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Harvesting microalgae grown on wastewater

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HIGHLIGHTS

- Algae were grown in a photobioreactor with wastewater centrate as the feed source.
- Multivalent metal salts and cationic polymers were capable of >91% recovery.
- There is a tradeoff between the environmental impacts and costs of coagulants.
- Belt presses are recommended for algae dewatering prior to oil extraction.

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ABSTRACT

The costs and life cycle impacts of microalgae harvesting for biofuel production were investigated. Algae were grown in semi-continuous culture in pilot-scale photobioreactors under natural light with anaerobic digester centrate as the feed source. Algae suspensions were collected and the optimal coagulant dosages for metal salts (alum, ferric chloride), cationic polymer (Zetag 8819), anionic polymer (E-38) and natural coagulants (*Moringa Oleifera* and *Opuntia ficus-indica* cactus) were determined using jar tests. The relative dewaterability of the algae cake was estimated by centrifugation. Alum, ferric chloride and cationic polymer could all achieve >91% algae recovery at optimal dosages. Life cycle assessment (LCA) and cost analysis results revealed that cationic polymer had the lowest cost but the highest environmental impacts, while ferric chloride had the highest cost and lowest environmental impacts. Based on the LCA results, belt presses are the recommended algae dewatering technology prior to oil extraction.

1. Introduction

With growing concerns about global climate change and energy security, the use of renewable biofuels has received a large amount of attention. Microalgae have very high biomass productivities and are able to fix CO_2 through photosynthesis, while using wastewater as a nutrient source (Min et al., 2011; Yuan et al., 2012). In particular, centrate from the anaerobic digestion of municipal sludge has been proposed as a culture medium for algae production, due to the synergy between algal biofuel production and anaerobic digestion infrastructure (Yuan et al., 2012). The low biomass concentrations usually achieved in microalgal cultivation systems; however, lead to significant costs for downstream processing of algal suspensions (Uduman et al., 2011). It has been estimated that harvesting can account for 25–60% of the total cost of microalgae production (Grima et al., 2003). Previous Life Cycle Assessment (LCA) studies have also shown that one of the major contributors to energy consumption and greenhouse gas (GHG) emissions in the algae harvesting step is the use of coagulants (Lardon et al., 2009).

Microalgae have a wide range of properties that affect their separation from water including size, shape, specific gravity, surface charge, motility, growth phase, presence of appendages and extracellular organic matter (EOM) composition and concentration (Henderson et al., 2008). Properties of the culture medium, including cell concentration, pH and ionic strength, also have a strong impact on algae harvesting efficiency. Single spherical cells with diameters of 3–5 μm and specific gravities only slightly higher than water are common for green algae, such as *Chlorella* sp., that grow well in wastewaters (Yuan et al., 2012). Microalgal properties that affect bioflocculation, such as EOM concentration, can change depending on algal growth stage and substrate concentrations (Danquah et al., 2009). In addition, microalgae are capable of restoring their negative surface charge after coagulation, by transferring ions across their cell membrane (Pieterse and Cloot, 1997).

Significant research has been carried out on removal of microalgae from drinking water source waters (see Henderson et al., 2008 for review). Fewer studies have been conducted on harvesting

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microalgae at the higher biomass concentrations required for biofuel or aquaculture feed production industries. In these applications, algae harvesting usually involves a two-step process (Wiley et al., 2011). The first step involves destabilization of the algal cells using coagulants, followed by sedimentation or floatation. In the second step, the wet algae sludge is dewatered by filtration, centrifugation or thermal processes.

Polyvalent metal salts have long been used to destabilize particles for drinking water treatment and have been recently studied for algae harvesting (Papazi et al., 2010; Wyatt et al., 2012). Papazi et al. (2010) screened twelve different salts for their potential as coagulants for harvesting *Chlorella minutissima* cultures. Chloride and sulfate salts of aluminum and ferric iron were the most efficient coagulants, although aluminum caused some cell lysis and ferric salts caused cell discoloration. Wyatt et al. (2012) achieved >90% recovery of *Chlorella zofingiensis* at optimal ferric chloride dosage and pH. Electrocoagulation has also been used successfully as an alternative method of metal hydroxide generation. Uduman et al. (2011) achieved greater than 98% algae recovery using electrocoagulation when voltage and run times were optimized.

