

Review

# A Review of Landfill Leachate Treatment by Microalgae: Current Status and Future Directions

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**Abstract:** Solid waste generation has been projected to increase worldwide. Presently, the most applied methodology to dispose of solid waste is landfilling. However, these landfill sites, over time release a significant quantity of leachate, which can pose serious environmental issues, including contamination of water resources. There exist many physicochemical and biological landfill leachate treatment schemes with varying degrees of success. With an increasing focus on sustainability, there has been a demand for developing eco-friendly, green treatment schemes for landfill leachates with viable resource recovery and minimum environmental footprints. Microalgae-based techniques can be a potential candidate for such a treatment scenario. In this article, research on microalgae-based landfill leachate treatments reported in the last 15 years have been summarized and critically reviewed. The scale-up aspect of microalgae technology has been discussed, and the related critical factors have been elucidated. The article also analyzes the resource recovery potential for microalgal techniques with respect to leachate treatment and explores possible methodologies to minimize the environmental footprints of the microalgae-based treatment process. The future research potential in the area has been identified and discussed.

**Keywords:** landfill leachate; microalgae; resource recovery; leachate treatment; sustainability; resource recovery; phycoremediation; leachate remediation

## 1. Introduction

### 1.1. Status of Municipal Solid Waste (MSW) Generation, Landfilling Options and Landfill Leachate Generation Rate and Composition

According to the US Environmental Protection Agency (EPA), municipal solid waste (MSW) are those waste items that consumers throw away after they are used. These include, among other items, bottles, corrugated boxes, food, electronic items, furniture, office stationery items, etc. In 2015, the total MSW generated in the US was 262.4 million tons, which was approximately 8.4 million tons and 54 million tons more than the years 2014 (254 million tons) and 1990 (208 million tons), respectively. The generation of MSW has consistently increased since 1960 with the exception of 2005 to 2010 when

a 1% decrease was noticed and since then generation has continued to increase by 5% from 2010 to 2015 [1].

Traditionally, most of the MSW are landfilled. However, with the increased focus on integrated solid waste management which includes source reduction of solid waste before they enter the waste stream and resource recovery from generated waste for recycling or composting, the share of landfilling has seen a decline over the years from 94% in 1960 to 53% in 2015. This has led to more recycling (25% of MSW), composting and energy recovery (12.8% of MSW) from MSW. Still, landfilling accounts for disposing of the majority of MSW generated in the US [1].

On a global scale in 2016, the MSW generation is pegged at 2.01 billion tons annually according to a World Bank report; on per person per day basis the MSW generation averages around 0.74 kg with a wide range of 0.11–4.54 kg. By 2050, the global MSW generation is expected to reach 3.4 billion tons. On a global scale too, most solid waste is presently dumped or disposed of in some forms of landfills (37% landfills, 8% sanitary landfills, and 31% open dumping). Therefore, landfilling, due to economic advantages, remains the main mode of waste elimination in the world [2].

MSW after being disposed of in landfills degrade and overtime release gaseous products and liquid waste, the latter is known as landfill leachate. Leachate is a liquid waste flowing out of a landfill due to precipitation (rainfall and snow), ground-water intrusion, moisture content of waste (particularly important when sludge or liquids are disposed of), and rate of evaporation (daily cover during the filling period and final cover design). The leachate stream is generated when soluble components are dissolved or leached out by the percolating water. The volume and composition of leachate generated varies over time. This depends on several factors such as the amount of rainfall, solubility, and moisture content of waste, size of landfill, the quantity of waste contained in it, and the age of landfill sites. An overview of quantities and the range of some physicochemical characteristics from different sites have been summarized in Tables 1 and 2 [3–16].

