Life Cycle Environmental and Cost Impacts of Using an Algal Turf Scrubber to Treat Dairy Wastewater

Brendan T. Higgins and Alissa Kendall

Keywords

algae
anaerobic digestion
bioenergy
industrial ecology
life cycle assessment (LCA)
nutrient trading

Supporting information is available on the *JIE* Web site

Summary

Using algae to simultaneously treat wastewater and produce energy products has potential environmental and economic benefits. This study evaluates the life cycle energy, greenhouse gas (GHG) emissions, eutrophication potential, and cost impacts of incorporating an algal turf scrubber (ATS) into a treatment process for dairy wastewater. A life cycle inventory and cost model was developed to simulate an ATS treatment system where harvested algae would be used to generate biogas for process heat and electricity generation.

Modeling results show that using an ATS significantly reduces eutrophication impacts by reducing chemical oxygen demand, nitrogen, and phosphorus in the wastewater. With low water recirculation rates through the ATS and high algae productivity, inclusion of the ATS results in net energy displacement and a reduction of GHG emissions compared to a system with no ATS. However, if high water recirculation rates are used or if algae biosolids from the digester are dried, the system results in a net increase in energy consumption and GHG emissions.

The life cycle treatment cost was estimated to be $1.42 USD per cubic meter of treated wastewater. At this cost, using an ATS would only be cost effective for dairies if they received monetary credits for improved water quality on the order of $3.83 per kilogram of nitrogen and $9.57 per kilogram of phosphorus through, for example, nutrient trading programs.

Introduction

Treating wastewater through algae cultivation could reduce greenhouse gas (GHG) emissions and eutrophication, and simultaneously generate cost-competitive renewable fuels and electricity (Lundquist 2010). Experimental research indicates that algal treatment systems may be particularly well suited for treating wastewater from livestock and dairy operations (Kebede-Westhead et al. 2003; Mulbry et al. 2008; Woertz et al. 2009). While algae-based biofuels have garnered significant attention recently (DOE 2010a), biogas is an alternative energy product obtained from algae systems. Anaerobic digestion of algae can yield biogas and subsequent electricity generation, along with nutrient rich solids that can be applied as fertilizer. Such systems could have positive synergy with livestock wastewater treatment because biogas can be generated from both wastewater and digestion of algal biomass.

Despite the potential advantages of algal wastewater treatment, limited research has been done to quantify its environmental and cost impacts. Clarens and colleagues (2010) conducted a life cycle assessment (LCA) on the use of algae grown in raceway ponds for the production of biomass energy. They

Address correspondence to: Alissa Kendall, Department of Civil and Environmental Engineering, University of California, Davis, One Shields Ave., Davis, CA 95616. *Email:* amkendall@ucdavis.edu

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developed scenarios in which algae cultivation on chemical fertilizer was compared to cultivation on municipal wastewater effluent, and found that algae grown on wastewater reduced the energy, GHG, and eutrophication impacts of the process. Several other LCAs have also been carried out on the production of algal biofuel, but none of these studies examined algae grown on wastewater (Batan et al. 2010; Lardon et al. 2009; Luo et al. 2010).

In addition to using raceway ponds for algae cultivation, algal turf scrubbers (ATSs) have been used to provide tertiary treatment of both municipal and agricultural wastewaters (Green et al. 1996; Kebede-Westhead et al. 2003; Craggs et al. 1996b). An ATS is a long, inclined flow-way that supports a biofilm of algae and bacteria. Wastewater washes over the flow-way in a series of bursts and provides the biofilm with nutrients (Craggs et al. 1996a). The algae remove inorganic nutrients, including nitrogen and phosphorus, from the wastewater, thereby reducing downstream eutrophication impacts.

The nutrients are removed from the system when the algal biomass (5% to 6% solids) is harvested by scraping (Johnson and Wen 2009; Pizarro et al. 2006). Nutrient removal reduces eutrophication in natural waterways, mitigating eutrophication impacts, which cause ecosystem disruption and an estimated USD $2.2 billion in economic damage in the United States (Dodds et al. 2009).

Traditional dairy wastewater treatment often relies on anaerobic lagoons for dairy wastewater management. These have very high nutrient effluent content (about 2,000 milligrams per liter of total Kjehldahl nitrogen [mg/L TKN][1] and about 500 mg/L of phosphorus) (Mukhtar et al. 2004). These systems are not intended for high-quality treatment, but rather limit runoff into surface waters.

Previous research on ATS systems has focused on their effectiveness in treating wastewater and the economics of the process, but has not applied life cycle methods to quantify environmental impacts. Kebede-Westhead and colleagues (2003) showed that ATS systems can effectively treat dairy wastewater in lab-scale studies. HydroMentia Inc. has also installed ATS systems to treat agricultural runoff at commercial scales. Pizarro and colleagues (2006) conducted an economic assessment of an ATS fed with dairy wastewater, with the resulting algal biomass applied to fields as a soil amendment.

No research has been published, however, on the life cycle environmental impacts of using ATS technology to treat wastewater or produce bioenergy. Algal biomass generated by ATSs has a low fatty acid content and is therefore poorly suited for biodiesel production (Mulbry et al. 2008). Thus, in this study, algal biomass is anaerobically digested to produce biogas that is combusted for electricity generation.

Goal and Scope

In this study we apply LCA to the treatment of flushed dairy wastewater using an anaerobic digester for secondary treatment and an ATS for tertiary treatment. The purpose of this research is to estimate the cost and environmental impacts of including an ATS in the treatment system. The treated wastewater is the primary service provided by the system, while electricity and biosolids are coproducts. The ATS system will reduce eutrophication impacts; however, utilization of the ATS system may increase other environmental and cost impacts of wastewater management. These trade-offs will be calculated using the LCA and cost estimation functions of the model described in this article, and the most influential design parameters will be identified. In addition, modeling outcomes will be contextualized in environmental policy scenarios that could make the system cost effective for dairies.

