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RESEARCH ARTICLE

water ENVIRONMENT RESEARCH

Integrating dairy manure for enhanced resource recovery at a WRRF: Environmental life cycle and pilot-scale analyses

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Abstract

The Twin Falls, Idaho wastewater treatment plant (WWTP), currently operates solely to achieve regulatory permit compliance. Research was conducted to evaluate conversion of the WWTP to a water resource recovery facility (WRRF) and to assess the WRRF environmental sustainability; process configurations were evaluated to produce five resources—reclaimed water, biosolids, struvite, biogas, and bioplastics (polyhydroxyalkanoates, PHA). PHA production occurred using fermented dairy manure. State-of-the-art biokinetic modeling, performed using Dynamita's SUMO process model, was coupled with environmental life cycle assessment to quantify environmental sustainability. Results indicate that electricity production via combined heat and power (CHP) was most important in achieving environmental sustainability; energy offset ranged from 43% to 60%, thereby reducing demand for external fossil fuel-based energy. While struvite production helps maintain a resilient enhanced biological phosphorus removal (EBPR) process, MgO_2 production exhibits negative environmental impacts; integration with CHP negates the adverse consequences. Integrating dairy manure to produce bioplastics diversifies the resource recovery portfolio while maintaining WRRF environmental sustainability; pilot-scale evaluations demonstrated that WRRF effluent quality was not affected by the addition of effluent from PHA production. Collectively, results show that a WRRF integrating dairy manure can yield a diverse portfolio of products while operating in an environmentally sustainable manner.

Practitioner points

- Wastewater carbon recovery via anaerobic digestion with combined heat/power production significantly reduces water resource recovery facility (WRRF) environmental emissions.
- Wastewater phosphorus recovery is of value; however, struvite production exhibits negative environmental impacts due to MgO_2 production emissions.
- Bioplastics production on imported organic-rich agri-food waste can diversify the WRRF portfolio.
- Dairy manure can be successfully integrated into a WRRF for bioplastics production without compromising WRRF performance.

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- Diversifying the WRRF products portfolio is a strategy to maximize resource recovery from wastewater while concurrently achieving environmental sustainability.

KEYWORDS
life cycle assessment, modeling, resource recovery, wastewater treatment

INTRODUCTION

Wastewater treatment plants (WWTPs) principally, and necessarily, operate with a focus on treating wastewater to produce effluent in compliance with a regulatory permit; conversely, water resource recovery facilities (WRRFs) operate to achieve recovery of valuable raw materials present in wastewater, while concurrently achieving permit compliance. In this regard, wastewater resource recovery solutions have been advocated for decades (Clark, 1930; Ganotis & Hopper, 1976; Miller, 1973; Schneider, 2011) and numerous real opportunities exist (Coats & Wilson, 2017; Guest et al., 2009; Puchongkawarin et al., 2015). Perhaps, the greatest value of wastewater is the water itself (Kehrein et al., 2020); the ability to produce potable or non-potable water from wastewater can improve situations in which water access is limited (Daigger, 2008, 2009). WRRFs also can produce valuable fertilizers, including struvite and nutrient-rich biosolids. Another significant opportunity for resource recovery is in the form of energy. In the United States alone, WRRFs consume over 3% of total electricity (Cornejo et al., 2016; Pabi et al., 2013), and electricity use accounts for 25%–40% of WRRF operating budgets (EPA, 2013). Energy recovery can be realized through anaerobic digestion of primary and/or waste activated sludge; produced methane can be combusted to heat the digesters as well as offset a portion of the energy demands at a WRRF. Others have even suggested that wastewater contains enough energy that WRRFs have the potential to be net energy producers (McCarty et al., 2011).

Beyond these conventional, established resources, WRRFs could potentially produce bioplastics. Specifically, polyhydroxyalkanoates (PHAs) are an intracellular carbon storage granule that also exhibit thermoplastic material properties, either purified from the microbial cell (Wei et al., 2014) or utilized unrefined (Coats et al., 2008). PHA can be universally substituted for petro-plastics (Shen et al., 2010), with applications including films, utensils, and packaging (Madison & Huisman, 1999). PHAs are currently produced commercially using pure microbial cultures fed synthetic substrate; however, the potential to produce PHAs using mixed microbial cultures—such as those enriched in WRRFs (Coats et al., 2016)—using wastewater has been demonstrated (Coats et al., 2007; Guho et al., 2020).

To enhance wastewater resource recovery and diversify the products portfolio, nutrient-rich non-municipal waste streams could be integrated into an existing municipal WWTP. Importantly, such a scenario would utilize the well-established skillsets of wastewater professionals who are highly trained in managing and operating such systems, while concurrently leveraging existing WWTP infrastructure and enhancing environmental sustainability. Numerous organic-rich agri-food waste streams are potentially available for co-resource recovery within municipal WWTPs; the selection would, in part, be geographically based. One viable and plentiful waste stream is dairy manure (Coats et al., 2013). Over 9.3 million milk cows generate >226 billion kg/year of wet manure in the United States (Liebrand & Ling, 2009; USDA, 2019); these numbers exclude non-lactating dairy cows, and thus, the resource quantity is even greater. Not only is this valuable source of carbon insufficiently recovered under current waste management practices, current nutrient management strategies for dairies present significant environmental challenges. Manure land application can yield excess soil P (Hristov et al., 2006) and contribute to surface water eutrophication associated with water runoff; groundwater nitrate concentrations can also be elevated (Wang et al., 1999). Recognizing these risks, the U.S. EPA has tightened dairy operation rules (EPA, 2008); more regionally, a 2015 settlement in Washington State, based on a federal court finding that dairy manure was a waste to be regulated under the federal Resource Conservation and Recovery Act, portends significant challenges to dairy operations. The dairy industry needs engineered systems that provide opportunities to "pivot" from a legacy waste management approach to solutions that concurrently achieve economic and environmental resilience; strategic integration with WWTPs represents one potential scenario.

While conceptually wastewater resource recovery, WRRFs, and potential barriers to implementation have been well-covered in the literature (Coats & Wilson, 2017; Guest et al., 2009; Holmgren et al., 2015; Puyol et al., 2017; Smith et al., 2014), as noted by Kehrein et al. (2020), more site-specific investigations are needed that evaluate, assess, and demonstrate the integration of resource recovery technologies to achieve the WWTP-to-WRRF conversion. In achieving such an outcome, metric-based environmental sustainability assessments can provide important decision-support data; indeed, it cannot simply be assumed that a newly established WRRF operates in a "sustainable" manner relative to the original WWTP that solely focused on permit compliance. Finally, moving from WRRF conceptualization to reality and concurrently integrating a proximate waste stream requires appropriate demonstration; in particular, considering the importance of WRRF permit compliance, the impacts of integrating such a high-strength waste must be carefully considered.

In an effort to contribute to the WRRF momentum, this study evaluated conversion of a municipal WWTP located in Twin Falls, Idaho to a WRRF, integrating dairy manure. The southern Idaho region is the heart of Idaho's dairy industry, with ready access to dairy manure as a co-substrate. Research conducted in this study evaluated deployment of technologies at the Twin Falls WWTP to recover and produce five high-value resources—reclaimed water, biosolids, struvite, biogas, and bioplastics (PHA)—leveraging readily available dairy manure. Research applied state-of-the-art process modeling software coupled with environmental life cycle assessment (ELCA) to describe real WRRF opportunities and quantify sustainability. Complementary to the ELCA analyses, pilot testing was conducted to further vet the integrated dairy manure-municipal wastewater WRRF.