### 1.2. Landfill Leachate Treatment Methods

Leachate is a potential pollutant with high concentrations of ammoniacal nitrogen, biological oxygen demand (BOD), and chemical oxygen demand (COD). The chemical composition of the landfill depends on multiple internal and external factors. Specifically, the age of a landfill plays a vital role in the composition of the landfill leachate [4,8–10]. For example, young landfill leachates (< 1-year-old) contains high concentrations of COD ( $> 15 \text{ g L}^{-1}$ ) and relatively small concentrations of ammoniacal nitrogen ( $< 400 \text{ mg L}^{-1}$ ). In contrast, stabilized landfill leachates ( $> 5$  years old) contains a high concentration of ammoniacal nitrogen ( $> 400 \text{ mg L}^{-1}$ ) and a very low concentration of COD ( $< 3 \text{ g L}^{-1}$ ) [10]. It poses a contamination threat to groundwater and surface water. The pollution arising out of leachate contamination poses a threat to human health, aquatic organisms, and ultimately to the environment. Therefore, the leachate cannot be discharged into the environment and must be treated. To safeguard the environment, the regulatory authorities review environmental standards regularly, and these are becoming ever stricter. Various methods used for leachate treatment reported in the literature have been summarized in Table 3 along with their advantages and disadvantages.

Current treatment technologies are mainly focused on the reduction/removal of nutrients and COD [16]. However, there is a growing need for advanced sustainable treatment technologies focused on resource recovery with minimum environmental footprints which can lead to the removal of contaminants of emerging concern including pharmaceuticals and personal care products (e.g., perfluorinated chemicals (PFCs)). Therefore, it is vital to develop sustainable, cost-effective on-site leachate treatment technologies focusing on near and longer-term contaminant issues.

Algal-based methods are ecological, low-cost and relatively novel alternatives to conventional aerobic and anaerobic treatment methods. Algae are abundantly found in nature and can grow even in adverse climatic and ecological conditions while removing nutrients and toxic metals by biosorption and bioaccumulation [17]. To promote sustainability and resource recovery, the algal approach can also lead to the production of economically valuable biomass that can be utilized for producing biofuels [18],

bioactive compounds, and nutrient supplements [19]. Additionally, algal approach in certain systems such as microbial fuel cells (MFC) offer distinct advantages, for example, in MFC cathode algae is involved in carbon fixation via photosynthesis which leads to the generation of higher dissolved oxygen (DO) than mechanical aeration and, thereby the system operates at a higher efficiency [20].

The literature survey reveals that many review articles are available for leachate treatment technologies, however, algal-based treatment methods have not been covered adequately. The applications of algae in treating industrial and municipal wastewater are fairly reported. However, only a few pieces of literature have reported landfill leachate treatment in the past 15 years. As this area of research needs to grow as it leads to the environmental and economic sustainability of waste treatment, a review article summarizing the available results is necessary. Therefore, the current review focuses on following key areas: 1) summary of the progress and results achieved so far in the field, with emphasis on what species have been utilized so far, how effective is single culture vs. multi-culture vs. algal-bacterial consortium and pretreatment methods; 2) analysis of the process parameters that affect the performance of landfill leachate treatment using microalgae. Furthermore, a critical study has been conducted to comment on as to which process parameters are central to scaling up and enhancing the rate kinetics of the algal-based treatment schemes; 3) resource recovery potential and minimization of environmental footprints of the algal-leachate systems.

**Table 1.** Leachate generation.

Location	Type of Leachate	Quantity	Area	Reference
Keele Valley Landfill, Ontario, Canada	Intermediate (8 years); MSW	75–342 m <sup>3</sup> day <sup>-1</sup>	99 hectares	[3]
Ouled Fayet Site, Algiers, Algeria	Young (5 years); MSW	47 m <sup>3</sup> day <sup>-1</sup>	40 hectares	[4]
19 in the Northern part of Western Germany; 1 in Bern, Switzerland	20 sites all with < 12 years of age; MSW	0–24 m <sup>3</sup> hectares <sup>-1</sup> day <sup>-1</sup>	-	[5]
Southeast US	Intermediate; coal combustion residual landfill leachate	20,000–60,000 liters hectare <sup>-1</sup> cm <sup>-1</sup> of the rainfall between November 2010 to May 2016; 0–7.4 million liters per month in this period 16.4–26.8 m <sup>3</sup> per day during dry seasons and 27.8–36.6 m <sup>3</sup> per day in wet seasons in 2001; the corresponding figures were 8.1–9.1 m <sup>3</sup> per day in dry seasons and 11.1–12.1 m <sup>3</sup> per day during wet seasons in 2010	7.9 hectares	[6]
Ampang Jajar Landfill Site, Malaysia	10 years old semi-aerobic landfill; MSW		2.9 hectares	[7]