The intended audience for this study includes dairy operators, researchers, and regulatory authorities who are interested in treatment strategies for dairy effluent. Due to the novelty of algae technologies, no dairies currently employ commercial-scale ATS systems. However, ATSs are being studied as a potential treatment system for dairies, and life cycle impacts should be considered prior to adoption.

The modeled treatment process is shown in figure 1. The functional unit is one cubic meter (m^3) of screened, flushed dairy wastewater entering the process.[2] Although the volume of wastewater can be correlated to the number of animal units, this was not done because manure flush rates vary among dairies. Moreover, because there are interactions between the secondary and tertiary treatment phases, modeling the ATS alone is not ideal. Instead, the primary and secondary treatment processes were modeled with and without the ATS system in order to determine its effect on life cycle performance.

As a result, the effluent qualities of the compared systems are not identical, and thus the function performed by the two systems is not equal. This deviates from typical comparative LCA methodology in which comparable processes should have identical functions. In the present study, the function of interest is the treatment of wastewater entering the process. The varying effluent quality of alternative technologies is captured in the eutrophication impact category and used to calculate the cost of abatement for eutrophying nutrients.

Screening is used to separate solids in the primary treatment phase and these solids can be applied to agricultural fields as a soil amendment. Operation of the dairy, primary treatment, and field application of screened solids were not included in the life cycle model as these activities would take place regardless of the downstream treatment process.

Anaerobic digestion was chosen for secondary treatment because it reduces chemical oxygen demand (COD) while also producing biogas that can be combusted for energy. The ATS provides tertiary wastewater treatment that results in the removal of inorganic nutrients and residual COD. Algal biomass harvested from the ATS is then anaerobically digested to produce additional biogas and biosolids. The biogas is combusted to generate electricity and steam for process heat and the biosolids are applied to agricultural fields.

The impact categories analyzed are total primary energy, global warming potential (GWP), eutrophication potential, and cost. Environmental impacts and costs were included for facility construction and operation, but end of life was excluded

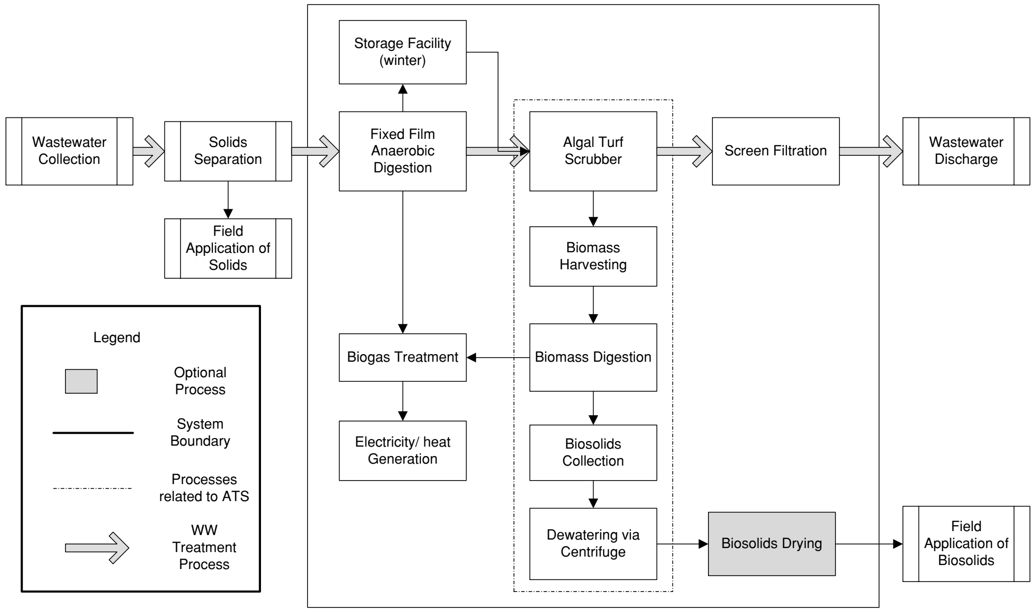

Figure 1 Flow chart of the treatment process including system boundary. ATS = algal turf scrubber; WW = wastewater.

due to uncertainty about the decommissioning of treatment plants (Renou et al. 2008).

Full construction costs were included, but environmental impacts resulting from construction were limited to the manufacture of concrete, steel, plastic liners, and process equipment. Environmental impacts from on-site construction activities are difficult to quantify (Renou et al. 2008), but are likely to be less than 1% of total impacts based on previous LCAs of municipal wastewater treatment systems (Emmerson et al. 1995). Although the entire facility is expected to operate for 40 years, plastic liners are expected to be replaced after 20 years and mechanical equipment every 10 years.

Wastewater production and treatment operations are likely to vary by season. Average annual values were assigned to parameters and used in mass and energy balance calculations. Maximum parameter values were used to ensure adequate equipment sizing.

The baseline scenario in the model was developed using average or expected performance characteristics. Alternative scenarios were also developed by varying process parameters that were found to significantly influence the results. These alternative scenarios were designed to test worse-than-expected performance. Also, three policy scenarios were developed in order to assess system performance under different policies that target eutrophication, GHG emissions, and renewable energy generation.

Methods

A spreadsheet-based life cycle model was developed for the treatment system using data from lab- and pilot-scale studies (Kebede-Westhead et al. 2003, 2004; Mulbry et al. 2005, 2008; Wilkie 2000). Mass and energy balances were developed from the empirical data and used to develop most life cycle inventory data. Economic input-output LCA data were used to estimate inventories for process equipment (Green Design Institute 2002).

Fixed-film Anaerobic Digestion of Wastewater

The modeled facility treats 1,066 m^3/day of flushed wastewater, the approximate wastewater flow rate from a 750-cow dairy (Wilkie et al. 2004). A schematic of the modeled wastewater treatment system is shown in figure 2. Additional data on wastewater characteristics and process components can be found in the supporting information available on the Journal's Web site.