METHODOLOGY

Description of Twin Falls, Idaho WWTP

The Twin Falls WWTP operates at an average maximum monthly flow rate of 30,240 m^3/day, focusing on ammonia, 5-day biochemical oxygen demand (BOD_5), and total suspended solids (TSS) removal. Raw wastewater undergoes preliminary, primary, and secondary treatment (Figure 1). The secondary treatment configuration is operated as a Virginia Initiative Process (VIP) configuration (Tchobanoglous et al., 2014), and also includes Integrated Fixed film Activated Sludge (IFAS); the VIP configuration, which is an enhanced biological phosphorus removal (EBPR) process, is not operated to achieve biological P removal. Anaerobic digestion of primary solids (PS) and waste activated sludge (WAS) occurs using two completely mixed mesophilic anaerobic digesters operated in series. The City produces U.S. EPA class B biosolids, which are beneficially used at local farms; the City bears all costs in producing and providing this product. Biogas is used for digester heating, with the excess flared. Twin Falls discharges treated effluent into the Snake River in accordance with their NPDES permit. The permit does not limit total nitrogen or nitrate, nor does it contain stringent phosphorus limits—the latter being expected in a future NPDES permit. Additional details on the existing WWTP are included in the supplementary data. Historically, Twin Falls has been able to comply with its NPDES permit without issue (Table 1); influent constituents are comparable to values typically associated with "high-strength" wastewater (Tchobanoglous et al., 2014).

Idaho dairy industry

Twin Falls is uniquely positioned to achieve resource recovery in large part because southern Idaho is the heart of the Idaho dairy industry (IDA, 2020); there are approximately 400,000 dairy cows in the Twin Falls area, with several

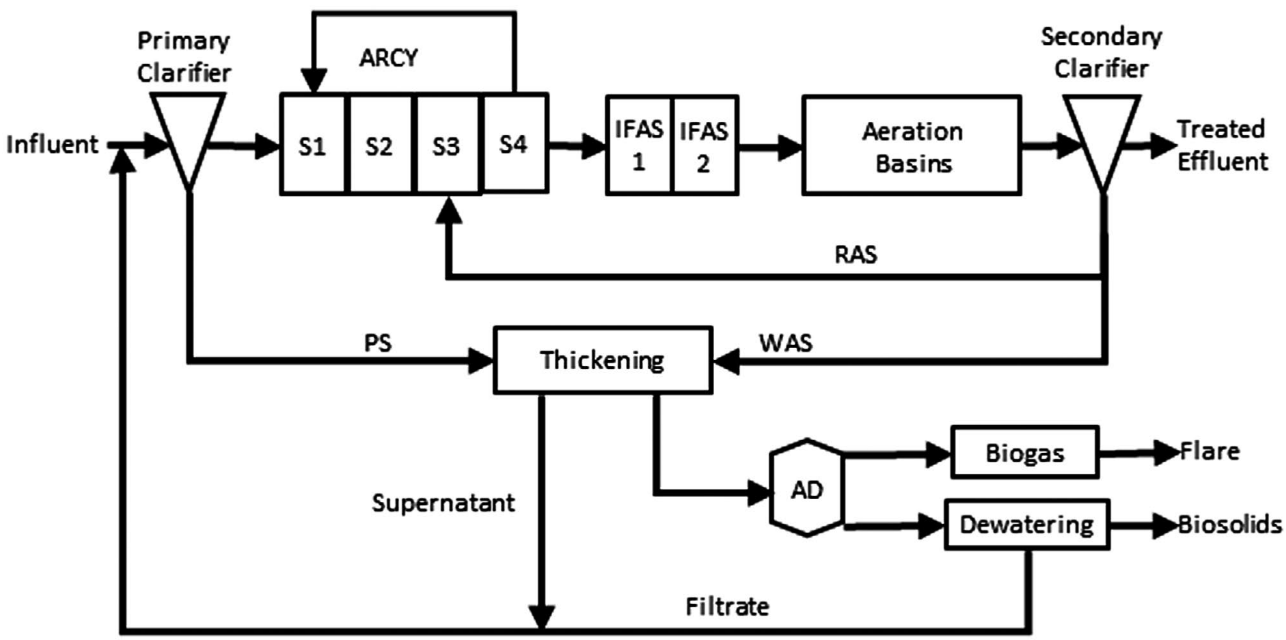

FIGURE 1 Schematic diagram of the existing Twin Falls, Idaho WWTP

large dairies (>1500 head) within 10–20 miles of the City's WWTP, and thus, significant substrate is available to enhance resource recovery. Moreover, Idaho dairies are facing increasing environmental scrutiny for manure management, with land application options becoming more limiting, and thus, alternative manure management approaches are needed. Finally, research has demonstrated that manure is an excellent substrate for deployment of a resource recovery technology that generates PHA (Guho et al., 2020). For this study, it was assumed that a re-configured WRRF would have ready access to manure produced by a 5000 head dairy.

Pilot-scale WRRFs

Two pilot-scale WRRFs were operated as part of the research to evaluate the impact and assess potential integration of PHA production on dairy manure with EBPR. Details on the dairy manure PHA pilot (Figure S10) are provided in Guho et al. (2020). The municipal EBPR scale model WRRF (Figure S11), located at the Moscow, Idaho WRRF, processes screened and de-gritted wastewater from the Moscow WRRF. The system includes an activated primary fermentation system (Krause, 2010), with a 900 L CSTR fermenter, a primary clarifier (approximate volume of 1000 L), and a positive displacement pump driven by a variable frequency drive (VFD) to return settled sludge to the fermenter; SRT, which is controlled by wasting sludge on a regular basis into Moscow's WRRF, was set at 5 days, consistent with Romenesko and Coats (2018). The secondary biological treatment system was operated to achieve post-anoxic EBPR (Coats, Mockos, et al., 2011; Winkler et al., 2011), and includes anaerobic (three CSTRs in series at 770 L each), aerobic (two CSTRs in series at 1325 L each), and anoxic (6800 oxidation ditch) environments, with a secondary clarifier (approximate volume of 2000 L) providing return activated sludge (RAS) to the first anaerobic basin. Influent wastewater and RAS are pumped using positive displacement pumps driven by VFDs. Aeration is achieved using a VFD-driven rotary lobe blower and fine bubble diffusers. Secondary system solids residence time (SRT) is controlled via Garrett wasting from the anoxic ditch; for this study, the SRT was set at 14 days. The scale model is operated under ammonia-based aeration control (ABAC) using a Hach ANISE probe (Hach, Loveland, CO, USA) and Hach SC-200 controller. The ANISE probe is installed where aerobic basin wastewater enters the anoxic ditch; the controller operates an electronically actuated valve that provides air to fine bubble diffusers in the anoxic ditch immediately downstream of the probe to maintain a maximum NH_4–N concentration of 2 mgN/L. Aeration basin DO is controlled using a Hach LDO probe and SC-200 controller; the LDO probe is installed in the 2nd aerobic CSTR, and the controller seeks to maintain a DO concentration of 2 mgO_2/L by varying the blower speed. The EBPR scale model was operated in three phases for this study: Phase 1 (56 days of operation) excluded effluent from the PHA pilot; Phase 2 (18 days of operation) followed Phase 1 and included PHA pilot effluent; and Phase 3 (71 days of operation) followed Phase 2 and excluded effluent from the PHA pilot.

