct). Flaring is sometimes needed as the HSOWs are not delivered on a strict schedule and biogas production can exceed on-site storage capacity.

EBMUD currently operates five centrifuges to separate liquids and solids. The dewatering building, which houses the centrifuges along with thickening of waste activated sludge and HVAC/odor control, has a total power draw of 520 kW. Given that no additional metering exists and the centrifuges consume the vast majority of power, it was assumed that centrifugation requires 520 kW of electricity. Centrifugation also requires approximately 25 kg (55 lb) of dewatering polymer (Solenis cationic emulsion polymer) per metric ton of solids. Of the average of about 3500 m^3 per day (0.76 MGD) of digestate influent, approximately 300 m^3 per day (0.06 MGD) of biosolids are collected and transported for land application, landfill daily cover, and composting. Approximately 3200 m^3 per day (0.7 MGD) of centrate is returned to the plant headworks.

2.2.2. Current Treatment of Centrate and Biosolids. Currently at EBMUD, the centrate is returned to the head of the plant without any additional treatment. No additional chemicals and energy are needed, but no nutrients are removed and no beneficial products are made. Three destinations currently exist for the separated biosolids: 57% (26,000 kg TS/day) is land-applied, 36% (16,000 kg TS/day) is applied as daily landfill cover, and 7% (3000 kg TS/day) is composted. The biosolids require transportation via flatbed truck (which returns empty to the WRRF) for land application (25% transported 145 km to Sacramento and 75% transported 217 km to Merced), to the landfill (73 km), and to the composting facility (209 km). When modeling biosolids management emissions, land application included fugitive CH_4 and N_2O emissions, with a total GWP impact of approximately 65 g $CO_{2,eq}$ per tonne of dry biosolids, and a urea offset credit.[20] Modeling of composting emissions included fugitive emissions, a urea offset credit, and a credit for decreased soil erosion.[30] Direct emissions of composting include methane and nitrous oxide; energy impacts from compost operations are assumed to be negligible. Current biosolids treatment was compared to an alternate biosolids treatment of 100% composting.

2.2.3. Anammox. Anammox has been implemented for sidestream treatment at both the pilot and full scale.[31] Partial nitrification and ammonium oxidation under anaerobic conditions, which convert ammonium and nitrite into nitrogen gas and nitrate, can shortcut the nitrogen cycle and eliminate the need for organic carbon as an electron donor and reduce oxygen demand.[32,33] Anammox was assumed to require 1.11 kWh of electricity per kg NH_4−N for aeration and mixing.[32]

EBMUD and project partners were awarded an EPA grant in 2014 to pilot a sidestream anammox treatment system with the goal of reducing nitrogen discharge. Data were collected over 17 months from April 2013 to September 2014 on 1 m^3 (260 gallons) attached-growth and suspended-growth anammox sidestream reactors treating digestate derived from wastewater and food waste.[1] One conclusion of the study was that the anammox process can remove approximately 97% of NH_4−N from the centrate, and this removal value was used in the present study;[1] however, no phosphorus was removed and no fertilizer was produced.

2.2.4. Struvite Precipitation. Struvite precipitation reactors have been widely implemented at full scale to treat centrate.[23,34] During operation, struvite ($NH_4MgPO_4 \cdot 6H_2O$) can be precipitated from the centrate through the addition of a base (e.g., NaOH) and magnesium (e.g., $MgCl_2$).[16,35] It was assumed that approximately 1.04 kg of NaOH is needed per kg N, with a molar ratio (Mg:P) of 1.6:1.[36] Approximately 0.56 kWh of electricity was assumed per kg N for mixing.[36] Outputs include the solid struvite fertilizer and the depleted liquid effluent. The removal of ammonium and phosphate during struvite precipitation is dependent on the influent concentrations as struvite has a molar ratio (Mg:NH_4:PO_4) of 1:1:1, but phosphorus removal is typically much higher than nitrogen removal (by mass) due to the higher molecular weight of phosphate.