Dewatering polymers are widely used for wastewater sludge management and have also been examined for algae harvesting (Tenney et al., 1969; Knuckey et al., 2006). Tenney et al. (1969) screened cationic, anionic and non-ionic polymers for recovery of indigenous algae. No flocculation was observed with the anionic and nonionic polymers; however, low residual supernatant turbidity was observed when the cationic polymer (Dow C-31) was used at relatively low concentrations (2.5 mg/L). The greater amounts of EOM produced during late log and declining growth phases appeared to aid in the coagulation process. Knuckey et al. (2006) achieved >80% recovery of marine microalgae using a non-ionic polymer (Magnafloc LT-25) when pH was adjusted between 10 and 10.6.

Several natural polymers have also been investigated as coagulants for algae harvesting. These materials can be produced in a more sustainable way than synthetic organic polymers (Buttice et al., 2010) and have the potential to generate an algae product that is free from undesirable and potentially toxic contaminants (Ahmad et al., 2011; Teixeira et al., 2012). Chitosan is produced from chitin, a structural element in the exoskeletons of crabs and shrimp. Ahmad et al. (2011) achieved >99% recovery of *Chlorella* sp. with relatively low concentrations of chitosan, once chitosan concentration, mixing rate and sedimentation time were optimized. Divakran and Pillai (2002) recovered >90% of fresh water algae, *Spirulina*, *Oscillatoria* and *Chlorella* sp., at pH 7.0 with a low dosage (15 mg/L) of chitosan. *Moringa oleifera* seed flour has been used traditionally for drinking water treatment in parts of Asia and Africa, and was investigated by Teixeira et al. (2012) for harvesting *Chlorella vulgaris*. Although not previously investigated for algae harvesting applications, mucilage extracted from *Opuntia ficus-indica* cactus was shown to increase sedimentation rates for clay particles and bacteria (Buttice et al., 2010).

Dewatering processes are usually employed after the harvesting step to increase the solids content of algae slurries prior to downstream processing, such as oil extraction. Various technologies, including belt filter presses, thermal drying and centrifugation have used for algae dewatering in previous studies (Shelef et al., 1984; Grima et al., 2003; Cooney et al., 2011). Selection of an appropriate dewatering technology is typically based on the initial solids content of the algae slurry and the final solids content required.

LCA is a framework used to assess the environmental impacts associated with the life cycle of a product or production process. Previous LCA studies on algae biofuel production have produced controversial results depending on assumptions made regarding cultivation process, harvesting technologies and final biofuel products (Lardon et al., 2009; Soratana and Landis, 2011; Beach et al., 2012). Only one study (Beach et al., 2012) focused on the algae harvesting step and investigated different coagulants (chitosan biopolymer, ferric sulfate, and alum) as well as dewatering technologies (centrifugation and filtration/chamber press processes). The cost of algae harvesting; however, was not evaluated in this study (Beach et al., 2012) and the algae were not cultivated using wastewater.

The goal of this research was to conduct a cost analysis and comparative LCA of different harvesting alternatives that can be used for algae grown from wastewater for biofuel production. Algae were grown in semi-continuous culture on centrate from anaerobic digestion of municipal wastewater sludge. Optimal dosages of multivalent metal salts, synthetic polymers and natural coagulants were determined using jar tests. The relative dewaterability of the algae cake was estimated by centrifugation. Data from these experiments as well as data obtained from literature for the energy intensity of different dewatering technologies were used to estimate the cost, life cycle energy consumption and GHG emissions of the harvesting process.