TABLE 1 Average historic influent and effluent data for the Twin Falls, Idaho WWTP (2015–2018)

Parameter	Influent	Effluent	Removal %
BOD_5 (mg/L)	364.2	4.6	99
TSS (mg/L)	263.41	6.92	97
TP (mgP/L)	10.72	5.04	53
NH_4 (mgN/L)	42.86	0.16	99.6
TKN (mgN/L)	59.15	2.67	95
NO_3 (mgN/L)	0.25	15.9	N/A
pH	8.24	8	N/A

Abbreviations: BOD_5, 5-day biochemical oxygen demand; NH_4, ammonia-nitrogen; NO_3, nitrate-nitrogen; TKN, total kjeldahl nitrogen; TP, total phosphorus; TSS, total suspended solids.

TABLE 2 Summary of the respective processes and complete resource recovery alternatives integrated into each scenario

Scenario	EBPR	CHP	Struvite	PHA	
1	1a: Base Operations				
	1b: Base Operations, modified		x		
	1c: Base Operations, modified			x	
2	2a: Adding Primary Solids Fermentation	x		x	
	2b: Adding Primary Solids Fermentation	x	x	x	
3	Dairy manure for PHA + Primary Solids Fermentation for EBPR	x	x	x	x
4	Dairy manure for PHA		x		x

Abbreviations: CHP, combined heat and power; EBPR, enhanced biological phosphorus removal; PHA, polyhydroxyalkanoate.

Resource recovery scenarios

Research was conducted to establish, evaluate, and compare seven resource recovery alternatives integrated into the Twin Falls WWTP (Table 2); Figure 1 illustrates the existing WWTP, while Figures S1–S7 illustrate the proposed WRRF scenarios. Scenario 1a represents the base case (i.e., current operation) that served as the control to which other alternatives were compared. Scenario 1b integrated CHP into the WWTP, producing electricity to offset facility usage. Scenario 1c integrated struvite crystallization using the phosphorus-rich dewatering stream. Scenario 2a upgraded the VIP process to the A2O EBPR process to improve P capture, with struvite production; primary solids fermentation provided VFAs to drive EBPR. Residual primary solids and WAS would be anaerobically digested, producing biosolids but not CHP. Scenario 2b added CHP to scenario 2a. Scenario 3 built upon Scenario 2b, with the integration of dairy manure to drive PHA production as a sidestream system, consistent with the process configuration proposed by Guho et al. (2020). Finally, Scenario 4 integrated sidestream PHA production and CHP utilizing dairy manure as substrate but excluded struvite production.

Modeling resource recovery scenarios

The SUMO process modeling package (Dynamita (Lyon, France)) was used to evaluate the existing WWTP and alternate WRRF scenarios. The SUMO2 model utilizing two-step nitrification and denitrification and the Barker–Dold model (1997) for P removal was selected for this research. SUMO2 was chosen for its completeness in modeling each step of the biological process; the integrated fixed film activated sludge (IFAS) carriers in the Twin Falls WWTP contain several layers of biofilm in which oxygen limiting conditions are likely, thus contributing to the production of some nitrite. Model calibration was performed consistent with Melcer et al. (2003) using physical facility data, operational data, performance data, and influent loading data provided by facility staff. A tiered approach was taken to adjusting kinetic and stoichiometric parameters. The simulation was re-run after each parameter adjustment until the error comparing the model predicted data to the field collected data was minimized. Additional calibration details are provided in the supplementary data. Overall, the calibrated model captured the intricacies of the facility, including the limited phosphorus removal, and nitrification/denitrification, to sufficiently inform the ELCA analyses and thus serve the needs of this research.

Struvite production was modeled in SUMO as a single CSTR with no solids recycle. Although metabolic models have been developed for aerobic dynamic feeding (ADF)-driven PHA synthesis by mixed microbial consortia on waste streams (Dias et al., 2008; Wang, Carvalho, et al., 2018), to date, no model has been integrated into commercial process models. Instead, a simple stoichiometric approach was employed herein to model PHA production, consistent with data from Guho et al. (2020) and using the DAIRIEES model (Guillen, 2017; Guillen et al., 2018). With sidestream PHA production at a WRRF, effluent must be included in the secondary treatment process; phosphorus would be available for struvite production, but nutrients would also impose an additional treatment load. Thus, in the PHA production scenarios, a separate state variable stream was created in SUMO to include the additional influent N and P load. To generate data for the LCA, the PHA reactors were modeled in SUMO with fermenter liquor characteristics from Stowe et al. (2015).

Combined heat and power production was based on the biogas output from SUMO (methane percentage and biogas flow rate; Table 3). It was assumed that an internal combustion engine with electricity generation efficiency of 40% was utilized (Wiser et al., 2010). To obtain a value of electricity production, the lower heating value of methane (35,800 kJ/m^3) was used with the biogas flow, methane percentage, and 40% electrical efficiency. Alternatives involving the integration of dairy manure augment the anaerobic digesters with additional waste solids to increase biogas output. Residual solids from the dairy manure fermentation process (75.7 m^3/day) were fed to the digester; the volumetric distribution of VFA-rich fermenter liquor to PHA production and residual fermenter solids to AD was based on Stowe et al. (2015), and was modeled in SUMO using an additional influent stream. The characteristics of this stream were based on data collected by Stowe et al. (2015); a value of 2.3 gCOD/gVS was assumed to convert values to a COD basis.

Primary solids fermentation modeling was performed in SUMO using an anaerobic CSTR with a thickener, maintaining an 8-h HRT and 5-day SRT of the primary solids fermenter; Romenesko and Coats (2018) found a 5-day SRT to be optimal for VFA production on primary solids. To maintain compliance with the facility's NPDES permit, all aeration basins had to be utilized, which increased blower demands and electricity usage.

Environmental life cycle assessment (ELCA)

Environmental life cycle assessment was performed in accordance with ISO standards (ISO-14040, 1997; ISO-14044, 2006). Successful integration of ELCA into wastewater studies has been achieved numerous times (Coats, Watkins, et al., 2011; Corominas et al., 2013; Ishii & Boyer, 2015). Studies have utilized a functional unit of a volume of wastewater (Coats, Watkins, et al., 2011; Postacchini et al., 2016;

TABLE 3 Summary of emissions for each scenario associated with treated effluent quality, and resources produced

	Scenario						
	1a	1b	1c	2a	2b	3	4
Treatment emissions							
COD, mg/L	183	183	183	183	183	181	183
BOD_5, mg/L	2.3	2.3	2.3	2.6	2.6	2.7	2.3
TSS, mg/L	5	5	5	5	5	5	5
NH_4, mgN/L	0.2	0.2	0.2	0.6	0.6	0.6	0.2
NO_3, mgN/L	14.9	14.9	11.8	11.7	11.7	11.4	17.2
TP, mgP/L	6.3	6.3	1	0.6	0.6	0.6	8.6
PO_4, mgP/L	5.8	5.8	0.4	0.1	0.1	0.1	8.1
Biogas, m^3/day	5749	5749	5953	4485	4485	5546	5790
Biogas methane, %	64.2	64.2	64.2	59.7	59.7	59	62
Resources produced							
Electricity, kW	0	579	0	0	420	515	566
% Power offset[a]	0	59.6	0	0	43.2	54.7	58.3
Biosolids, t/day	5.2	5.2	5.9	6.2	6.2	8.4	6.3
Struvite, kg/day	0	0	1410	1371	1371	919	0
PHA, kg/day	0	0	0	0	0	358	358

Abbreviations: BOD_5, 5-day biochemical oxygen demand; COD, chemical oxygen demand; NH_4, ammonia-nitrogen; NO_3, nitrate-nitrogen; PHA, polyhydroxyalkanoate; PO_4, phosphorus; TP, total phosphorus; TSS, total suspended solids.