**Table 2.** Leachate composition with landfill age.

Components	Young (<1 years)	Intermediate (1–5 years)	Stabilized (>5 years)	Reference
pH	< 6.5; 7.6–8.5	6.5–7.5; 6.8–8.4; 8.27	> 7.5; 7.4–7.6	[4,8–10]
Specific conductivity ( $\mu\text{S cm}^{-1}$ )	3089–28430	2606–10440; 17870–41500; 28560	3870–4120; 6380–15030	[8,9,11]
Total solids ( $\text{g L}^{-1}$ )		1.14	2.027–2.267	[4,8]
<b>Organic matter (<math>\text{g L}^{-1}</math>)</b>				
BOD <sub>5</sub>	0.036–0.984	0.006–0.033; 0.98	1.5–3	[4,8,9]
COD	> 15; 0.411–7.16	3–15; 0.19–0.748; 1.5428–7.1253;	< 3; 10.4–12; 0.6952–2.424	[4,8–11]
		3.792		
BOD <sub>5</sub> /COD ratio	0.5–1	0.1–0.5	< 0.1	[10]
<b>Inorganic components (<math>\text{mg L}^{-1}</math>)</b>				
Total phosphorus		17.97–34.9; 58.22	0.62–14.83	[4,11]
Chloride	160–2620	130–669; 4569	660–780	[4,8–10]
Alkalinity	998–9682	10–2100; 8049–18,162	1754–5573	[9,11]
Sulphate	7.2–1950	21–445; 3056	40–42	[4,8,9]
Ammonium–N	< 400; 130–4000	63–378; 1564.2–4251; 85.805	> 400; 1803–2593	[4,8–11]
Nitrate		14.59	22.36–35.09	[4,8]
Cyanide	0.006–1.164	0.006–0.081		[9]
Total Nitrogen	120–4027	120–1083; 1753.5–4368.2	428.2–1489.8	[9,11]
Iron		8.32	11.16–12.04	[4,8]
<b>Heavy Metals (<math>\mu\text{g L}^{-1}</math>)</b>	> 2000	< 2000	< 2000	[10]
Arsenic	11–412	14.6–155		[9]
Cadmium	0.1–7.4	0.1–1.6; < 30	24–35	[4,8,9]
Chromium	3.3–1436	28–284; 200	1.1–2.4	[4,8,9]
Cobalt		85		[4]
Copper	3–2423	11–157; 390	2–151	[4,8,9]
Lead	0.6–1047	0.9–8.2; 3490	220–300	[4,8,9]
Mercury	0.02–1.07	0.02–2.05		[9]
Nickel	10–661	22–151; 370	683–1339	[4,8,9]
Zinc	10–7639	10–303; 1430	2.4–3	[4,8,9]
Silver	10–2187	10–280	35–121	[8,9]

Table 3. Existing leachate treatment methods.