After solids screening, the flushed dairy wastewater passes into the fixed-film digester. Data for this digester were obtained from the University of Florida's Dairy Research Unit (DRU) (Wilkie et al. 2004). The high bacterial accumulation in fixed-film digesters allows for a low hydraulic residence time (HRT) of 3.75 days and ambient operating temperatures (27°C) (Wilkie et al. 2004). The model requires digester heating if ambient temperatures fall below 27°C, as temperature-dependent data were not available.

The DRU digester removed 48% of COD, and biogas was produced at levels of 0.97 m^3 of biogas/m^3 of wastewater. The DRU biogas was composed of 80% methane and 20% carbon dioxide (CO_2) (Wilkie 2000). In the baseline scenario, leakage and periodic maintenance of the digester's interior account for direct biogas emissions at a rate of 2% of total biogas (Moller et al. 2009).

The walls and roof of the digester are reinforced concrete coated with 0.5 inch polyurethane insulation, with heat transfer coefficients of 1.38 watts per square meter per degree Celsius ($W/m^2/°C$)[3] for the walls and 1.49 $W/m^2/°C$ for the roof (EPA

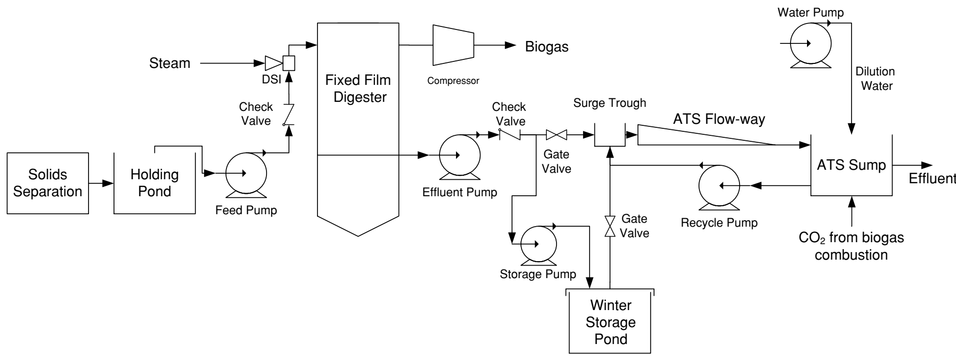

Figure 2 Schematic of wastewater treatment process. DSI = direct steam injection.

1979). Influent water temperature from the barn and milking parlor was assumed to be 24°C (Wilkie et al. 2004), and direct steam injection is used to maintain the digester temperature (equation 1):

$$\dot{m}_{st}h_{st} = \dot{m}_w c_{p,w}(T_D - T_w) + U_{D,i}A_{D,i}(T_D - T_a) \quad (1)$$

where

$\dot{m}_{st}$ = mass flow rate of steam;
h_{st} = enthalpy of steam;
$\dot{m}_w$ = mass flow rate of wastewater;
$c_{p,w}$ = heat capacity of wastewater;
T_D = digester temperature (27°C);
T_w = temperature of influent wastewater;
T_a = atmospheric temperature;
$U_{D,I}$ = digester heat transfer coefficients distinguished between walls and roof; and
$A_{D,I}$ = digester surface area exposed.

Sludge is not recovered from the DRU digester and is therefore not included in the model for the fixed-film digester. Fixed-film digesters are only compatible with wastewaters containing less than 1% solids. In this application, the wastewater solids concentration averages 0.36% by mass and the flow properties are assumed to be comparable to those of water.

Pipe sizes were selected to maintain flow velocities in the range of 1 to 3 meters per second (m/s).[4] The cost of the DRU fixed-film digester was obtained from the U.S. Environmental Protection Agency (EPA 2003) and scaled for inflation using the Engineering News Record (ENR) construction cost index. A scaling factor of 0.6 was used, appropriate for farm-scale digesters (AgStar 2010; Ghafoori and Flynn 2006).

Algal Turf Scrubber

The ATS flow-way surface is typically constructed using a landfill liner, and a concrete sump is constructed at the base of the ATS (Mulbry et al. 2008). Experiments have demonstrated that an ATS removes 74% to 95% of COD, 47% to 92% of nitrogen, and 64% to 96% of phosphorus from flushed dairy wastewater (Kebede-Westhead et al. 2003; Mulbry and Wilkie 2001). Those same studies have shown that loading the ATS with undiluted flushed dairy wastewater can result in poor growth performance. Wastewater was diluted by a factor of 3.5:1 to achieve stable algae growth, consistent with the methods of Kebede-Westhead and colleagues (2003).

Water consumption was not quantified in this analysis, but could present a significant environmental impact, particularly in water-scarce regions. Most pilot ATS systems are located in the southeastern United States, where water scarcity is less of an issue than in many other U.S. regions. However, the ATS may be feasible in dryer regions if the effluent can be used for irrigation.

The biomass productivity of the ATS directly affects facility size: higher productivity yields a smaller facility with equivalent nutrient removal. The baseline productivity was assumed to be 19 grams per square meter per day (g/m^2/day) for ten months of the year.[5] This figure is based on an average value observed by Kebede-Westhead and colleagues (2003) in lab-scale studies. The corresponding ATS requires 18 hectares (ha)[6] of land (100 m × 1,800 m). Productivity of 4 g/m^2/day was assumed for two months during the winter, which is consistent with winter productivity observed by Craggs and colleagues (1996a) when operating a 991 m^2 pilot-scale ATS on municipal wastewater. Craggs and colleagues (1996a) observed nonwinter productivities in the range of 10 to 25 g/m^2/day. Mulbry and colleagues (2008) observed 5.3 to 14.6 g/m^2/day for a pilot-scale ATS operating on various livestock wastewaters over the course of a three-year period. These pilot studies suggest that productivity is subject to considerable uncertainty. Average annual productivity has a significant impact on environmental performance and is explored in the scenario analysis.

Capital costs for the ATS were computed on a per hectare basis using estimates from Pizarro and colleagues (2006). These costs were adjusted with a 0.6 scaling factor. A design factor of 8% and a 7% construction overhead factor were added (Peters et al. 2003). More detailed information on costs can be found in the supporting information on the Web.