[a]Based on annual 2018 power utilization for the existing WWTP; assumes 95% up-time for the CHP system.

Rahman et al., 2016); however, this study opted for a functional unit of 454 kg of COD. By incorporating a mass of COD as the functional unit, the ELCA fully incorporates and accounts for the integration of dairy manure, which exhibits much a higher COD concentration relative to the municipal influent flow rate.

Life cycle inventory analysis

Data for the life cycle inventory analysis were generated predominantly using SUMO; changes in power demand due to a removal of the ARCY pump and/or an increase in aeration demand were separately calculated. Table 3 summarizes treated effluent quality and resources produced for each scenario. Emissions from offset of electricity were estimated using Idaho Power's fuel source mixture (Idaho Power, 2018) and a study by Turconi et al. (2013). The offset of mineral fertilizer was calculated based on production emissions for diammonium phosphate (DAP; (Manjare & Mohite, 2012)), with an NPK value of 18-45-0, similar to other studies (Foley et al., 2010; Sørensen et al., 2015). The use of biosolids and struvite in lieu of DAP assumed application on a P-limiting basis. Emissions for the production of chemicals used in the struvite reactor used data from other studies, with sodium hydroxide utilized for pH adjustment (Thannimalay, 2013) and magnesium hydroxide used for magnesium source (Li et al., 2015). Bioplastic use was assumed to replace a 50:50 mixture of high-density polyethylene and low-density polyethylene, with emissions for these petroleum plastics (Harding et al., 2007) offset by the use of PHA-based bioplastic produced at the WRRF. Transportation of both bioplastic and petroleum plastic was not considered.

Potential impacts associated with infrastructure and capital construction were not included in this study; these elements would incur a one-time environmental impact and, relative to total environmental emissions, would be dwarfed by the operation of these facilities (which exhibit a long lifespan [25–50+ years]). Similar assumptions were made in comparing wastewater P recovery (Coats, Watkins, et al., 2011); wastewater biosolids management systems (Peters & Rowley, 2009); in comparing municipal wastewater biogas management and sludge application systems (Pasqualino et al., 2009); in comparing four full-scale municipal wastewater treatment plants (Hospido et al., 2008); and in developing guidance criteria for planning metropolitan water systems (Lundie et al., 2004).

Life cycle impact assessment

The U.S. Environmental Protection Agency (EPA) ELCA model "Tool for the Reduction and Assessment of Chemical and other Impacts (TRACI)" (Bare, 2011) was selected for this research. Applying TRACI, ELCA was performed for six categories: global warming—air; acidification—air; eutrophication—air; eutrophication—water; smog—air; and human health particulate—air. Within each category, impacts were further binned based on contributions from transportation, plastics offset, effluent, struvite chemicals, electricity change, and fertilizer offset. "Transportation" captures the environmental impact of transporting biosolids and dairy manure, and sodium hydroxide and magnesium hydroxide for struvite production, as well as the reduction in transportation of synthetic fertilizer. Transportation distances were assumed as follows: 20 miles for dairy manure (10 trips per day), 25 miles for biosolids land application (one trip per day), 230 miles for sodium and magnesium hydroxide (from a supplier Utah), and 1400 miles for DAP (from a supplier in Minnesota). "Plastics offset" is the reduction in emissions associated with the usage of PHA in lieu of petroleum plastics. "Effluent" captures the impacts of wastewater constituents (COD, P, and N) discharged to the Snake River. "Struvite chemicals" captures emissions associated with the production of magnesium hydroxide and sodium hydroxide. "Electricity change" is the net change in electricity at the WRRF due to implementation of CHP, removal of the ARCY pump, and changes in aeration. "Fertilizer offset" relates to the avoided emissions due to the replacement of DAP with biosolids and struvite.

Total emission quantities from each scenario were input into TRACI and potential impacts to the environment were quantified. For each environmental impact category, the TRACI model normalizes all emissions to a single impact indicator. For example, the category indicator for Global Warming-Air is CO_2; thus, emissions that could affect this category are each multiplied by a unique characterization factor (inherent with TRACI) to normalize on CO_2-equivalence. Further, individual emissions were applied in full to each applicable category and not allotted just to a single category; thus, emissions could impact multiple categories. This approach is consistent with ISO 14042 (Curran, 2006; ISO-14042, 2000). Finally, in regards to quantifying and assessing potential environmental impacts, TRACI has been developed to characterize at the mid-point level on the cause-and-effect pathway for contaminant (Bare, 2011).

In conducting the ELCA the current Twin Falls, Idaho WWTP was established as the "base case," against which all resource recovery scenarios were compared; therefore, emissions that transcended all scenarios were excluded from the ELCA. For example, while WWTPs exhibit relatively significant energy demands—Twin Falls is no exception—and such energy demands can contribute to mid-point ELCA categories such as Global Warming Potential and Smog-air (Coats, Watkins, et al., 2011), only energy demands and associated emissions for wastewater treatment and effluent production that differed from the base case were incorporated into this ELCA.

RESULTS AND DISCUSSION

The concept of executing wastewater resource recovery within a real WWTP was evaluated from two perspectives: (i) Environmental life cycle assessment was applied to evaluate the concept of converting the Twin Falls, Idaho WWTP to a WRRF, integrating dairy manure as a carbon- and nutrient-rich substrate to produce reclaimed water, Class B biosolids, struvite, bioplastic, and electricity; and (ii) Complementing the ELCA, pilot-scale WRRF investigations were performed to assess elements of the conceptual Twin Falls WWTP retrofit: bioplastic production on fermented dairy manure, with bioplastic system effluent integrated into an EBPR pilot WRRF.

Assessing resource recovery environmental impacts

Scenario 1a – Base case with no modifications

The purpose of ELCA is to compare and contrast processes in a quantitative manner to assess relative environmental sustainability. Considering the concept of wastewater treatment, research has demonstrated the net environmental benefits can be significant (Lassaux et al., 2007), although not without some adverse impacts (Bisinella de Faria et al., 2015; Coats, Watkins, et al., 2011; Foley et al., 2010); moreover, the target level of treatment relative has been debated (Foley et al., 2010; Lassaux et al., 2007; Lundie et al., 2004; Rahman et al., 2016). Thus, in considering WWTP upgrades and/or WRRF scenarios, impacts are considered relative to a "base case" to help inform the decision-making process.