Method	Scheme	Advantages	Disadvantages	References
Leachate transfer	Leachate treatment in conjunction with domestic sewage Leachate recycling (leachate is recycled back to the landfill)	Low operation cost and easy maintenance Mutually complements nutrients needed for treatment; nitrogen (contributed by leachate) and phosphorus (contributed by sewage) and thereby prevents additional requirements Leachate recycling increases the moisture content of the waste; provides the distribution of nutrients between methanogens and solid/liquids phases; improves the leachate quality and shortens the stabilizing time from several decades to 2–3 years	For effective treatment, maintaining the ratio of leachate and sewage is critical Presence of low biodegradable organic compounds and heavy metals in the leachate reduces the process efficiency Only a small fraction of the leachate can be recycled without hampering the performance of methanogens	[12–14]
Biological	Aerobic schemes Anaerobic schemes Both schemes further classified as suspended growth (lagooning activated sludge & sequencing batch reactor (SBR) and attached growth (trickling bed filters, moving bed, suspended carrier biofilm reactors) biomass systems	Simple operation & low maintenance (aerated lagoons) Less affected by frequent organic load or ammonium nitrogen variation (SBR) Immobilization of active biomass and nitrification at low temperatures (attached growth systems) Ability to treat high strength organic effluents, COD >10 g L <sup>-1</sup> (anaerobic systems) Sulfide removal and good elimination of all the pollution parameters	Susceptibility of micro-organisms to heavy metals, high pH and ammonia toxicity (lagoons, annamox) High ammonia, sulfide, and methane in the effluent (anaerobic schemes) High capital cost Inadequate sludge settling Longer aeration time needed (lagoons) Sludge bulking & poor clarification (SBR) Requires skilled personal and regular monitoring	[12]
Physical/Chemical	Chemical precipitation Coagulation/flocculation Chemical oxidation processes Air stripping Adsorption Membrane processes Ion-exchange systems Electrochemical processes Flotation systems	Useful for removing non-biodegradable and refractory components of leachates, targeted pollutant removal and as post or pre-treatment step in biological methods Effective in removing volatile organic compounds (VOCs) but limited success in removing COD (air stripping) Adsorption using activated carbon can be used to enhance nitrification and sludge dewaterability during biological treatment. Electrochemical systems show promising results particularly for non-biodegradable systems	Limited COD removal (air stripping, coagulation/flocculation) High cost of chemical required, sludge handling and waste disposal Additional ammonia control required for exiting air (air stripping); calcium carbonate scaling in the tower Cannot provide full treatment for the wide range of contaminants found in leachates Adsorption/ion-exchange systems require regeneration at regular intervals Membrane processes suffer from fouling of organic/suspended matter and high energy requirements. High installation and maintenance costs.	[10,15,16]

## 2. Algal-Based Landfill Leachate Treatment: Current Approach and Results

The application of algae to remove contaminants from waste streams is termed as phycoremediation [21]. The algal approach for treating wastewater has been widely reported in the available literature. However, phycoremediation techniques are relatively novel in the field of landfill leachate treatment. Since 2007, 22 research articles were found to focus on the leachate treatment utilizing microalgae. These studies were mostly performed in either lab scale or pilot scale. There exist certain limitations that must be overcome through research for the large-scale implementation of phycoremediation of landfill leachate.

### 2.1. Lab-Scale Studies

These studies were largely performed in a lab setting in batch mode with treated volume ranging from 15 mL to 12.5 L. For example, Lin et al. (2007) [22] utilized 3 algal strains: *Chlorella pyrenoidosa*, *Chlamydomonas snowiae* (both isolated from high ammonia leachate pond), and *Chlorella pyrenoidosa* (isolated from a clean river) to treat diluted landfill leachate samples (10%, 30%, 50%, 80%, and 100% concentration of leachate) in a lab-scale study. The study reported high algal growth for *C. pyrenoidosa* (for both the strains) and moderate growth for *C. snowiae*. Inhibitory effect of high leachate concentration on the algal growth was seen (50% leachate and beyond for *C. pyrenoidosa*, and 30% leachate and beyond for *C. snowiae*). The key reasons for growth decline were high ammoniacal concentration ( $> 670 \text{ mg L}^{-1}$ ) and depletion of carbon dioxide which increased the pH and led to the volatilization of  $\text{NH}_3$ . The algal strain *C. pyrenoidosa*, isolated from leachate pond, showed ammonia tolerance to an extent. Edmundson and Wilkie (2013) [23] studied the growth of *Scenedesmus cf. rubescens* and *Chlorella cf. ellipsoidea* in 100% landfill leachate (LL) and compared it with the respective growth rate in Bold's Basal media (BBM) and HCl adjusted LL. Both the species grew in BBM and HCl adjusted LL with *Scenedesmus cf. rubescens* showing higher growth rate and biomass production than *Chlorella cf. ellipsoidea*. The study suggests the role of pH in the acidic range is critical to the favorable algal growth, as unregulated pH makes the media alkaline which subsequently favors free  $\text{NH}_3$  production in the media, making it toxic to algal growth. A system dynamic approach using a simple microalgae-detritus model to estimate the heavy metal removal, lipid formation, and algal growth rate in leachate-hypersaline water was explored by Richards and Mullins [24]. To study the impact of temperature the model included a skewed-normal distribution for temperature. The four species of algae studied were *Nanochloropsis gaditana*, *Pavlova lutheri*, *Tetraselmis chuii*, and *Chaetoceros muelleri*. The experimental data showed that after 10 days 95% of the metals were removed from the solution with *N. gaditana* and *C. muelleri* showing the highest growth rate and lipid content. The study did not discuss COD/BOD and ammonia removal.