2. Methods

2.1. Algae culturing studies

Algal growth experiments were conducted under natural illumination in a temperature controlled (25–32 °C) greenhouse at the University of South Florida Botanical Gardens in Tampa Florida between November 1st and December 19th 2011. Two vertical hanging tubular plastic bag PBRs were obtained from Professor Leiv Mortensen of the Faculty of Plant and Environmental Sciences at the Norwegian Life Sciences University in Oslo. Each cell had a height of 273 cm, a diameter of 12 cm, a total volume of 10 L and a working volume of 7 L. Industrial grade CO_2 (AirGas Inc., Kennesaw, GA) was mixed with compressed ambient air to achieve a 2% CO_2/air mixture. The gas flow rate to each PBR cell was maintained at 500 mL/min using rotameters supplied with needle valves (Cole Parmer Inc., Vernon Hills, IL) connected to coarse bubble diffusers. Based on visual observation, the gas flow maintained the reactor contents in a well-mixed condition.

One PBR cell was fed an anaerobic digester centrate feed and the second cell was operated with a 50:50 mixture of centrate and synthetic aquaculture wastewater. Centrate was collected weekly from a sump that collects reject water from gravity belt thickeners at the Howard F. Curren Advanced Wastewater Treatment Facility (HFCAWTF) in Tampa, FL. HFCAWTF digests a mixture of primary and waste activated sludge in a mesophilic (35 °C) single-stage anaerobic digester with a 21-day SRT. Biosolids are dewatered using a gravity belt thickener, with polymer addition. The belts are periodically washed with treated wastewater effluent, which significantly dilutes the centrate. In order to maintain a relatively constant influent feed concentration, the total nitrogen (TN) and total phosphorous (TP) concentrations in the centrate were measured on the day of collection and adjusted as needed to between 200–250 mg/L TN and 25–76 mg/L TP, by dilution with groundwater or addition of $(NH_4)_2SO_4$ or KH_2PO_4. In addition, the centrate was filtered using a filter cloth to reduce turbidity and improve the light transmission of the algae culturing medium.

The aquaculture-centrate mixture (ACM) was based on a prior study in our research group (Watson et al., 2011) showing that incorporation of algae and anaerobic digestion could reduce the nutrient impacts of land based recirculating aquaculture systems. ACM was formulated by mixing 50% TN adjusted centrate and 50% synthetic aquaculture wastewater. Synthetic aquaculture wastewater feed was based on observed nutrient concentrations

in aquaculture wastewater (Watson et al., 2011) and contained 200 mg/L NO_3–N (KNO_3) and 25 mg/L TP (KH_2PO_4) in local groundwater.

Chlorella-like wild algae harvested from the surface of a secondary clarifier at the HFCAWTF were used to inoculate the reactors at an initial concentration of 620 mg TSS/L. The reactors were maintained at a mean cell residence time of 7-days for approximately nine months by removing 1 L of the contents of each PBR cell on a daily basis and replacing it with fresh nutrient feed. Data is reported for the period of October 8 – December 22, 2012, after acclimation of wild algae and PBR operating protocols were established.

2.2. Harvesting studies

Jar tests were conducted using reagent grade (Fisher, Pittsburgh, PA) alum ($Al_2(SO4)_3{\cdot}12H_2O$) and ferric chloride ($FeCl_3{\cdot}6H_2O$), Zetag (BASF Chemical Company, Suffolk, VA) 8800 series cationic polymers (8814, 8816, 8818, and 8819), two anionic polymers (Magnafloc E-38 and E-34; BASF Chemical Company, Suffolk, VA) and two natural flocculants (*M. Oleifera* and *Opuntia ficus-indica* cactus). Cactus mucilage was prepared by Ms. Audrey Buttice of the Department of Chemical and Biomedical Engineering at University of South Florida (USF). Briefly, cactus pads were obtained from Living Stones Nursery (Tuscon, AZ). The pads were cleaned, cut into 2.5 cm pieces and boiled in deionized water for 20 min. The mixture was blended in an Osterizer blender and the pH was adjusted to 7 by addition of NaOH. The mixture was spread out in pans with a thickness of approximately 0.6 cm and dried in an oven at 60 °C for 24 h. The dried mucilage was then ground using a mortar and pestle. *Moringa* was obtained from ECHO (North Fort Myers, FL) and prepared using recommendations from Price (2007). Briefly, seeds were removed from the shell, crushed and mixed with water and the solution was poured through a filter cloth.