Harvesting

Harvesting of the algal biomass is performed by a diesel tractor equipped with a scraping attachment. This is the harvesting method employed at HydroMentia's commercial-scale ATS in

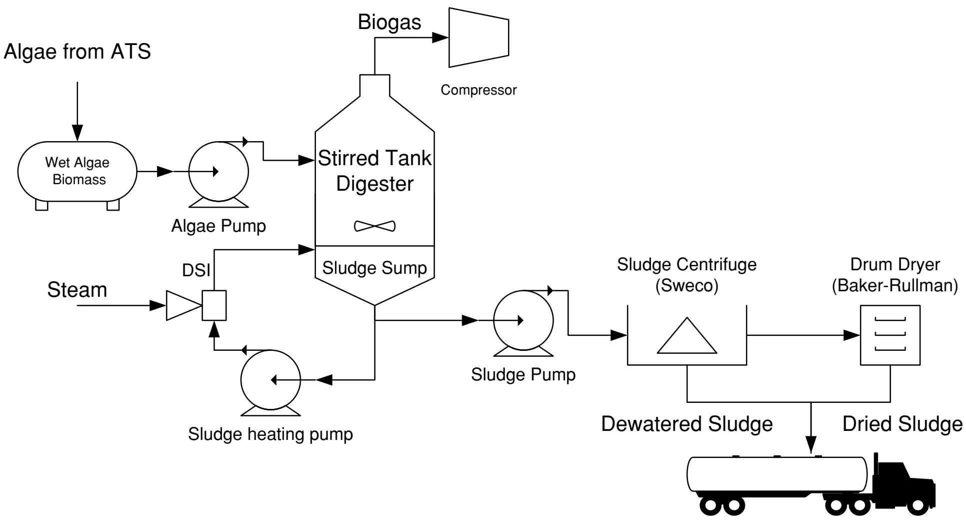

Figure 3 Algae processing schematic. DSI = direct steam injection.

Florida, which is used to treat agricultural runoff (HydroMentia 2011). Tractor fuel consumption was estimated using statistical models published by the American Society of Agricultural Engineers (ASAE 2003) and the calculations are shown in the supporting information on the Web. The annual hours of tractor operation were determined by assuming that algae must be harvested once wet biomass levels reach 12.7 metric tons per hectare (t/ha),[7] consistent with harvesting practice at HydroMentia (McCully 2009).

HydroMentia's operation has also shown that one hour of labor is required per acre[8] harvested. Operating cost was estimated based on the hours of operation with a labor cost of \$15/hour and an annual equipment maintenance cost of 3% of the capital cost (Pizarro et al. 2006). The tractor capital cost was determined from vendor quotes (Deere 2011).

Stirred Tank Anaerobic Digestion of Algae

After harvesting, the algal biomass (5% to 6% solids) is transferred to a heated and stirred anaerobic digester (figure 3). The fixed-film digester used for the dairy wastewater is inappropriate for algae digestion due to the high solids content of the harvested algae; instead, a stirred tank digester is required. The size of the digester was determined based on an assumed 25-day HRT. The walls and roof were assumed to be reinforced concrete with the same heat transfer coefficients as the fixed-film digester. A 35°C operating temperature is maintained by direct steam injection into a heating loop whose operation was modeled using equation 2 (variables are defined in equation 1):

$$\dot{m}_{w,in} h_{w,in} + \dot{m}_{st} h_{st} = \dot{m}_{w,out} h_{w,out}, \qquad (2)$$

where the w subscript indicates water and the st subscript indicates steam.

In addition to mixing from the heating loop, the digester is mixed intermittently by an internal impeller to prevent the settling of solids. Continuous mixing was explored but was found to have little impact on the results. The impeller's electrical power consumption was assumed to be 6.6 W/m^3 digester volume (EPA 1979). Biogas leakage was also assumed to be 2% for this digester.

Data are not available in the literature for the anaerobic digestion of ATS algae. Therefore parameters were obtained from the digestion of suspended algae cultures. The benthic algae from the ATS and the suspended algae used in digestion studies by Golueke and colleagues (1957) have comparable levels of nitrogen and phosphorus. Sialve and colleagues (2009) conducted a literature review of anaerobic digestion of algae and found that biogas yields were in the range of 0.15 to 0.45 cubic meters per kilogram (m^3/kg)[9] volatile solids with a methane fraction of 62% to 76%.

In the baseline scenario, biogas production of 0.36 m^3/kg volatile solids with a methane fraction of 68% was assumed. The volatile solids of algae were calculated from the total solids by assuming an ash content of 7% (Kebede-Westhead et al. 2004). Methane production levels were included in the scenario analysis.

The capital cost for the stirred tank digester was estimated based on a cost function derived by the EPA AgStar (2010) program for manure digesters. The construction cost of an algae digester was assumed to be comparable to that of a manure digester.

Water and Sludge Pumping

Pumping energy is required to extract and transport dilution water, wastewater influent, and recirculation water through the ATS. Water flow rates over the ATS were calculated using the Manning equation (equation 3):

$$V = \frac{k}{n} R_h^{\frac{2}{3}} S_o^{\frac{1}{2}}, \qquad (3)$$

where
$k = 1$;
$n = 0.03$, equivalent to a weedy excavated earth channel;
R_h = hydraulic radius; and
S_o = slope [1% consistent with Mulbry et al. (2008)].

The resulting velocity for 1% slope and 1 centimeter (cm)[10] depth was 0.15 m/s with a corresponding recirculation flow rate of 2.68 m^3/s. The volume of water pumped through the ATS has a significant impact on electricity consumption and is explored in the scenario analysis. In the model, pipes were sized for fluid velocities in the range of 1 to 2 m/s. Energy requirements were determined for fluid pumping by multiplying the total dynamic head equation (equation 4) by the fluid flow rate:

$$H_D = \Delta h + f\frac{Lv^2}{2Dg} + k_i\frac{v^2}{2g}, \tag{4}$$

where
Δh = elevation change;
f = friction factor obtained from the Moody chart,
L = pipe length,
v = fluid velocity,
D = pipe diameter,
g = gravitational acceleration, and
k_i = minor loss coefficients (Munson et al. 2006).