2.2.5. Ammonia Stripping. Ammonia stripping is established at full scale with systems in the United States and Europe. At high airflow rates through a packed bed tower, ammonium in centrate can be converted into gaseous ammonia at ambient temperatures and a pH of 11 or higher,

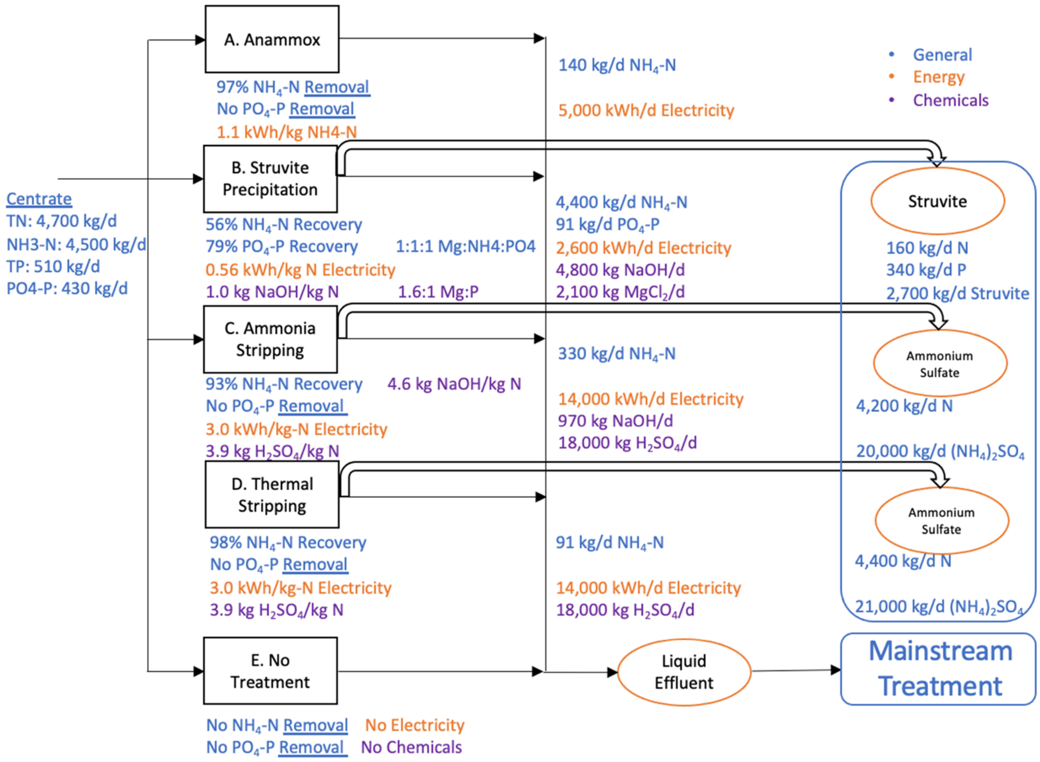

Figure 2. Five sidestream nutrient management scenarios and their associated nutrient removal efficiencies, energy and chemical input, and resulting daily fluxes with an input of centrate derived from sludge and HSOW. The quantities of nitrogen (N) and phosphorus (P) in the resulting struvite and ammonium sulfate products are shown on the right-hand side of the figure. The quantities of N and P returned to the mainstream and the energy and chemicals used during sidestream treatment are shown in the middle column of the figure.

which is then followed by acid absorption to produce a liquid fertilizer solution (e.g., $(NH_4)_2SO_4$), which is marketable as a standard agricultural fertilizer product.[32,37] Operation requires inputs of a base (e.g., NaOH), an acid (e.g., H_2SO_4), and energy. It was assumed that 4.63 kg of NaOH, 3.9 kg of H_2SO_4, and 2.95 kWh of electricity are needed per kg N.[32,36] Outputs include liquid fertilizer and the stripping residue effluent. NH_4−N recovery by the ammonia stripping unit was assumed to be 93%, and no recovery of phosphorus.[36]