As noted, the current Twin Falls, Idaho WWTP was established as the "base case," against which all resource recovery scenarios were compared; emissions that transcended all scenarios were excluded from the ELCA. Thus, the "base case" ELCA metrics for many contributions within each category were zero (Table 4). Specific categories for the "base case" that exhibited positive or negative environmental impacts were associated with emissions due to (i) a fertilizer offset from the use of biosolids and (ii) secondary effluent discharged to the Snake River. Nominal positive environmental benefits were observed in all categories associated with use of biosolids, in lieu of DAP, associated with transportation and fertilizer offset. The only negatively impacted categories were those associated with

TABLE 4 Environmental Life Cycle Assessment Results for Each Scenario for (a) Global Warming—Air (kg CO2-eq); (b) Acidification—Air (kg SO2-eq); (c) Eutrophication—Air (kg N-eq); (d) Eutrophication—Water (kg N-eq); (e) Smog—Air (kg O3-eq); (f) Human Health Particulate—Air (kg PM2.5-eq)

	Scenario (Global Warming)						
(a) Contribution	**1a**	**1b**	**1c**	**2a**	**2b**	**3**	**4**
Transportation	−1.38E+00	−1.38E+00	−9.37E−01	−8.18E−01	−8.18E−01	3.32E+02	3.32E+02
Plastics offset	0	0	0	0	0	0	0
Effluent	0	0	0	0	0	0	0
Struvite chemicals	0	0	5.29E+06	5.14E+06	5.14E+06	3.24E+06	0
Electricity change	0	−2.55E+07	0	−8.18E+05	−1.93E+07	−2.21E+07	−2.34E+07
Fertilizer offset	0	0	0	0	0	0	0
Total	−1.38E+00	−2.55E+07	5.29E+06	4.32E+06	−1.41E+07	−1.88E+07	−2.34E+07
	Scenario (Acidification—Air)						
(b)Contribution	**1a**	**1b**	**1c**	**2a**	**2b**	**3**	**4**
Transportation	0	0	0	0	0	0	0
Plastics offset	0	0	0	0	0	0	0
Effluent	0	0	0	0	0	0	0
Struvite chemicals	0	0	2.43E+04	2.37E+04	2.37E+04	1.49E+04	0
Electricity change	0	−1.44E+05	0	−4.63E+03	−1.09E+05	−1.25E+05	−1.32E+05
Fertilizer offset	−7.50E−02	−7.50E−02	−3.04E−01	−3.12E−01	−3.12E−01	−2.55E−01	−8.48E−02
Total	−7.50E−02	−1.44E+05	2.43E+04	1.90E+04	−8.55E+04	−1.10E+05	−1.32E+05
	Scenario (Eutrophication—Air)						
(c) Contribution	**1a**	**1b**	**1c**	**2a**	**2b**	**3**	**4**
Transportation	0	0	0	0	0	0	0
Plastics offset	0	0	0	0	0	0	0
Effluent	6.74E−01	6.74E−01	1.90E−01	1.64E−01	1.64E−01	1.51E−01	8.38E−01
Struvite chemicals	0	0	3.47E+02	3.38E+02	3.38E+02	2.13E+02	0
Electricity change	0	−4.35E+03	0	−1.40E+02	−3.29E+03	−3.77E+03	−3.99E+03
Fertilizer offset	−2.57E−02	−2.57E−02	−1.04E−01	−1.07E−01	−1.07E−01	−8.75E−02	−2.91E−02
Total	6.49E−01	−4.35E+03	3.48E+02	1.98E+02	−2.96E+03	−3.55E+03	−3.99E+03
	Scenario (Eutrophication—Water)						
(d) Contribution	**1a**	**1b**	**1c**	**2a**	**2b**	**3**	**4**
Transportation	0	0	0	0	0	0	0
Plastics offset	0	0	0	0	0	0	0
Effluent	6.54E+00	6.54E+00	3.39E+00	3.22E+00	3.22E+00	3.00E+00	7.49E+00
Struvite chemicals	0	0	2.28E+03	2.22E+03	2.22E+03	1.40E+03	0
Electricity change	0	−2.86E+04	0	−9.17E+02	−2.16E+04	−2.48E+04	−2.62E+04
Fertilizer Offset	−1.68E−01	−1.68E−01	−6.79E−01	−6.98E−01	−6.98E−01	−5.70E−01	−1.90E−01
Total	6.38E+00	−2.86E+04	2.29E+03	1.30E+03	−1.94E+04	−2.34E+04	−2.62E+04
	Scenario (Smog—Air)						
(e) Contribution	**1a**	**1b**	**1c**	**2a**	**2b**	**3**	**4**
Transportation	−4.94E−05	−4.94E−05	−3.37E−05	−2.94E−05	−2.94E−05	1.19E−02	1.19E−02
Plastics offset	0	0	0	0	0	6.86E−03	6.86E−03
Effluent	0	0	0	0	0	0	0

(Continues)

TABLE 4 (Continued)

	Scenario (Smog—Air)						
(e) Contribution	**1a**	**1b**	**1c**	**2a**	**2b**	**3**	**4**
Struvite chemicals	0	0	1.95E+05	1.89E+05	1.89E+05	1.19E+05	0
Electricity change	0	−2.43E+06	0	−7.82E+04	−1.84E+06	−2.11E+06	−2.24E+06
Fertilizer offset	0	0	0	0	0	0	0
Total	−4.94E−05	−2.43E+06	1.95E+05	1.11E+05	−1.65E+06	−1.99E+06	−2.24E+06

	Scenario (HHP—Air)						
(f) Contribution	**1a**	**1b**	**1c**	**2a**	**2b**	**3**	**4**
Transportation	0	0	0	0	0	0	0
Plastics Offset	0	0	0	0	0	0	0
Effluent	0	0	0	0	0	0	0
Struvite chemicals	0	0	1.21E+03	1.17E+03	1.17E+03	7.41E+02	0
Electricity change	0	−5.32E+03	0	−1.71E+02	−4.03E+03	−4.61E+03	−4.89E+03
Fertilizer offset	−1.14E−02	−1.14E−02	−4.61E−02	−4.73E−02	−4.73E−02	−3.86E−02	−1.29E−02
Total	−1.14E−02	−5.32E+03	1.21E+03	1.00E+03	−2.86E+03	−3.87E+03	−4.89E+03

eutrophication, driven by the discharge of phosphorus to the Snake River.

Scenario 1b – Base case with CHP

Significant energy potential exists in wastewater (CHP/EPA, 2011; Fisher, Rogers, et al., 2015) and as a result the concept of energy-neutral WRRFs has gained much attention (Fisher, Donnelly, et al., 2015; Fisher, Rogers, et al., 2015). Indeed, implementing combined heat and power at a WRRF can substantially offset energy demands (Kehrein et al., 2020); moreover, recognizing that the AD substrate is biogenic in origin and that most publicly produced power includes fossil fuel usage, use of CHP can significantly reduce WRRF GHG emissions (EPA, 2013). CHP is a logical resource recovery add-on to the Twin Falls WWTP: AD is currently employed and the heat generated could replace existing boilers. Analyses indicated that integrating CHP could offset WRRF electrical demands by ~60% (Table 3). More critically, adding CHP shows promising results in all ELCA categories (Figure 2; Table 4). Although Idaho Power maintains a significant portion of its power production in the form of the clean hydropower, coal and natural gas combined account for roughly 30%, both of which are associated with higher amounts of carbon dioxide, nitrogen oxides, and sulfur dioxide emissions; with CHP reducing external power demands by ~60%, Global Warming Potential was reduced by 2.55×10^{-7} kg CO_2 equivalent (Table 4a); results are consistent with those observed by Morelli et al. (2020).

Scenario 1c – Base case with struvite production

Municipal wastewater contains approximately 15%–17% of mined phosphorus, and nearly all consumed P is excreted as waste (Kehrein et al., 2020; Mayer et al., 2016). Phosphorus capture from wastewater is arguably an imminent and critical need in order to both mitigate accelerated eutrophication of surface water and to stem losses to the water environment that will create significant future challenges for recovery. Wastewater P can be readily recovered from nutrient concentrated streams as struvite; while not necessarily the most optimal product (Kehrein et al., 2020; Mayer et al., 2016), at this time struvite production is one of the most commercially viable and mature technologies for large scale wastewater phosphorus recovery (Le Corre et al., 2009). Struvite production at the Twin Falls WWTP makes sense because (i) AD and biosolids dewatering is employed, thus a N/P-rich sidestream is available, (ii) agriculture dominates the region, thus providing outlets for the produced fertilizer, and (iii) a future NPDES limit on P is expected. Modeling results indicate Twin Falls could produce relatively significant quantities of struvite (Table 3). Moreover, integrating struvite production to capture internally recycled P removes approximately 55% of the average influent P; with reduced influent P, modeling of the existing VIP process indicates that improved EBPR can be realized, with effluent P decreasing by 84% relative to the "base case."