Head losses from sludge pumping were found to be 1.5 to 4 times those of water according to a survey of treatment plants (Sparr 1971). Electrical pump efficiencies were obtained from ITT wastewater and sludge pump curves based on calculated flow rates and total dynamic head (ITT 2010). Details can be found in the supporting information on the Web.

Sludge Dewatering and Drying

Centrifugation was selected as the dewatering method for the biosolids obtained from the algae digester. Centrifugation is commonly used to dewater anaerobically digested municipal wastewater sludge and therefore data are available for this application (Zenz et al. 1978). The effect of centrifugation on digested ATS biosolids has not been studied, but the model assumes comparable dewatering characteristics to those of digested municipal wastewater sludge. The sludge cake was assumed to contain 20% solids with 57% and 77% of the nitrogen and phosphorus, respectively, retained in the centrifuge cake. Supernatant nutrients recirculate back into the ATS sump for additional absorption by the algae.

The Sweco 414 decanting bowl centrifuge was selected in order to estimate the cost and electricity use for dewatering because it appeared to have sufficient g-force and capacity for this application (Mook 2010; Zenz et al. 1978). The centrifuge would operate at 2,149 g-force with a power consumption of 24.2 kilowatts (kW)[11] for approximately five hours per day, resulting in a class B biosolid suitable for field application based on EPA guidelines (EPA 1993).

ATS algae grown on dairy wastewater has been shown to have low heavy metal concentrations and is appropriate for use as a fertilizer substitute (Kebede-Westhead et al. 2004). The cost of the selected centrifuge was quoted at $125,000 (Mook 2010). Lang factors were used to estimate the cost of installation, peripheral piping, electrical, and instrumentation as shown in the supporting information on the Web (Peters et al. 2003).

The biosolids may be dried to 10% water content in order to achieve a class A biosolids designation. Drying may also be desirable if the biosolids are to undergo long-term storage. Due to the high energy demands and capital costs, however, drying is not included in the baseline scenario (though it is included in one of the alternate worst-case scenarios). Drying was assumed to take place in a triple-pass drum dryer powered by biogas generated on-site. Dryer cost was estimated using installed cost curves provided in work by Peters and colleagues (2003), which were adjusted for inflation. Lang factors were applied for peripheral equipment, design, and construction fees. Additional details can be found in the supporting information on the Web.

Biogas Utilization

Biogas produced from the fixed-film manure digester and stirred-tank algae digester is compressed for use in a biogas generator, biogas boiler (if required), and optional drum dryer. The model uses the biogas first to heat the digesters and optional dryer and then to produce electricity, which is sold to the grid. Waste heat is also captured from the generator to produce low-pressure steam for digester heating. A linear optimization program was used to maximize the electricity output from the biogas, given operational constraints (see the supporting information on the Web).

Prior to combustion, the biogas must undergo cleanup in order to remove hydrogen sulfide and excess water vapor (figure 4). Cleanup results in a parasitic load on the generator, which was estimated at 7.7% of generator output (Silva 2010). Because not all biogas is supplied to the generator, the parasitic load was determined as a function of total biogas combustion (equation 5):

$$\textit{Parasitic load (kW)} = \frac{7.7\%}{\textit{Electrical efficiency}} \times \frac{\textit{MJ biogas}}{\textit{day}} \times \frac{\textit{day}}{24\,hr} \times \frac{kWh}{3.6MJ}. \tag{5}$$

Data for the biogas generator were obtained from personal communication with a director at Calpwr, a vendor for gas generators (Silva 2010). Thermal efficiency and residual heat recovery efficiency for a typical biogas-powered reciprocating engine were assumed to be 38% and 56.5%, respectively. Generator GHG emissions were estimated using emissions from biogas generators employed at wastewater treatment plants in California (ARB 2009).

A dual-fuel gas boiler was selected in order to estimate overall efficiency and electricity usage for the biogas boiler. This boiler achieves 85% efficiency using natural gas and consumes 255 W of electricity (Cleaver-Brooks 2009). The LCA model assumes

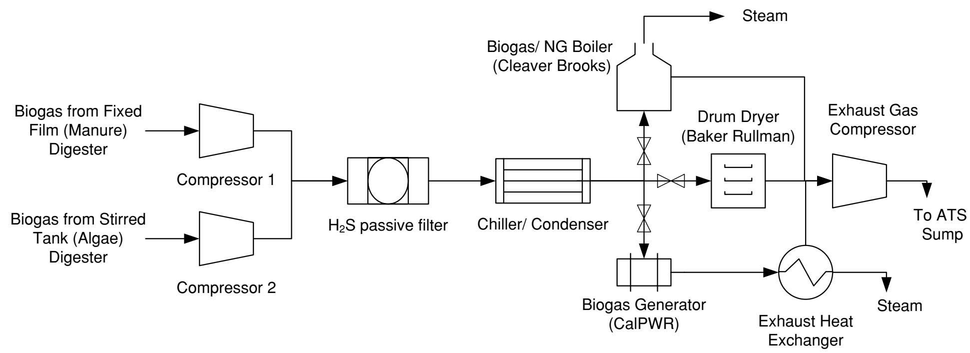

Figure 4 Biogas processing and combustion schematic. H_2S = hydrogen sulfide; NG = natural gas.

comparable boiler efficiencies using cleaned biogas in place of natural gas. Sensitivity analysis demonstrated that results are not sensitive to this parameter.

The boiler is capable of producing low-pressure steam at 15 psig[12] for digester heating via direct steam injection. Emissions for the biogas boiler were estimated using emission factors for external combustion of natural gas (EPA 1998). A compressor is required to remove the biogas from the digester and to pressurize it to the level required for combustion equipment. Compressor power was calculated to be in the range of 107 to 220 W assuming adiabatic compression. The exhaust gas must also be compressed in order to provide CO_2 aeration to the ATS sump, and similar calculations resulted in compressor power of 4.2 kW (see the supporting information on the Web).