2.2.6. Thermal Stripping. Thermal stripping is a process similar to ammonia stripping in which ammonia in raw digestate or centrate is stripped under a slight vacuum (0.3−1 m of water column) at typical operating temperatures of 70−90 °C without caustic addition. The stripped ammonia is absorbed using a selected acid (e.g., H_2SO_4) in the scrubber to produce $(NH_4)_2SO_4$. Based on a pilot study, it was assumed that 98% of ammonium was removed from centrate derived from a codigestion facility that blends food residues and food processing waste with primary and secondary process solids.[19] The inputs required were assumed to be 3.9 kg of H_2SO_4 per kg N[32] steam generated using biogas (1200 cfm of biogas estimated to produce steam required for centrate flow of 0.6 Mgal/day; the biogas needed could be excess gas that is available or would need to be diverted from CHP), and electrical energy for blower and minor uses (~597 kW).[19] The amount of biogas needed to produce steam (1200 cfm) is just over half the biogas produced by the digesters (2300 cfm); therefore, no additional emissions are produced, but less electricity would be sent back to the grid, reducing electricity revenue.

2.2.7. Fertilizer Offset. Fertilizers produced by the above sidestream nutrient recovery technologies were assumed to displace urea fertilizer (46% N) on a 1:1 mass nitrogen basis and monoammonium phosphate (26% P) on a 1:1 mass phosphorus basis. EBMUD has recently conducted a market assessment for fertilizers that could be produced at their facility. Based on this study, it was assumed that struvite has a strong market and buyers would pick up struvite; therefore, costs for transport would not be required, but the environmental impacts of struvite transport were still included. The study determined that liquid ammonium sulfate produced by ammonia stripping and thermal stripping would be transported approximately 322 km (200 miles) to customers. Similarly, it was assumed that struvite would be transported approximately 322 km (200 miles).

2.3. Life-Cycle Assessment. We conducted an LCA for each scenario with respect to GWP and eutrophication potential. The ISO 14044 (ISO 2006) standard and recommendations provided by Corominas et al. and Heimersson et al. were followed for the LCA.[38,39] Life-cycle emissions of greenhouse gases (GHG) (CO_2, CH_4, and N_2O) and major nitrogen-containing pollutants (NO_x and NH_3) were assessed using Agile-Cradle-to-Grave (Agile-C2G), an input−output-based hybrid LCA model.[20] For total life-cycle GWP, we consider the gaseous GHG emissions. For total life-cycle eutrophication potential, we consider gaseous N-containing emissions and N- or P-containing compounds in liquid effluent discharged to water. A life-cycle inventory for each nutrient management scenario, details on methods for calculation of GWP and eutrophication potential, sensitivity analyses, and other additional data and assumptions for each phase are included in the SI.

The system boundary for our analysis included AD, centrifugation, operation of sidestream treatment and biosolids

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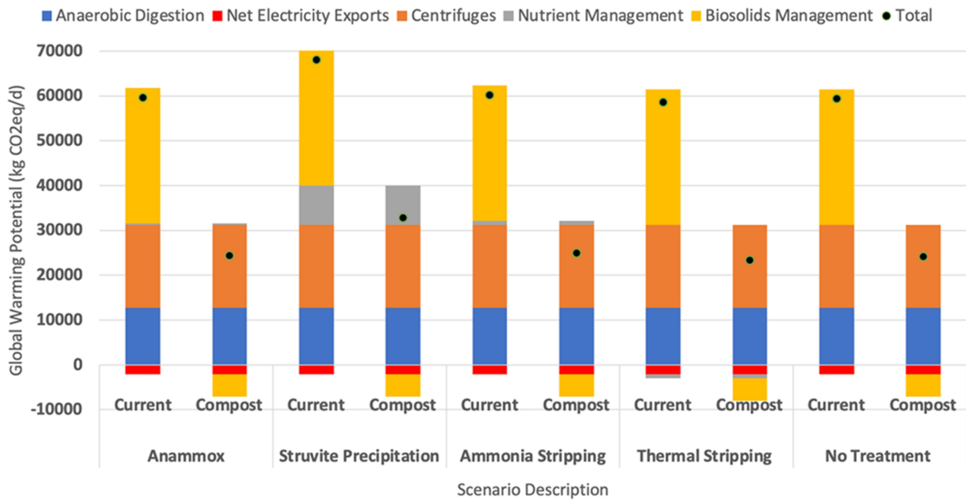