Coagulation tests were conducted on suspensions collected from the algal PBRs using a Phipps & Bird Jar Test Apparatus (Richmond, Virginia). After coagulant addition, the suspensions were rapid mixed for 1-min at 70 rpm, followed by 15-min of flocculation at 25-rpm and then 1-h of settling. Jars without added coagulant were used as controls. Supernatant was drawn off with a wide bore pipette and pH, turbidity and TSS were measured as described below. The optimal coagulant concentration was taken as the point when supernatant TSS and turbidity leveled out. If no level point was achieved, the jar tests were repeated with a different range of coagulant concentrations.

A preliminary dewatering study was performed using the method described by Jin et al. (2003). Optimal recovery was found at centrifuge speed and time of 2000 rpm for 10 min, respectively (data not shown). For selected jar tests, settled algae were centrifuged using a Thermo Scientific (Pittsburgh, PA) CL2 centrifuge at 2000 rpm for 10 min. The total solids (TS) concentration was measured on the algae cake after dewatering. The percentage algae recovered was calculated from a material balance on the system.

2.3. Analytical methods

Temperature and light intensity in the greenhouse were measured and recorded using a HOBO® U12 temperature/relative humidity/light intensity meter and datalogger (Oncet, Bourne, MA). Total Suspended Solids (TSS), dissolved oxygen (DO), and pH were measured in the PBR effluent on a daily basis. Influent and effluent concentrations of TN, TP, Chemical Oxygen Demand (COD), NO_3^-–N and NH_4^+–N were measured on weekly basis. *Standard Methods* (APHA et al., 2012) were used to measure TS (2540G), TSS (2540B), DO (4500-O C), NO_3^-–N (4500-NO_3 B), TN (4500-N), TP (4500-P C), COD (5220 D). NH_4^+–N concentrations were measured by the Salicylate Method using Hach (Loveland, CO) test vials. pH was measured using a calibrated pH meter and probe (Metrohm, Riverview FL). Turbidity was measured using a calibrated Hach (Loveland, CO) 2100Q portable turbidimeter. The method detection limits (MDL) for TN, TP, NH_3–N were 7.0, 0.06 and 0.6 mg/L, respectively.

2.4. Cost analysis and life cycle assessment

The cost analysis and LCA studies were conducted to quantify the cost, energy and GHG emissions associated with the algae harvesting process using alum, ferric chloride and cationic polymer (Zetag 8819). Anionic polymer, non-ionic polymer, *moringa* and cactus mucilage were not considered due to the relatively low harvesting efficiencies achieved in the laboratory experiments. Costs of coagulants were obtained from bulk vendors of industrial chemicals (Alibaba, BASF, Dow Chemical). For the LCA study, the functional unit was the production of one metric ton (MT) of dried algae. The relevant amount of each coagulant required to harvest one MT of dried algae was based on the jar test results.

Cumulative Energy Demand Analysis (CEDA) in Simapro 7.2 software was used to estimate the life cycle energy (direct energy use as well as the upstream energy consumption) required to produce the amount of alum or ferric chloride needed to harvest one MT of dry algae (Frischknecht and Weidema, 2010). The Tool for the Reduction and Assessment of Chemical and Other Environmental Impacts (TRACI) developed by U.S. Environmental Protection Agency (EPA) was used to estimate the GHG emission resulting from the production of the required amount of alum or ferric chloride (Bare, 2012).