Material and Energy Inputs

Impacts for the production of concrete, steel, and high-density polyethylene (HDPE) plastic were included in the life cycle inventory (LCI), but material transport and installation were excluded. Concrete inventory data were obtained from the Portland Cement Association (Marceau et al. 2007). Steel inventory data were obtained from the World Steel Association (2010) for rebar and plate steel that were used in ATS and digester construction. HDPE plastic landfill liners were used for the construction of the ATS and winter holding pond, with inventory data obtained from work by Bousted (1999).

Process equipment manufacturing was modeled using EIO-LCA methods (Green Design Institute 2002). The manufacture of harvesting equipment was modeled based on the economic sector "farm machinery and equipment" and the centrifuge, rotary dryer, and biogas generator were modeled as "other industrial equipment." All equipment costs were adjusted to 2002 dollars for consistency with the EIO model used.

Displacement credit was assigned to two coproducts from the treatment process: electricity and digested algae biosolids. The biogas generator is expected to operate as a base-load power source and was assumed to displace electricity from the average U.S. grid mix. The U.S. grid mix inventory was obtained from the PE International (2009) database accessed through PE International's GaBi4 software. Biosolids were given displacement credit for ammonium nitrate and phosphoric acid fertilizers based on bioavailable nitrogen and phosphorus content. Inventory data on materials are included in the supporting information on the Web.

Transportation of Sludge and Fertilizer

Transportation of algae biosolids to nearby agricultural fields was assumed to occur by tractor trailer with an average round-trip travel distance of 120 kilometers (km),[13] within the range assumed by a Canadian study for biosolids application (Brown et al. 2010). So long as biosolid transport distances are less than 800 km, overall conclusions about energy consumption and GHG emissions are not affected. The displaced impacts of shipping chemical fertilizer were subtracted from the impacts of transporting biosolids. It was assumed that chemical fertilizer travels by tractor trailer for 800 km based on statistics from the Bureau of Transportation Statistics (BTS 2007). Environmental impacts for trucking were obtained from the USLCI database accessed through GaBi4 software (NREL 2009).

Trucking costs were determined from BTS statistics for trucking and were adjusted for inflation using the producer price index, resulting in a cost of $0.133USD/mile[14] (BTS 2009). Loading and unloading costs of $6 (Canadian dollars)/t were used by Ghafoori and Flynn (2006). This cost was converted to U.S. dollars at the 2006 exchange rate and then inflated to 2009 dollars, resulting in a cost of $6.33USD/t of biosolids.

Process Scenarios

Six process scenarios were developed to capture the performance of the system under a range of process characteristics (table 1). The first scenario excludes the ATS. The remaining five scenarios all include the ATS and vary parameters that were found to significantly impact the results. Scenarios were designed to test worse-than-expected performance.

Policy Scenarios

In addition to the process scenarios, legislated policies could also affect the cost of an ATS system. The two baseline process scenarios were reanalyzed under three policy scenarios: a

Table 1 Scenario design, where parameters shown in the top row vary among scenarios

Scenario	*Average ATS productivity (10 months/ annual) (g/ m^2/ day)*	*Algae digester methane (m^3/kg VS)*	*Biogas methane (%)*	*ATS water recirculation (m^3/s)*	*Dry biosolids*
Baseline without ATS	—	—	—	—	—
Baseline with ATS	19/16.5	0.25	68%	2.68	No
Low algae productivity	13.4/11.9	0.25	68%	2.68	No
Poor algae digester performance	19/16.5	0.17	62%	2.68	No
High ATS water recirculation	19/16.5	0.25	68%	9.02	No
Dry biosolids	19/16.5	0.25	68%	2.68	Yes

Note: ATS = algal turf scrubber; g/m^2/day = grams per square meter per day; m^3/kg = cubic meter per kilogram; VS = volatile solids; m^3/s = cubic meters per second.

nutrient trading scheme, GHG emission trading, and a renewable energy credit program.

Results and Discussion

Process Scenarios

The results from the process scenarios indicate that total primary energy consumption and GHG emissions vary significantly among scenarios (figure 5). The results also show that the ATS provides significant eutrophication benefits, in accordance with its main purpose as a wastewater treatment mechanism. Nearly all of the eutrophication impacts of the system result from the direct discharge of the treated wastewater, not upstream processes.

Total costs, which include annualized capital costs and operating costs, are significantly higher with the ATS. For the baseline scenario with and without the ATS, the costs of treatment are \$1.42 USD and \$0.20/m^3 of wastewater, respectively.

The baseline scenario shows that inclusion of the ATS results in net energy and GHG emissions displacement. The primary energy and GHG benefits of using the ATS are lost when less favorable parameters are used for algae productivity, digester performance, water recirculation, and biosolids drying.

Lower algae productivity necessitates the use of a larger ATS. The larger ATS requires greater water recirculation and thus more pumping energy. When average annual algae productivity falls below 11.8 g/m^2/day, the ATS becomes a net emitter of GHG emissions, assuming other parameters are held constant.

Poor digester performance also adversely affects energy and GHG emission performance because less electricity is produced. The use of high water flow rates in the ATS also results in net energy consumption and GHG emissions as a result of increased pumping energy demand. Likewise, drying of biosolids has a negative energy and GHG performance due to the high biogas consumption of the dryer. Dryer operation leaves less biogas for electricity generation, and consequently less grid electricity is displaced. Still, the substantial reduction in eutrophication potential provided by the ATS may justify some increase in GHG emissions and energy consumption. Because this analysis adjusted the size of the ATS to provide a constant level of treatment among different scenarios, eutrophication potential had little variation among scenarios where the ATS was included.