Figure 3. Global warming potential (kg $CO_{2,eq}$/day) for nutrient management and biosolids management options at East Bay Municipal Utility District (including codigestion of high-strength organic waste). Five nutrient management scenarios are considered, each with a baseline biosolids scenario (57% land application, 36% landfill, and 7% composting) and a proposed biosolids scenario (100% composting).

treatment, transport of biosolids and fertilizer products, avoided energy and fertilizer production, and relevant upstream processes (Figure 1). The functional unit was the daily mass of digester feedstocks (44,500 kg/day primary sludge and 37,800 kg/day waste activated sludge, plus an additional 59,800 kg/day for scenarios with HSOW). Sewage sludge, HSOW, chemicals, and energy were inputs into the system. Outputs included compost, struvite, liquid fertilizer from the nutrient management technologies, excess electricity generated from biogas combustion, gaseous emissions, and the depleted sidestream effluent. Real process data was utilized for current operation during AD, centrifugation, and biosolids management while utilizing sewage sludge and HSOW. Data from an EBMUD report produced by Brown and Caldwell on sidestream nutrient management unique to EBMUD were used for most prospective sidestream scenarios. Emissions from the construction phase were excluded as previous studies have shown these to be insignificant compared to the operating phase.[40]

2.4. Life-Cycle Cost Analysis. EBMUD has recently contracted with the environmental engineering firm Brown and Caldwell to conduct a nutrient reduction cost analysis. The resulting report assessed seven nitrogen and four phosphorus removal technologies considering 2050 flow and loads. The report recommended that deammonification technologies be carried forward for pilot testing, and physical−chemical ammonia recovery be pursued if deammonification is not successful in practice. For phosphorus removal, the firm recommended chemical addition of ferric chloride due to its low capital costs. Our analysis built off of this work by considering a broader scope, namely the fate of solid byproducts, as well as an additional treatment option (thermal stripping), and byproduct revenue streams. The cost analysis considers the same system boundary as the LCA, described in Section 2.3, and uses 2020 dollars for all calculations. Details on capital costs, capital maintenance costs, operational costs, resource recovery revenues, and fertilizer markets are found in the SI (Section 4).

3. RESULTS AND DISCUSSION

3.1. Nutrient Production Potential. The five sidestream nutrient management strategies differed widely in their daily mass of fertilizer produced from the nutrients present in the centrate (Figure 2). With no sidestream treatment and HSOW augmentation of anaerobic digesters, the sidestream contains 4530 kg/day of NH_4−N and 433 kg/day of PO_4−P (current operation). Anammox removed 97% kg/day of NH_4−N, leaving only 136 kg/day of NH_4−N in the sidestream return flow; however, the nitrogen was converted into N_2 gas and not recovered as fertilizer. Struvite precipitation, which typically recovers approximately 79% of phosphate, was limited by the quantity of phosphate in the centrate.[13] Given that struvite precipitates at a 1:1:1 molar ratio of Mg:NH_4:PO_4, 155 kg/day of NH_4−N and 342 kg/day of PO_4−P were recovered as struvite, while 4380 kg/day of NH_4−N and 91 kg/day of PO_4−P remained in the sidestream return flow. Ammonia stripping recovered 93% (4200 kg/day) of NH_4−N from the centrate, leaving only 326 kg/day of NH_4−N in the sidestream return flow. Similarly, thermal stripping recovered 98% (4400 kg/day) of NH_4−N and only 91 kg/day remained in the sidestream return flow. The daily production of struvite (2710 kg/day via struvite precipitation) or liquid ammonium sulfate (19,800 kg/day via ammonia stripping or 21,000 kg/day via thermal stripping) has the potential to be sold as a replacement for synthetic fertilizers. Note that anammox, ammonia stripping, and thermal stripping were all similar in their ability to remove NH_4−N (all above 90%).