While struvite production metrics and impact on effluent are encouraging, applying ELCA to struvite production reveals a potential environmental conundrum. Although the discharge of less P to the Snake River reduces the effluent

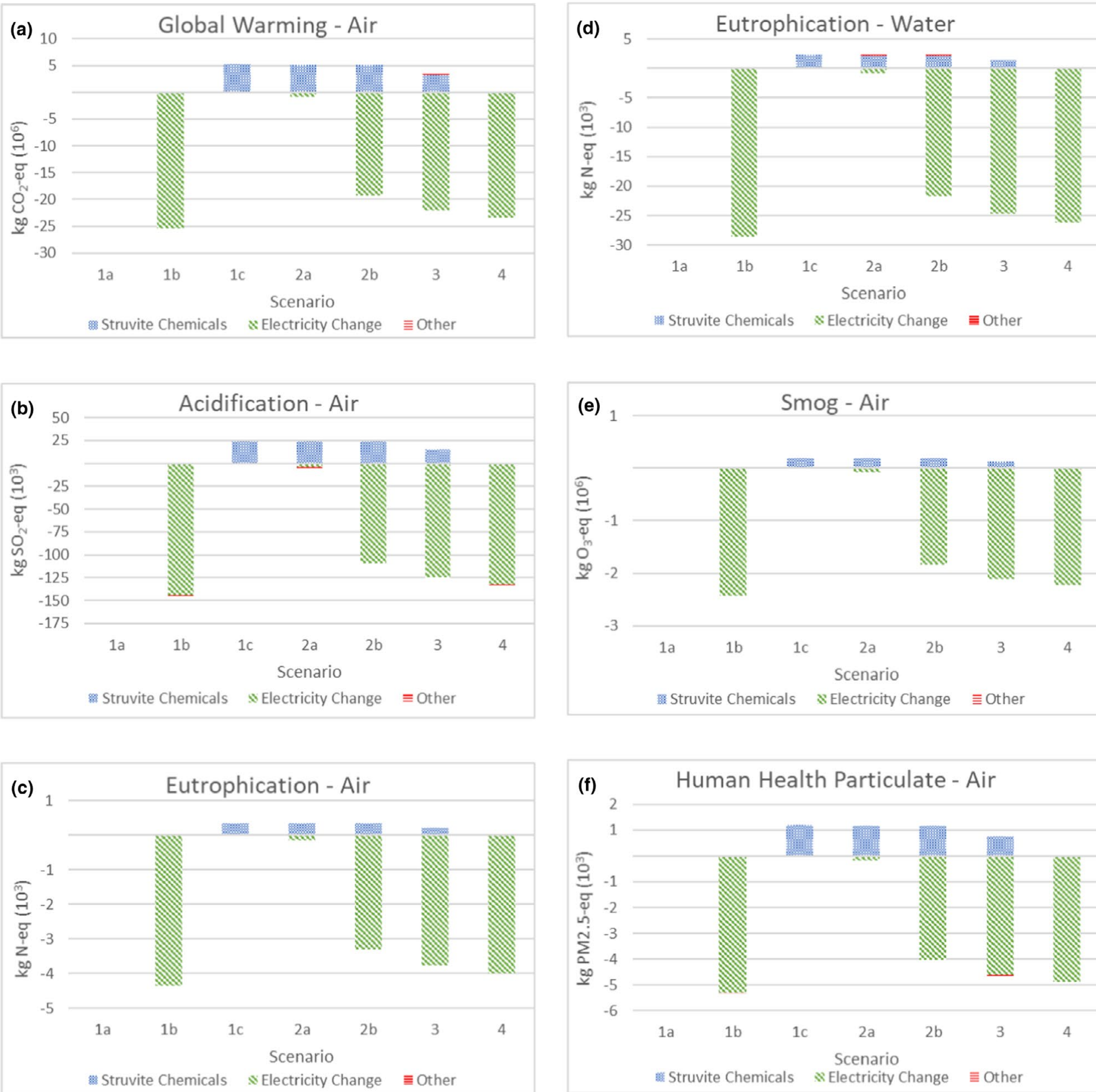

FIGURE 2 Environmental life cycle results for each WRRF scenario in six categories: global warming—air (a), acidification—air (b), eutrophication—air (c), eutrophication—water (d), smog—air (e), and human health particulate—air (f). All values shown are relative to the base case, Scenario #1

eutrophication potential relative to the "base case," overall the addition of struvite crystallization exhibits greater negative environmental impacts (Figure 2; Table 4). In particular, Global Warming Potential shifted from −1.38 kg CO_2 equivalent for the "base case" to +5.29*106 kg CO_2 equivalent under this scenario. Production and transportation of raw chemicals for struvite production are particularly environmentally taxing due to energy and material requirements; transportation effects are only modestly offset by the reduction in synthetic fertilizer transport. Considering some recent works, while ELCA procedures and process configurations varied, results on integrating struvite production have generally revealed negative (Bisinella de Faria et al., 2015; Ishii & Boyer, 2015; Wang, Daigger, et al., 2018) or potentially neutral (Wang, Daigger, et al., 2018) environmental consequences.

Scenario 2a – Addition of EBPR with struvite production

In theory, struvite production should be further enhanced when integrated with the A2O process (Kehrein et al., 2020), as this EBPR configuration exhibits greater potential for P sequestration vs. the existing VIP process. While results showed that this modified configuration did reduce effluent P by ~40% (Table 3), struvite production did not increase with the addition of EBPR (Table 2). Instead, biosolids production increased by ~5%; as modeled, more of the P-rich PAOs remained in the biosolids and were not lysed via AD. Commensurate with reduced struvite production, results indicate that improved EBPR reduced the environmental impact of this alternative relative to scenario 1c across all categories (Figure 2; Table 4); for example, Global Warming Potential impacts were reduced by 18%, and Eutrophication-Water impacts were decreased by 43%. Overall, results align with prior work demonstrating the relative environmental value of EBPR (Coats, Watkins, et al., 2011).

Scenario 2b – EBPR with struvite production and CHP

Considering the contrasting environmental impacts for adding CHP and struvite to the WWTP, the two resource recovery technologies were paired; as expected based on the scenarios 1 and 2a evaluations, results show that the combination made a significant difference on the overall WRRF environmental impact (Figure 2; Table 4). Relative to scenario 2a, Global Warming Potential was decreased approximately 430% and Eutrophication-Water was decreased almost 1600%; in fact, for all categories assessed, the relative emissions shifted from a negative environmental impact to positive. While primary solids fermentation to drive EBPR reduces the quantity of substrate fed to the AD, thereby reducing digester biogas output and electricity production for scenario 1b vs. 2b by 27%, from an environmental perspective, CHP can nonetheless be employed as a mechanism to environmentally justify struvite production. However, this comparative assessment does demonstrate the challenges in recovering wastewater carbon; since fermented carbon (i.e., VFAs) is ultimately catabolized to CO_2 in the EBPR process, some of the influent carbon is utilized for either nutrient removal or energy production, but not both. The subject of carbon management is widely discussed within the literature; carbon is necessary to support biological removal of N and P, and this oxidation of carbon prevents energy recovery from being maximized (Jimenez et al., 2015; Sancho et al., 2019). Nevertheless, scenario 2b performs significantly better than 2a in all categories due to the addition of CHP.