Nevertheless, Zhao et al. (2014) [25] conducted a lab-scale study (500 mL volume) to treat municipal wastewater spiked with 0%, 5%, 10%, 15%, and 20% LL using a bacteria-microalgal consortium with *C. pyrenoidosa* as the algal strain. The study noted that the maximum biomass concentration and specific growth rate were observed for 10% LL ratio. The consortium showed a 6-day adaptation period for a 20% LL ratio, suggesting the existence of an initial inhibitory effect on the algal growth. The optimum spiking ratio reported was 10% LL with initial  $\text{NH}_4\text{-N}$  as  $183 \text{ mg L}^{-1}$  and 99%  $\text{NH}_4\text{-N}$  final removal. Although the biological uptake of ammonia was 52%, with the remaining removal taking place due to stripping by aeration. The study also suggested the possibility of bacterial breakdown of various nitrogenous compounds into forms that can be utilized by the microalgae as feed. However, the role of bacteria was suspected in lowering the lipid content of the biomass in the range of 14.5% to 20.8% which for *Chlorella vulgaris* has been reported to be 30% as the maximum in another study. Another synergistic lab-scale study involving microalgae and the bacterial consortium was carried out by Kumari et al. (2016) [26] where a *Scenedesmus* species were grown together with a *Paenibacillus* bacterial strain in 20% (v/v) leachate sample. The study reported the removal of toxic contaminants such as heavy metals and organic compounds with a significant reduction in cytotoxicity and genotoxicity of the treated leachate.

In addition, Paskuliakova et al. (2016) [27] studied 34 algal strains obtained from marine, freshwater and polluted environments for their growth and phycoremediation performance in raw, treated leachate from the membrane bioreactor plant and permeate leachate (25–50% strength). Sixteen strains, mostly belonging to freshwater sites, were able to survive and/or grow in the leachate samples. Most marine species were not able to survive at any magnitude of leachate dilution with an exception of *Tetraselmis* sp. SW01cMA which showed growth at 25% leachate dilution with seawater and an unidentified chlorophyte strain OT03aMA which grew in treated leachate. The study also highlighted that the species that were isolated from landfill sites performed better in leachate stress studies and not so well in other media. This indicates a criterion for selecting algal strain based on the waste streams to be treated. Paskuliakova et al. (2016) [28] evaluated 4 algal strains (*Chlamydomonas* sp. (strains SW13aLS and SW15aRL) and *Scenedesmus* sp. (strains OT11aTL and OT08aTL)) for phycoremediation of 10% permeate and 10% raw leachate at relatively low temperature (15 °C) and light intensity (14:10 h, light: dark, 22  $\mu\text{mol m}^{-2} \text{s}^{-1}$ ), conditions akin to temperate climatic conditions. For 10% permeate the algal strains grew very slowly and insignificantly; in 10% raw leachate only *Chlamydomonas* sp. SW15aRL showed major growth, which further achieved higher growth on supplementing with phosphorus (8.4-fold and 12.1-fold increase in biomass for phosphate non-supplemented and supplemented studies, respectively). However, total oxidized nitrogen (TON) reduced for all the strains in the case of 10% permeate, with a most pronounced decrease was noticed for *Chlamydomonas* sp. SW13aLS (mean 41.4% and 41.8% in phosphate supplemented). Total ammonia nitrogen (TAN) removal in 10% raw leachate was 22.4% and 16.6% for phosphate non-supplemented and supplemented, respectively. *Chlamydomonas* sp. SW15aRL showed significant TAN removal of ~90% by day 30. Pereira et al. (2016) [29] studied the growth of *C. vulgaris* in three different assays of pre-treated landfill leachate samples collected at the exit of an aerated stabilization pond. The microalgal growth was compared to three different N:P ratio (12:1, 23:1, and 35:1) obtained by externally adding  $\text{KH}_2\text{PO}_4$ ; also, a comparison was made for the case when no external P was added. The performance was evaluated in terms of microalgal growth, nutrient, and heavy metals removal. *C. vulgaris* showed growth in all the assays with biomass productivity varying between 0.02–0.11  $\text{g L}^{-1} \text{d}^{-1}$ . The growth rate was not significantly lower for the cases when no P was added externally. Up to 77%  $\text{NH}_4^+ \text{-N}$  removal was observed in all the cases, whereas  $\text{NO}_3^- \text{-N}$  concentration remained constant throughout (between 21–27%). The nutrient removal was observed to be enhanced by the addition of P, however, the study failed to meet the discharge standards prescribed by the European Union.