As data for cationic polymer was not available in the Simapro database, the life cycle energy required and GHG emission to produce one MT of cationic polymer was obtained from Tripathi (2007). Tripathi (2007) calculated the life cycle energy based on direct energy needed to produce a unit of polymer and associated GHG emission based on the average US electricity grid (Kim and Dale, 2005).

3. Results and discussion

3.1. Photobioreactor studies

A summary of the biomass production and nutrient removal data from the algal PBR studies is shown in Table 1. More detailed results have been presented elsewhere (Dalrymple et al., 2013). Low algal productivity was observed compared with other recent studies of microalgal growth on anaerobic digester centrate (Yuan et al., 2012). Yuan et al. (2012) achieved a maximum algal productivity of 6.8 g/m^2-d under batch conditions. Min et al. (2011) achieved algal productivities of 34.6 g/m^2-d in an open pilot-scale PBR. Both studies used municipal sludge centrate, wild type *Chlorella* species and artificial light conditions. The low productivity

Table 1
Mean and maximum algae growth and nutrient removal data from the centrate and ACM PBRs. Standard deviations shown in parentheses.

Parameter	Centrate		ACM	
	Mean	Max	Mean	Max
Productivity (g/m^2/day)	2.3 (0.77)	3.6	1.7 (0.89)	4.0
Biomass density (g/L as TSS)	664.4 (222)	1030	478.2 (259)	1,140
TN removal efficiency (%)	65	92	64	91
Effluent TN (mg/L)	76.2 (71.1)	180	60.3 (60.5)	130
NH_4^+ removal efficiency (%)	95	77	86	100
Effluent NH_4^+–N (mg/L)	50 (38.0)	92	5.0 (8.6)	15
TP removal efficiency (%)	72	79	73	93
Effluent TP (mg/L)	12.5 (23.6)	54.6	20.1 (34.2)	71.3
COD removal efficiency (%)	8	15	10	15

observed in this study may have been due to the low natural ambient light intensity (2.3–9.4 mol/m^2/day) during the wet season in Southwest Florida compared with the light intensity used by Min et al. (2.16 mol/m^2/day) and Yuan et al. (5.3 mol/m^2/day). Average algal productivity was higher for the 100% centrate reactor than for the ACM reactor, most likely due to the algae being adapted to centrate during a long acclimatization phase (March–October, 2011), during which the reactor was only fed centrate. Both reactors achieved good overall removal efficiencies for nitrogen and phosphorous. Low COD removal efficiencies were most likely due to the low bioavailability of residual organic matter in centrate from anaerobic digesters (Yuan et al., 2012).

3.2. Harvesting studies

An overall summary of the jar test results for all coagulants tested is shown in Table 2. Among the metal salts tested, $FeCl_3$ had a lower optimal dose (122 mg/L) and slightly higher recovery (93%) than alum. Harvesting efficiencies with polyvalent metal salts were similar to those observed by other authors (Papazi et al., 2010; Wyatt et al., 2012). When using $FeCl_3$, the color of culture changed from green to brown-yellow. This could have a negative impact on the final product if the algae are to be used for pigments (Papazi et al., 2010). Note that no pH adjustment was done in these tests and pH remained >6.0 for all tests performed. Wyatt et al. (2012) found that at pH values below 4.0, the surface charge of *C. zofingiensis* was positive and ferric chloride was ineffective as a coagulant.

An initial screening study was conducted with the Zetag 8000 series cationic polymers (8814, 8816, 8819, 8846, 8848) to determine the best cationic polymer for further study (data not shown). Although excellent algae recovery (92–98%) was observed at low polymer concentration (34–41 mg/L) for all of the cationic polymers tested, Zetag 8819 was selected for further study because it provided the highest harvesting efficiency (98%) at the lowest optimal dose (34 mg/L). Variations in performance of different cationic polymers were most likely due to variations in charge density and molecular weight, which govern charge neutralization and bridging between algal particles, respectively. Low molecular weight cationic polymers may either not cause any flocculation or may be required at high concentrations, while overdoses of high molecular weight polymers can reverse the algal surface charge and stabilize the suspension (Thapa et al., 2009; Uduman et al., 2011.). Zetag 8819 also had the overall best performance, in terms of coagulant dose required and algae recovery, of all of the coagulants tested. No flocculation was observed with the non-ionic and anionic polymers tested, as was observed by other authors (Tenney et al., 1969). Knuckey et al. (2006) achieved harvesting efficiencies >80% of marine micoalgae using a non-ionic polymer (Magnafloc LT-25) by adjusting the pH between 10 and10.6 using NaOH. However, pH adjustment was not done in this study.