The results of the scenario analysis for process parameters suggest that the ATS has value as a wastewater treatment mechanism, but its energy and GHG emissions benefits are dependent on process design and performance. In order to achieve the GHG emission reductions shown in the baseline scenario, average water depths on the ATS flow-way should not exceed 1.3 cm. Also, drying the biosolids never resulted in positive energy or GHG performance, even if optimistic assumptions were made about ATS performance. The energy used to dry the biosolids far outweighed the energy savings in the transport of biosolids.

The potential cost of employing ATS technology is significant if it is borne entirely by the dairy industry. The 750-cow dairy simulated in this model would spend \$1,514/day (\$736/cow/year) for treatment with the ATS versus \$217/day (\$106/cow/year) for treatment with only the fixed-film anaerobic digester. Capital costs account for the majority of these expenses. Because all of the scenarios result in net costs to dairy operators, none of these systems are likely to be adopted without incentives.

Policy Scenarios

Trading programs can create incentives that will encourage dairies to adopt the ATS system by reducing life cycle costs (figure 6). If operated within the parameters described in the baseline scenario, the ATS will result in improved environmental performance of dairy and livestock operations.

Nutrient trading schemes have emerged in recent years in order to address eutrophication issues in the Mississippi River

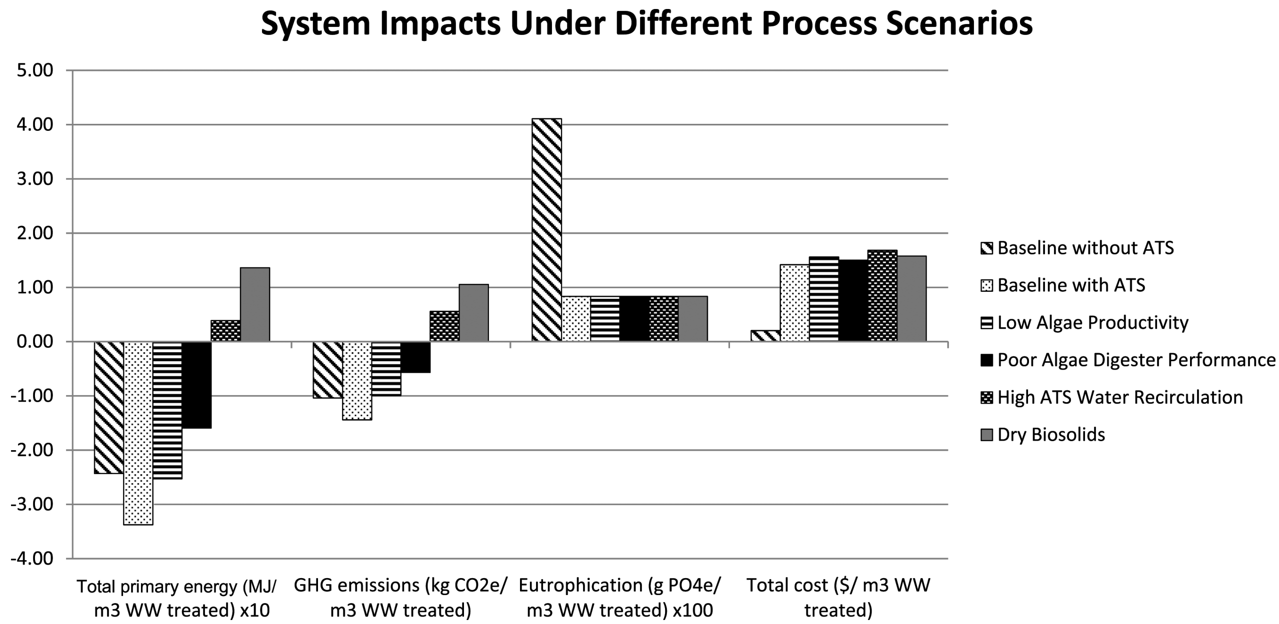

Figure 5 Environmental and cost impacts of the wastewater treatment process for different process scenarios provided in table 1. MJ = megajoules; CO2e = carbon dioxide equivalent; PO4e = phosphate equivalent; m3 WW = cubic meters wastewater; GHG = greenhouse gas. Further details provided in the supporting information on the journal Web site.

basin. Under these trading programs, municipal wastewater plants may pay farmers to reduce nitrogen and phosphorus waste as an offset to meet the EPA's new effluent standards. Greenhalgh and Sauer (2003) estimated the value of nutrient removal at \$4.40 to \$11/kg of nitrogen and \$11 to \$15.40/kg of phosphorus based on the Mississippi River basin trading program.

Using the lower bounds for these estimates, the treatment system without the ATS removed nutrients worth \$0.12/m^3 of wastewater, which is \$0.08 lower than the cost of treatment. The treatment system with the ATS resulted in nutrient removal worth \$1.81/m^3 of wastewater. This nutrient removal value is higher than the cost of treatment and results in net earnings of \$0.39/m^3 of wastewater. However, the net earnings associated with nutrient removal disappear if the prices drop below \$3.45/kg of nitrogen and \$8.62/kg of phosphorus.