For scenarios in which HSOW was not an input for AD, the daily quantity of NH_4−N in the centrate dropped from 4530 to 2860 kg/day with a similar reduction for PO_4−P from 433 to 334 kg/day (Figure S3). Given the same rates of nutrient removal or recovery, the daily production rates dropped for struvite (2100 kg/day) and liquid ammonium sulfate (12,500 kg/day via ammonia stripping or 13,200 kg/day via thermal stripping).

3.2. Global Warming Potential. The GWP (kg $CO_{2,eq}$/day) varied widely due to differences in nutrient management and biosolids management (Figure 3). The GWP for the five nutrient management options considered operation of the reactors and transportation of any recovered fertilizer as well as

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the offset from production and transport of synthetic fertilizer (Figure S4). Anammox had a relatively small GWP of 280 $CO_{2,eq}$/day, which was attributed to electricity use (Figure S4). The GWPs associated with operation of thermal stripping and ammonia stripping were much higher (10,300 and 9100 kg $CO_{2,eq}$/day, respectively) and were dominated by the chemical addition. However, after accounting for offsets from the production and transport of synthetic urea, the net GWP was slightly negative for thermal stripping (−830 kg $CO_{2,eq}$/day) and slightly positive for ammonia stripping (860 kg $CO_{2,eq}$/day). Thermal stripping benefited from utilizing steam generated directly from biogas, which offset the electrical energy demand. Struvite precipitation, which was the nutrient management option with the highest net GWP (8700 kg $CO_{2,eq}$/day), had a large contribution of GWP from chemicals as NaOH (6400 kg $CO_{2,eq}$/day) was needed to adjust pH and $MgCl_2$ (3500 kg $CO_{2,eq}$/day) was needed as a magnesium source. The GWP offset due to the transport of synthetic monoammonium phosphate fertilizer was much smaller than that of urea fertilizer due to its transport from Florida instead of China (Figure S4). No changes in GWP are expected for the mainstream treatment with the introduction of sidestream treatment because, based on internal analysis conducted by EBMUD, the difference in mainstream chemical and energy usage would be minimal as the sidestream only represents about 3% of the BOD in wastewater that goes to secondary treatment.

The 100% composting option had a net negative (−5000 kg $CO_{2,eq}$/day) GWP, much lower than the baseline biosolids option (30,000 kg $CO_{2,eq}$/day), due to the offset from compost application credits (Figure S6). AD and centrifugation had 11,000 and 19,000 kg $CO_{2,eq}$/day, respectively. The GWP from the anaerobic digesters was largely due to CHP operation (12,000 kg $CO_{2,eq}$/day) although benefits were realized by sending electricity back into the grid (−2100 kg $CO_{2,eq}$/day). The majority of emissions from centrifugation were due to use of a dewatering polymer (17,000 kg $CO_{2,eq}$/day). All scenarios with 100% composting had lower GWP (under 33,000 kg $CO_{2,eq}$/day) than the five scenarios with baseline biosolids (57% land application, 36% landfill daily cover, and 7% composted) (all above 58,000 kg $CO_{2,eq}$/day) (Figure 3).