Although related tests indicated that mucilage extracted from *Opuntia* spp. cactus acted as an efficient coagulant for bacteria and kaolin particles (Buttice et al., 2010), no significant algae recovery was observed with cactus mucilage in this study. Good harvesting efficiency (85%) was observed using *moringa* seed powder, although the dosage needed (4670 mg/L) was much higher than the dose required for other coagulants (Table 2). Teixeira et al. (2012) found that at a *moringa* concentration of 1000 mg/L, *C. vulgaris* recovery increased with increasing pH, reaching a maximum of 89% at a pH of 9.2. *Moringa* contains a coagulant protein, which is effective for treatment of high turbidity waters. It is important to note that natural coagulant tests were done using crude extracts of both cactus and *moringa*, with no pH adjustment. Buttice et al. (2010) used a solvent extraction procedure to produce both gelling and non-gelling extracts of the cactus mucilage rather than the crude preparation used in this study, which is more typical of what is done in developing countries. Further research is needed on the preparation, use and life cycle costs of natural coagulants for algae harvesting.

It is worth noting that periodic auto flocculation was observed in the PBRs several times during the course of this study. During these periods no coagulation tests were performed. This phenomenon had been shown to be associated with a number of mechanisms (Sukenik and Shelef, 1984) including elevated pH due to photosynthetic CO_2 consumption corresponding with precipitation of phosphate minerals, excretion of EOM by algae or associated bacteria, inhibited release of microalgal daughter cells and aggregation between microalgae and bacteria. Further research is needed on whether this could be an effective strategy for algae harvesting.

Bench-scale dewatering tests done by centrifugation at 2000 rpm for 10 min showed that a higher solids content (80%) and greater overall algae recovery was achieved with the cationic polymer than with ferric chloride (71%). Final solids contents achieved using the laboratory centrifuge were much higher than reported with full scale equipment (5–30% solids; Table 3) and can therefore only be used to understand the relative dewaterability of algae slurries produced using different coagulants.

3.2.1. Effect of algae concentration on required coagulant dose

Further tests were performed with ferric chloride and Zetag 8819 at varying initial algae concentrations (Fig. 1a and b). Although data were limited, a linear correlation was observed between algae concentration and optimal coagulant dose, as was observed by other authors (Papazi et al., 2010; Tenney et al., 1969). Wyatt et al. (2012) found that at low algae concentrations (50–120 mg/L), the required ferric chloride dose for >90% recovery increased linearly with algae concentration. In this region, the proposed coagulation mechanism was bridging of algae cells by positively charged ferric hydroxide precipitate "patches" associated with negatively charged algae surfaces. At high algae concentrations (>500 mg/L); however, the authors found that the required ferric chloride concentration was independent of algae concentration, most likely due to a sweep floc mechanism. Tenney et al. (1969) found that cationic polymer dose was linearly correlated with algae concentration at concentrations between 100–350 mg TSS/L.

3.3. Life cycle assessment results

3.3.1. Cost analysis and life cycle assessment of coagulants

Costs, energy use and GHG emissions associated with each coagulant are shown in Fig. 2. Ferric chloride was the most expensive option ($130/MT of algae), followed by alum ($65/MT of algae) and cationic polymer ($50/MT of algae). The low cost of cationic polymer is primarily due to its low required dosage (34 mg/L) compared with ferric chloride (122 mg/L) and alum (140 mg/L). However, it can be seen that using cationic polymer (Zetag 8819) for harvesting will result in the highest GHG emissions, with 92 kg CO_2 eq/MT algae, followed by alum (51 kg CO_2 eq/MT algae) and

Table 2
Summary of harvesting experiments, showing optimal dose for each coagulant.