Scenario 3 – EBPR with struvite production, CHP, and PHA production

The lone remaining "resource" yet to be examined in this study—production of PHA—was considered in scenario 3; PHA production would be achieved through use of fermented dairy manure (Guho et al., 2020), with residual solids processed via anaerobic digestion to enhance CHP (Stowe et al., 2015). As noted, wastewater carbon usage is typically one-dimensional when competing processes are involved; importing a carbon-rich substrate was necessary to avoid diversion of primary solids-derived VFAs from EBPR. Dairy manure has been demonstrated to be an excellent substrate for PHA production using mixed microbial consortia (Coats et al., 2016; Guho et al., 2020). As illustrated (Figure 2), this scenario achieves better overall environmental performance than the comparative scenario 2 in all categories, with excellent effluent quality; Global Warming Potential and Eutrophication-Water impacts were reduced 33%–535% and 20%–1900%, respectively. Although additional P is integrated into the WRRF associated with dairy manure fermentation, similar to scenario 2 the process modeling indicates that much of the additional P will remain in the biosolids vs. struvite (Table 3). While the ELCA metrics indicate that this more comprehensive WRRF scenario does not outperform the simple addition of CHP to the "base case" (Figure 2; Table 4), the net environmental gain is nonetheless positive. Moreover, this alternative yields a diversity of products from wastewater, thereby truly realizing the WRRF concept. Ultimately, the true value of integrating PHA production may be realized economically.

Scenario 4 – CHP and PHA production

As a final comparative analysis, scenario 4 was developed to focus exclusively on maximizing carbon capture from the influent wastewater through combined PHA production and energy generation via CHP. Without a focus on phosphorus recovery, scenario 4 achieved high environmental performance in several categories (Figure 2; Table 4). While the additional phosphorus added to the WRRF through the importation of dairy manure results in a 37% increase in effluent P vs. the base case (Table 3), largely due to the use of CHP, the overall Eutrophication-Water impact is reduced by over 1500 times and also shows a positive environmental impact (Table 4d). Similar to other CHP scenarios, Figure 2 illustrates that all the environmental benefits for scenario 4 arise from the decrease in required electricity from the grid. Overall, effluent constituents in the quantities discharged to the water environment have negligible impacts on all categories when compared to changes in electricity demand and use of chemicals for the struvite process. Comparatively, scenario 4 performs very similarly to scenario 1b, which employed only CHP with the base case. Considering that results herein indicate that EBPR can enhance P recovery in biosolids, even without struvite production, it is reasonable to predict that integrating EBPR would be a favorable resource recovery add-on process.

ELCA sensitivity analysis

Inputs for the base results were static and steady state, and reflected specific operational conditions; sensitivity analyses are an important element in conducting and assessing ELCAs. The first parameter tested was electricity source, performed in a similar manner to Guven et al. (2018). While Idaho Power has a relatively clean footprint of energy fuel sources, it maintains approximately 25% fossil fuel sources. In 2019, Idaho Power announced its plan to achieve 100-percent clean energy by 2045 (Idaho Power, 2019); this change in power source was applied to the ELCA. Given that the base-case analyses revealed significant environmental value by internally producing renewable power to offset demand from the utility, results were expected to decrease both the positive effects of CHP and the negative effects of increased power usage, while giving the other contributing factors a larger input in environmental effects. Indeed, results suggest that this is the case; Figure S8 illustrates that the benefits of internally generating electricity (CHP) are universally lessened, while the effects of struvite chemicals become the predominant environmental impact. The largely minimal "other" category remained nominal relative to the impacts of electricity and struvite chemicals. Scenarios 1b and 4 exhibit the overall best environmental footprint. Ultimately, these results reveal the importance of renewable energy sources, whether produced internally via CHP or externally by the utility; while results within the system boundary commensurately magnify the relative negative environmental impacts of struvite production, in reality, the net balance outside the system boundary likely remains similar to scenarios 1b, 2b, 3, and 4.

The second sensitivity analysis focused on improving struvite environmental metrics. Magnesium hydroxide production exhibited the largest negative effect, with emissions all due to fossil fuel-based energy production. Assuming that all energy for magnesium hydroxide production was renewable, and that the emissions associated with the production of sodium hydroxide estimated by Thannimalay (2013) were decreased by 10%, as illustrated in Figure S9, the negative environmental impacts associated with struvite production are nearly eliminated. Ultimately, the use of renewable energy for struvite chemical production results in scenarios 1b, 2b, 3, and 4 exhibiting strong environmental footprints.

Pilot-scale WRRF evaluations: Integrating bioplastics production and EBPR

Combined heat and power and struvite production at a WRRF are mature and proven technologies; as such, pilot-scale testing will not reveal operational issues that have not already been documented. Moreover, the positive environmental impacts of shifting to a renewable energy portfolio were clearly revealed in the ELCA results. However, the WRRF concept should be expanded beyond a focus on energy neutrality and phosphorus capture—but not at the expense of deteriorated wastewater treatment. In this regard, while the concept of integrating bioplastics production within wastewater treatment has been investigated at a pilot scale (Bengtsson et al., 2017; Conca et al., 2020; Crutchik et al., 2020), investigations have focused on the feasibility and economics of PHA production and not associated impacts on wastewater treatment and effluent.

Building from the modeling underlying the ELCA results associated with scenarios 3 and 4, a dairy PHA pilot (Guho et al., 2020) was integrated with a post-anoxic EBPR pilot to assess the potential implications on effluent quality. Specifically, effluent from the PHA production reactor was blended with the raw wastewater pumped into the EBPR pilot; compared to the ELCA analyses with dairy manure (waste from 5000 cows added to a 0.35 m^3/s WRRF), the EBPR pilot was equivalently loaded at 3500–4000 cows. As shown (Table 5), influent ammonia-N and phosphorus concentrations before, during, and after the addition of PHA effluent were relatively similar. More importantly, treatment performance was comparable. Effluent ammonia-N before PHA effluent addition was somewhat impaired due to ABAC process troubleshooting. Overall ammonia-N removal was 94%, 98%, and 99%, respectively. Ammonia-N concentrations across the system similarly exhibited consistent behavior with and without PHA effluent (Figure 3a).

Phosphorus removal in the scale model averaged 91%, 94%, and 89% before, during, and after the addition of PHA effluent, respectively; effluent concentrations were excellent, with the lowest concentration realized with the addition of PHA effluent (Table 5). Phosphorus cycling was consistent with EBPR theory (Figure 3b). Indicators of EBPR performance include the influent VFA:P ratio and anaerobic P release to VFA uptake (P:C ratio); average influent VFA:P ratios were 13.5, 9.9, and 8.3 $mgVFA_{COD}$:mgP, respectively, while P:C ratios (Pmol:Cmol) are shown in Figure 3b. Note that influent VFAs and P were both measured in the fermenter effluent. The VFA:P ratios were low relative to "optimum" for EBPR (Coats et al., 2017; Tchobanoglous et al., 2014); however, the P:C ratios were significant and indicative of an effective EBPR metabolic response (Coats et al., 2017). Of note, the P:C ratio increased with the addition of PHA effluent (Figure 3b), despite the lower VFA:P ratio.