3.3. Eutrophication Potential for Each Influent Biomass and Nutrient Management Scenario.

While AD (95 kg N_{eq}/day) contributed to eutrophication potential due to operational nitrogenous emissions (Table 1), the main source of eutrophication potential was nitrogen in the liquid wastewater effluent. To put the contribution from the sidestream into context, the eutrophication potential is reported for the entire mainstream discharge. For example, for the no treatment scenario in Table 1, the liquid effluent has an eutrophication potential of 13,000 kg N_{eq}/day; approximately 4600 kg N_{eq}/day of this is contributed from the sidestream flow. For the thermal stripping scenario, most of the nitrogen from the sidestream is removed, but the eutrophication potential for the liquid wastewater effluent is still 9800 kg N_{eq}/day (Table S16). Utilities like EBMUD that discharge to marine water bodies, which are more frequently nitrogen-limited, may choose to target nitrogen removal to reduce their eutrophication potential.[1] Results and discussion of daily nutrient fluxes for sidestream nutrient management at the EBMUD are found in the SI (Section 2, Figure S2, and Table S1). In scenarios without HSOW, eutrophication potential decreased during anaerobic digestion due to less feedstock being treated, but fewer nutrients were removed during sidestream treatment. Additionally, the HSOW that was not digested requires alternative disposal practices which have associated eutrophication potential, but this disposal takes place outside of the system boundary of EBMUD. Eutrophication potential increased from 3 kg N_{eq}/day in the baseline biosolids scenario to 30 kg N_{eq}/day in the 100% composting scenario due to increased composting.

Table 1. Local Eutrophication Potential for Anaerobic Digestion, Centrifugation, Nutrient Management, and Biosolids Management[a]

category	technology	eutrophication potential (kg Neq/day)—operation w/ HSOW	eutrophication potential (kg Neq/day)—operation w/o HSOW
	anaerobic digestion	95	39
	centrifuges	0.07	0.03
nutrient management	anammox	9800	7400
	struvite precipitation	12,300	8900
	ammonia stripping	10,000	7500
	thermal stripping	9800	7400
	no treatment	13,300	9600
biosolids management	baseline biosolids	2.9	1.2
	100% composting	30	12

[a]HSOW is high-strength organic waste. Eutrophication potential includes gaseous emissions and nutrients discharged to water.

3.4. Life-Cycle Cost Analysis.

Results from the 30-year nutrient recovery cost analysis in terms of net present value (NPV) are shown in Figure 4. Only the scenario with HSOW was modeled as capital cost estimates for the technologies were only available for with-HSOW nutrient levels (via the EBMUD consulting report). Without HSOW, capital and operating costs would be slightly lower (much less than linear cost reduction due to economies of scale) and the potential revenue would be significantly lower (Section 3.1). All nutrient recovery options assessed come at a net cost to the facility, with costs of capital and materials significantly outweighing revenues. The lowest-cost system is anammox (NPV of

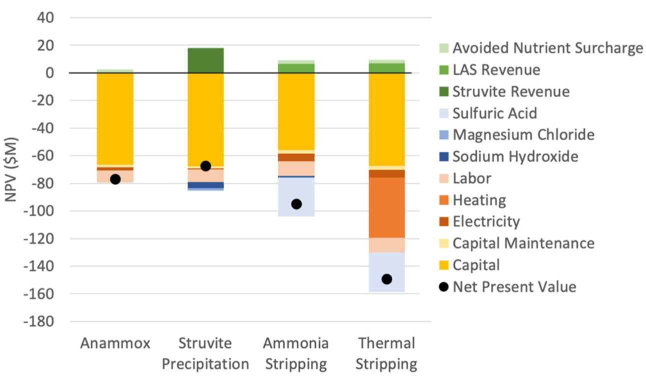

Figure 4. Net present value (NPV) reported in millions of U.S. dollars ($) for nutrient management strategies at the East Bay Municipal Utility District, Oakland (California). LAS = liquid ammonium sulfate.

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−$80M) though revenues from struvite make struvite precipitation the lowest net-cost system with an NPV of −$67M. Ammonia and thermal stripping technologies suffer from high chemical costs and relatively low revenues, plus significant energy costs for thermal stripping heating, resulting in NPVs of −$95M and −$150M, respectively. As mentioned already, the value of fertilizer may be significantly higher than shown here; if struvite market value was 4.8 times the amount used in this analysis (per the 2020 EBMUD report), and that value held over the lifetime of the facility, the recovery of the materials would come at a net positive revenue for the facility. Significant uncertainty also exists around the costs of thermal stripping as this is an earlier-stage technology that is not yet commercialized. Further research may be able to lower the energy costs of the technology though current LAS market revenue is only 1/20th the total costs of thermal stripping, and therefore cost reductions alone are unlikely to make this process profitable. If the process had a 75% cost reduction and market value of LAS increased 5×, the technology would potentially be economically favorable. A sensitivity analysis for costs can be found in the SI (Figure S8).