Coagulant	Optimal dose (mg/L)	Supernatant turbidity (NTU)	Supernatant TSS (mg/L)	Algae recovered (%)
Ferric chloride	122	7.65	15	93
Alum	140	5.40	30	91
Zetag 8819	34	6.05	20	98
Magnafloc E-38	NA	760	>500	~0
Moringa oleifera	4,670	20.0	25	85
Opuntia ficus-indica cactus	NA	740	>500	~0

Table 3
Solids contents before and after dewatering and associated electricity consumption for different dewatering technologies obtained from literature.

Dewatering technology	Solids content before dewatering (%)	Solids content after dewatering (%)	Electricity consumption (KWh/ton dry algae)	References
Dissolved air flotation (DAF)	0.001–0.5%	3–10%	70–1250	Wiley et al. (2011)
Centrifuge	0.1–6%	5–30%	26.5–950	Shelef et al. (1984), Grima et al. (2003), Lohrey and Kochergin (2012)
Solar drying beds	4%	30–40%	250–500	Pandey et al. (2011)
Belt press	0.1–35%	10–90%	400–700	Pandey et al. (2011), Lardon et al. (2009), Shelef et al. (1984)
Vacuum filter	0.10%	30%	5900	Shelef et al. (1984)
Polymer membrane (MF/UF)	0.10%	10%	50–500	Cooney et al. (2010)

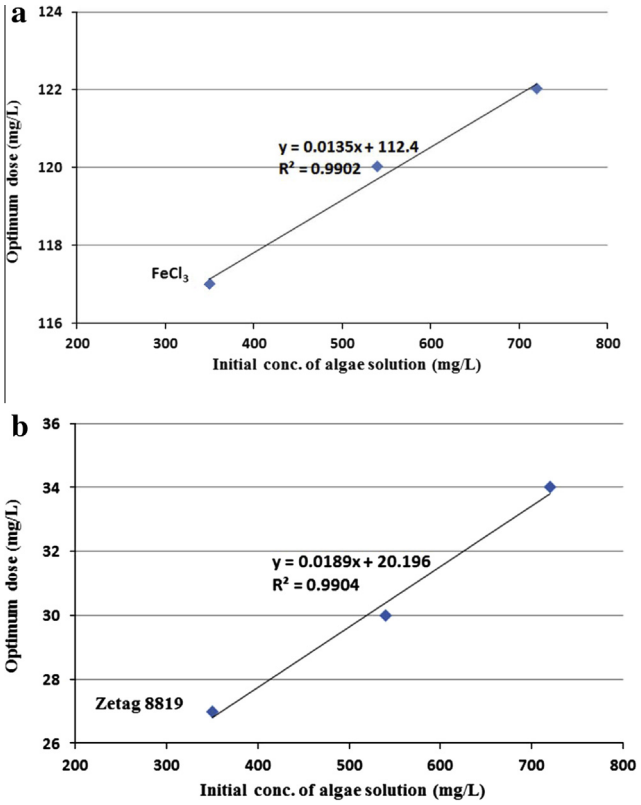

Fig. 1. Optimum dosages of (a) $FeCl_3$ and (b) Zetag 8819 at varying initial algae concentrations.

ferric chloride (13 kg CO_2 eq/MT algae). The energy consumption results are in line with the GHG emissions, showing that polymer has the highest energy consumption (1984 MJ/MT of algae), while ferric chloride has the lowest (204 MJ/MT of algae) among the three coagulants evaluated. High energy consumption and GHG emissions associated with polymer is primarily due to its production process (Karss, 1999). Clearly, there is a tradeoff between environmental impacts and costs of different coagulants. The results show that it is important to consider both economic and environmental aspects to identify the better coagulants.