Data interpretation and discussion: ELCA, sustainability pre-conceptions, and wastewater resource recovery

Environmental life cycle assessment studies are useful as we seek to better understand anthropogenic impacts to the

TABLE 5 Influent and effluent phosphorus and ammonia-nitrogen concentrations for the pilot-scale enhanced biological phosphorus removal system before (pre-PHA (polyhydroxyalkanoate)), during (with PHA effl), and after (post-PHA) the addition of effluent from a dairy manure PHA production system

	Concentration, mg/L	SD, mg/L	*n*
Phosphorus			
Influent			
Pre-PHA	4.85	1.22	22
With PHA effl	4.47	1.22	9
Post-PHA	4.33	0.69	14
Effluent			
Pre-PHA	0.39	0.55	14
With PHA effl	0.28	0.08	9
Post-PHA	0.49	0.21	14
Ammonia			
Influent			
Pre-PHA	38.05	5.62	21
With PHA effl	34.95	4.23	9
Post-PHA	38.95	10.62	14
Effluent			
Pre-PHA	2.15	1.54	9
With PHA effl	0.70	0.28	9
Post-PHA	0.34	0.28	14

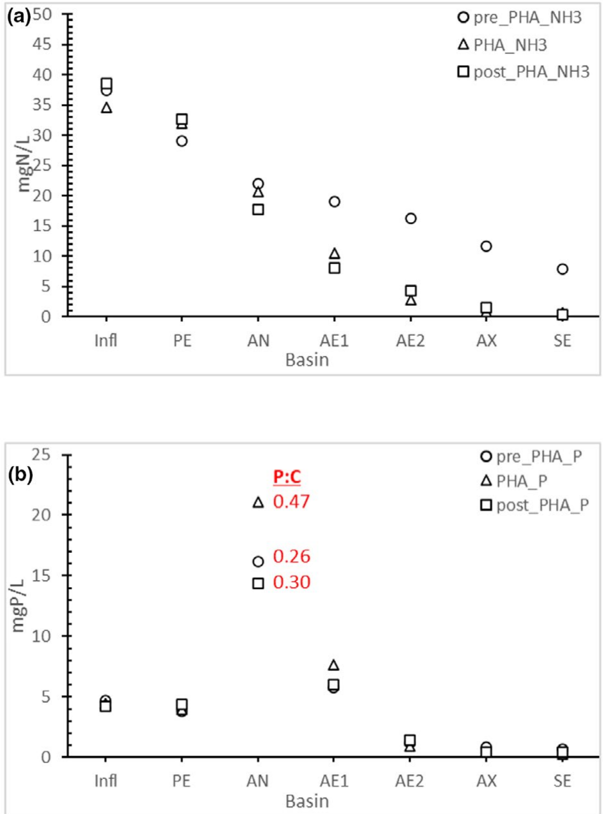

FIGURE 3 Performance of the EBPR pilot WRRF with (denoted "PHA NH3) and without (denoted pre_PHA_NH3 and post_PHA_NH3) the addition of effluent from the polyhydroxyalkanoate (PHA) pilot system (infl: raw wastewater; PE: effluent from the primary solids fermenter; AN: anaerobic basin; AE1, AE2: aerobic basins 1 and 2; AX: anoxic ditch; SE: secondary effluent). (a) Ammonia-N profile across the WRRF, and (b) Soluble Reactive Phosphorus profile across the WRRF (P:C is the mass of anaerobic phosphorus released to mass of VFAs consumed anaerobically, Pmol:Cmol)

natural environment and how to make better overall decisions. Moreover, ELCA studies—being quantitative—help dispel myths and pre-conceived notions regarding the concept of "sustainability." In this study, one hypothesis was that energy use—and production—at the Twin Falls WRRF would not be impactful, given that Idaho Power operates with high levels of hydropower and renewable energy. However, results indicate that electricity production, via CHP, results in far greater environmental impacts than focusing on effluent quality or resource production alone. Hao et al. (2019) similarly affirmed the value of energy recovery at a WRRF, relative to a base-case WWTP, albeit through thermal energy recovery and not CHP.

The positive effects of renewable energy were secondarily observed in the ELCA assessment of struvite production. It was hypothesized that WRRF elements that captured P would yield measurable impacts on eutrophication potential; while effluent P can be significantly reduced by employing struvite production and EBPR, ultimately energy sources induced the greatest impact. Indeed, the largest contributor to eutrophication was nitrogen oxides associated with fossil fuel-based energy sources. Similar results were observed by Rahman et al. (2016), in that, environmentally positive local effects due to improved effluent quality can translate into global negative environmental impacts associated with the increased demand for energy and chemicals.

Environmental life cycle assessment investigations are necessarily subjective, as they must be place- and/or product-based and actualize specifics of a process/product. Indeed, Lam et al. (2020) noted that ELCA studies are intrinsically unique and results can be difficult to compare across studies; this is particularly true for ELCA of wastewater struvite production, where results are presented relative to other nutrient removal alternatives or against other struvite production mechanisms (i.e., assessing struvite production as "less bad" environmentally, comparatively). However, ELCA does not capture the opportunity cost of phosphorus recovery from wastewater. Global phosphorus reserves are dwindling; phosphorus capture and recovery from wastewater can help close the anthropogenic phosphorus cycle. The impact analysis employed herein, while beneficial in comparing/contrasting environmental impacts associated with different processes/

products, ultimately does not account for the limited global supply of phosphorus, nor is the ELCA model set up to account for a future where P must be recovered from diffused water environment sources. Thus, it is important to recognize that while P recovery may create additional environmental emissions, this cost will likely be necessary to maintain phosphorus as a global resource. Additionally, capturing this scarce resource can mitigate environmental impacts associated with raw resource development (Kehrein et al., 2020; Mayer et al., 2016). Struvite economics are similarly conflicted; struvite alone as a WRRF add-on has also been shown to be economically disadvantageous as a commercial product, yet conversely economically beneficial in reducing WRRF operational costs associated with struvite scaling (Kehrein et al., 2020).

CONCLUSIONS

Research was conducted applying ELCA to assess the sustainability of converting an existing municipal WWTP to a WRRF, integrating dairy manure, to capture carbon for energy and bioplastic production while also recovering phosphorus for agronomic uses and producing reclaimed water in conformance with a discharge permit. Results demonstrate that efficient use of carbon is paramount in operating WRRFs in the most environmentally responsible manner. The production of electricity via CHP was the single most important resource to achieve a sustainable WRRF; energy offset potential at the WRRF ranged from 43% to 60%, and the environmental footprint was commensurately shifted such that WRRF operations had no negative impacts. Capturing wastewater phosphorus is important in the future portfolio for this critical macronutrient. Struvite production helps maintain a resilient EBPR process; struvite and biosolids usage was shown to offset a significant quantity of synthetic fertilizer, suggesting that sustainably sourced fertilizers make a difference environmentally. While struvite production exhibits a negative environmental impact—principally associated with chemical production and associated emissions from fossil fuel-based energy demands—integration with CHP negates the adverse environmental consequences. Integrating organic-rich industrial waste—in this case, dairy manure—expands the resource recovery portfolio to include bioplastics while capturing more phosphorus; the diversion of carbon from CHP to bioplastics production and EBPR does reduce energy output, but the WRRF nevertheless realizes a sustainable footprint for all categories evaluated. Recognizing the risk that might be perceived in integrating an additional organic waste load to the WRRF, pilot evaluations demonstrated that EBPR was not adversely affected by the addition of effluent from PHA production. Collectively, research demonstrates that a WRRF integrating dairy manure can yield a diverse portfolio of products while operating in an environmentally benign—sustainable—manner.

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SUPPORTING INFORMATION

Additional supporting information may be found online in the Supporting Information section.

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