Although nutrient recovery is not profitable, the costs relative to the baseline costs for AD, centrifuge, and solids disposal are relatively small, as shown in Figure S7, with the NPV of total costs amounting to 1175−1325 $M per year. If all solids were sent to composting, total NPV would be $31M higher than the baseline scenario. The net costs of struvite precipitation and anammox are only 6 and 7% higher, respectively, than the scenario with no nutrient recovery. Even the most expensive technology, thermal stripping, only raises total NPV by 13%. This context is important to consider if the plant has noneconomic goals in regards to nutrient recovery, such as meeting future requirements for nutrient discharge, and continuing to provide leadership for the industry by innovating on new resource recovery strategies. The present-day monetary costs of discharging nutrients into the environment (via permit surcharges) are included but are negligible. The small scale of nutrient revenues relative to the costs of these processes (and the plant as a whole) is important to consider, as it may not be worthwhile to invest time and effort in optimizing this revenue stream through market development.

3.5. System Tradeoffs. Results from GWP, eutrophication potential, and life-cycle cost assessments reveal system tradeoffs (Table 2). GWP was dominated by biosolids management and centrifugation, eutrophication potential was dominated by nutrient management, and cost was dominated by AD. The 100% composting scenario had much lower GWP than the baseline biosolids scenario (Figure 3). While the baseline biosolids was the largest contributor to GWP, it was better than the 100% composting scenario in terms of eutrophication potential and cost due to the nitrogenous emissions and higher costs of composting. Nutrient management technologies that removed nitrogen (thermal stripping, anammox, and ammonia stripping) ranked higher than management strategies that were not effective in removing nitrogen (struvite precipitation, no treatment). While ammonia stripping and thermal stripping were effective in reducing eutrophication potential, they ranked the worst among nutrient management strategies in terms of cost due to needed chemicals and/or energy. Conversely, not treating the sidestream was the lowest-cost sidestream nutrient management option and had the lowest GWP, but did not remove nutrients and, therefore, had the highest eutrophication potential. Struvite precipitation had the highest GWP of any nutrient management option but removed phosphorus and precipitated a revenue-producing fertilizer although revenues from struvite fertilizer are currently low. Removing HSOW as an input to AD reduced eutrophication potential, but at the expense of reduced biogas production, which was used to produce electricity and sent to the electric grid at a profit (Table S2).

Table 2. Rankings of Each Scenario (1 Is Highest and 5 Is Lowest) and Relative Importance of Each Management Category in Terms of Global Warming Potential, Eutrophication Potential, and Cost[a]

management category	scenario	global warming potential	eutrophication potential	cost
nutrient management	relative importance	low	high	low
	anammox	4	2	3
	struvite precipitation	5	4	2
	ammonia stripping	1	3	4
	thermal stripping	2	1	5
	no treatment	3	5	1
biosolids management	relative importance	high	low	low
	baseline biosolids	worse	better	better
	100% composting	better	worse	worse

[a]Rankings were based on study data shown in Figure 3 (global warming potential), Table 1 (eutrophication potential), and Figure 4 (cost).