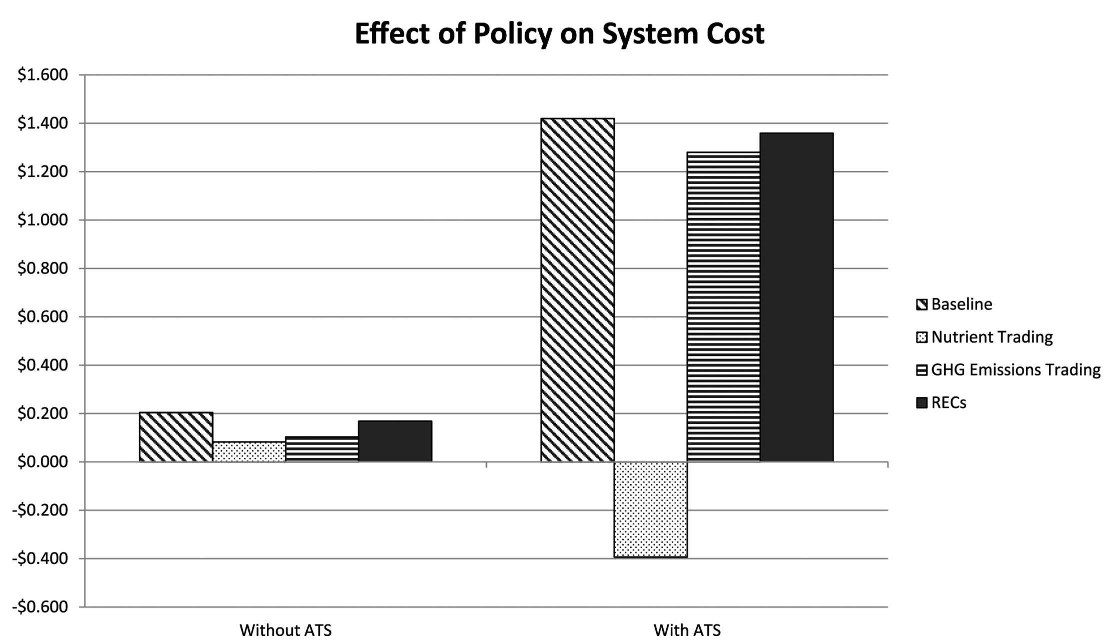

Figure 6 Life cycle system cost for three policy scenarios per m^3 of wastewater. \$ = US dollars. RECs = renewable energy credits.

Markets have also emerged for trading GHG emissions in recent years, though the United States lacks the legal framework required for carbon markets to operate effectively. Nevertheless, studies have been conducted to determine the cost of economic and ecological damage from GHG emissions. Tol (2005) conducted a literature survey and found that damage estimates were highly variable, ranging from −\$6.6 to \$1,667 per metric ton carbon dioxide equivalent (t/CO_2-eq).[15] The arithmetic mean of the surveyed studies was \$97/t CO_2-eq. Applying this value to the treatment processes results in a benefit of \$0.10/m^3 of wastewater for the system without the ATS and \$0.14/m^3 wastewater with the ATS in the baseline scenario.

The treatment system should also qualify as a source of renewable electricity, and therefore could benefit from the EPA's renewable energy credit (REC) program. As more states adopt renewable electricity standards, the prevalence of RECs could increase. Currently, electricity certified under the REC program commands a \$0.01 to \$0.03 premium per kilowatt-hour (kWh)[16] (DOE 2010b). If electricity could be sold at \$0.12/kWh rather than \$0.10/kWh (as it is in the baseline scenario), then the cost of wastewater treatment with and without the ATS drops to \$1.36 and \$0.17, respectively. Like GHG emissions trading, RECs would have little impact on ATS cost-effectiveness compared to nutrient trading programs.

Conclusion

The results of this research suggest that ATS technology can consistently lead to reduced eutrophication potential when applied to dairy wastewater treatment. The ATS may also lead to economic benefits under the right policy framework, particularly with nutrient trading programs. Credits for GHG emissions and renewable energy are not as effective as nutrient trading programs for enhancing ATS cost-effectiveness, but can still lead to small economic benefits to dairy operators. While the ATS always reduces the eutrophication potential of dairy wastewater, the energy and GHG benefits of installing an ATS depend on facility design and operation.

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Notes

1. One milligram (mg, SI) = 10^{-3} grams (g) ≈ 3.53×10^{-5} ounces (oz). One liter (L) = 0.001 cubic meters (m^3, SI) ≈ 0.264 gallons (gal). Total Kjehldahl nitrogen is a combination of organically bound nitrogen and ammonia.
2. One cubic meter (m^3, SI) = 10^3 liters (L) ≈ 264.2 gallons (gal).
3. One watt (W, SI) ≈ 3.412 British thermal units (Btu)/hour ≈ 1.341×10^{-3} horsepower (HP). One square meter (m^2, SI) ≈ 10.76 square feet (ft^2).
4. One meter (m, SI) ≈ 3.28 feet (ft).
5. One gram (g) = 10^{-3} kilograms (kg, SI) ≈ 0.035 ounces (oz).
6. One hectare (ha) = 0.01 square kilometers (km^2, SI) ≈ 0.00386 square miles ≈ 2.47 acres.
7. One metric ton (t) = 10^3 kilograms (kg, SI) ≈ 1.102 short tons.
8. One acre ≈ 0.405 hectares (ha) ≈ 4.05×10^{-3} square kilometers (km^2, SI) ≈ 1.56×10^{-3} square miles.
9. One kilogram (kg, SI) ≈ 2.204 pounds (lb).
10. One centimeter (cm) = 0.01 meters (m, SI) ≈ 0.394 inches (in.).
11. One kilowatt (kW) ≈ 56.91 British thermal units (Btu)/minute ≈ 1.341 horsepower (HP).
12. psig (pounds-force per square inch gauge) measures the pressure relative to the surrounding atmosphere. One psi ≈ 0.068 atmospheres ≈ 6,895 Pascals.
13. One kilometer (km, SI) ≈ 0.621 miles (mi).
14. One mile (mi) ≈ 1.61 kilometers (km, SI).
15. Carbon dioxide equivalent (CO_2-eq) is a measure for describing the climate-forcing strength of a quantity of greenhouse gases using the functionally equivalent amount of carbon dioxide as the reference.
16. One kilowatt-hour (kWh) ≈ 3.6×10^6 joules (J, SI) ≈ 3.412×10^3 British thermal units (Btu).

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About the Authors

Brendan Higgins is a master's student in the Institute of Transportation Studies and a PhD student in biological systems engineering at the University of California, Davis in Davis, CA, USA. **Alissa Kendall** is an assistant professor in the Civil and Environmental Engineering Department at the University of California, Davis.

Supporting Information

Additional supporting information may be found in the online version of this article:

Supporting Information S1: This supporting information details the technical aspects of algal turf scrubbing systems, biogas utilization, and the associated costs of both. This includes measurements and calculations of nutrient loads and removal, pump efficiency, and algae digesters as well as biogas output modeling.

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