3.3.2. Life cycle assessment of the dewatering process

Dewatering is employed to increase the solids content of algae slurries prior to downstream processing, such as oil extraction (Lardon et al., 2009). Technologies used for algae dewatering in

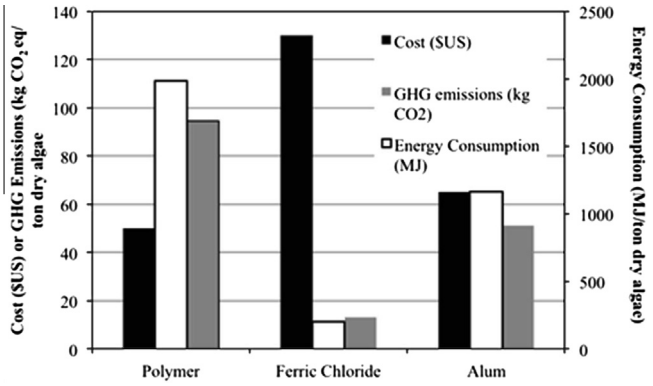

Fig. 2. Cost ($ US), GHG emissions and energy consumption per one metric ton dry algae for cationic polymer, ferric chloride and alum.

prior studies, the solids content before and after the dewatering stage and the electricity consumption per ton of dry algae processed (KWh/ton) are shown in Table 3. The experimental results of this study showed that the solids content of algae before dewatering ranges from 6.4% to 9.2%. Based on solids contents shown in Table 3, the appropriate dewatering technologies include centrifugation, solar drying beds, and belt presses, depending on the desired solids content after dewatering. The GHG emissions and energy consumption per ton of dry algae for these technologies are shown in Fig. 3. Centrifugation has the highest life cycle energy

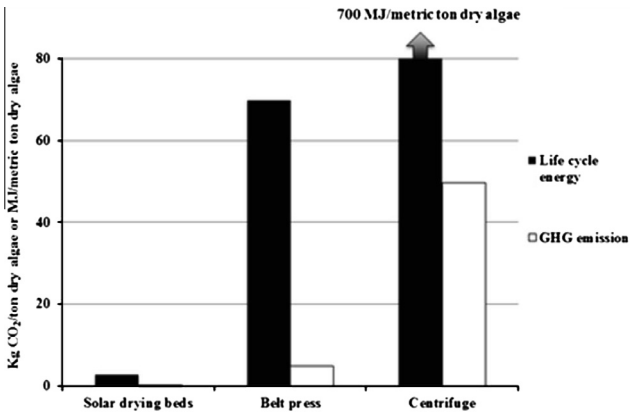

Fig. 3. Comparison of appropriate dewatering technologies from energy and GHG emission aspects.

consumption (700 MJ/dry tone of algae) and GHG emission (50 kg CO_2 eq./dry tone of algae), while solar drying beds have the lowest life cycle energy consumption (2.8 MJ/dry tone of algae) and lowest GHG emissions (0.2 kg CO_2 eq./ dry tone of algae) due to the utilization of solar energy. Solar drying beds; however, can only achieve a solids content of approximately 40%. If a solids content of 90% after dewatering is desired, which is the solid content required for the oil extraction process (Lardon et al., 2009), belt presses are recommended.

4. Conclusions

This study investigated the costs and life cycle impacts of microalgae harvesting for biofuel production. Algae were grown in PBRs with wastewater as the feed source. Several coagulants, including ferric chloride, alum and cationic polymers, could achieve >91% algae recovery in jar tests without pH adjustment. Ferric chloride had the highest cost but the lowest environmental impacts, while the cationic polymer had the lowest cost but the highest environmental impacts. Belt presses are recommended for dewatering because they can meet the solids content requirements for downstream processing with lower energy consumption and GHG emissions than other dewatering technologies.

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