3.6. Uncertainty. Uncertainty in the process data is low as the parameters are well established. EBMUD provided daily process data for the 2010−2020 period for parameters, such as flow, nitrogen, and phosphorus. However, moderate uncertainty exists with the alternate nutrient management and biosolids management scenarios as the scenarios are hypothetical and the literature values were used to estimate daily nutrient fluxes. More established technologies like struvite precipitation have more certainty than less established technologies like thermal stripping. Additionally, actual performance of sidestream nutrient management technologies is variable depending on influent characteristics. The transportation data for the various scenarios were estimated with confidence, and overall, transportation's contribution to the environmental and cost assessments are not significant (e.g., Figure S6). The uncertainty of emission factors used to estimate GWP of electricity is also relatively low as the emission factor for electricity (55 g CO_2/kWh) derived from hydropower is based on context-specific data from Western Area Power Administration, an electric utility in the San Francisco Bay Area.[41] Eutrophication potential conversion factors are highly variable and thus highly uncertain as they are context-specific. Although high variability typically exists for composting emission factors, measured emissions from digested food waste composting in California were utilized in this study, thus lowering uncertainty. However, the high initial share of sludge-derived compost increased uncertainty. The GWP impact assessment factors for the other components of the scenarios, as well as the eutrophication potential conversion factors, are well established and standard in environmental impact calculations. The LCC are based on specific studies that are comparable to our case study; therefore, they are accepted as representative. The level of uncertainty is also expressed in the visual presentation of the project data as results are presented with just one or two significant digits.

3.7. Implications. The questions posed in the introduction, "would recovering nutrients from digestate make sense at a WRRF that currently codigests food waste (or other HSOW streams) in addition to municipal sewage sludge?" and "what are the tradeoffs of nutrient recovery versus nutrient removal?", require complex answers. No technology ranked higher than another in all three categories (GWP, eutrophication potential, and cost). The differences between nutrient recovery technologies are vast; therefore, it may not be wise to think of nutrient recovery technologies as homogeneous in comparison to nutrient removal technologies. One key economic factor is the evolving nature of fertilizer markets. Most utilities in the United States likely do not have the experience or capacity to understand the market for fertilizer products, such as liquid ammonium sulfate, struvite, and compost. Additionally, the distance between production of energy and fertilizer products at a utility and the purchaser of such products could dramatically increase both costs and environmental impacts. However, given skyrocketing fertilizer prices,[42] the economics of utilizing HSOW feedstocks and nutrient recovery technologies become increasingly favorable.

Increasingly, stringent nutrient regulations are likely to drive more utilities, such as EBMUD, to implement nutrient management technologies regardless if they recover nutrients or not. Economic expenses for nutrient management will be necessary, and sidestream treatment may be an economical option compared to mainstream treatment, or as part of an interim or hybrid solution along with upstream solutions, such as urine diversion or decentralized reuse. While the nutrient recovery technologies evaluated here for sidestream treatment appear more costly than anammox, the difference is not that significant. Operational complexity and stability of the sidestream technologies are other factors that need to be considered. For example, anammox requires long start-up times and is prone to upsets;[43] physico-chemical processes may be advantageous in this regard. Given that high ammonia levels could inhibit digester performance, removing or recovering nitrogen in the sidestream could improve digester performance and allow for additional HSOW inputs. Equalizing feeding of HSOW into the digester could also promote more regular biogas output (and therefore more revenue from electricity) and decrease the need for flaring.

There is also not an easy answer to the question, "what are the nutrient management implications of accepting HSOW?" Policies can be written in a way that rewards utilities for digesting HSOW that would otherwise be landfilled; however, if a policy does not offer such reward, reducing (or not accepting additional) HSOW inputs could be the most viable option to meet regulatory requirements. A key takeaway from this study is that the eutrophication potential is similar for no treatment without HSOW (9600 kg N_{eq}) and the best sidestream nutrient management technology with HSOW (9800 kg N_{eq}) (i.e., sidestream treatment mitigated the effect of HSOW augmentation). Another key factor on this topic is carbon pricing. Utilities in California are already shifting biosolids management away from practices like landfilling and toward practices like composting due to SB 1383. That transition may accelerate if utilities will be required to pay for their carbon emissions, or required to reduce their emissions from a baseline. Similarly, if carbon pricing was higher, there would be economic incentives to switch away from fertilizers with large GWP footprints.

■ ASSOCIATED CONTENT

Supporting Information

The Supporting Information is available free of charge at https://pubs.acs.org/doi/10.1021/acs.est.2c04020.

Additional details, figures, and tables on-site description, scenario descriptions, life-cycle assessment, and life-cycle cost analysis (PDF)

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Notes

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