In a recent study, Tunisian landfill leachate was treated using *Chlorella* sp. in a study by El Ouaer et al. (2017) [30]. The leachate's impact on the microalgal growth and its capacity to remove COD (60% removal in 13 days) and ammoniacal nitrogen (90% removal in 13 days) was evaluated by varying the incubation time and leachate concentration. In 10% leachate concentration the biomass grew to 1.2  $\text{g L}^{-1}$  in 13 days which was 1.8 times the growth observed in Bold's medium. Lipid content showed a higher value in 10% leachate (4.74  $\text{g L}^{-1} \text{d}^{-1}$  as compared to Bold's medium (3.09  $\text{g L}^{-1} \text{d}^{-1}$ )). The study showed that 10% leachate can be used as a growth medium for the microalgal growth and bioremediation of COD and nitrogen. With 100% raw leachate, the microalgae were able to remove 50.7% COD and 90%  $\text{NH}_4^+ \text{-N}$  in 24 days. For leachate concentration greater than 10%, a decrease in cell density was noticed. Nguyen et al. (2017) [20] utilized microalgae in a microbial fuel cell (MFC) for COD and nutrient removal and energy production using a 5–40% strength leachate medium. The application of microalgae in MFC offers operational advantages such as the evolution of higher dissolved oxygen (DO) produced by the photosynthetic process of the microalgae and generate more power output than an MFC operation with mechanical aeration. The DO level obtained was twice the solubility level of  $\text{O}_2$  in water at 30 °C using microalgae isolated from domestic wastewater. In the study, MFC operation began with domestic wastewater during the start-up phase and then shifted to a leachate medium (10%) for algae growth. The study reported max DO increase up to  $19.57 \pm 3.04 \text{ mg L}^{-1}$  for 5% leachate and max voltage up to  $300 \text{ mV} \pm 11 \text{ mV}$ . In the anode chamber, 96.8% COD removal was observed and 52.9% in the cathode chamber.  $\text{NH}_4^+ \text{-N}$  concentration of  $300 \text{ mg L}^{-1}$  was completely removed in

4 days and total phosphorus (initial concentration: 3.01–12.71 mg L<sup>-1</sup>) reduced by 61.46% in 5 days. DO level showed inverse relationship with leachate concentration, also the lag period of oxygen generation increased with increasing leachate percentage. The cell voltage increased with leachate concentration up to 15% and beyond, there was no change in voltage generation; for 15–40% leachate, max voltage obtained was 163 ± 10 mV. However, the study did not discuss the growth of microalgae or how the dead biomass/sludge would be disposed of. The study also did not discuss any scale-up aspect or how the process parameters would vary with sizing up the reactor. This would require fundamental understanding and formulation of process parameters with the MFC performance.

To extend further on the lab-scale study, Nordin et al. (2017) [31] studied 3 algal strains (*Chlorella* sp., *Scenedesmus* sp. and *Oscillatoria* sp.) for their growth rate and nitrate removal in 10–30% strength nitrified landfill leachate. For *Chlorella* sp., *Scenedesmus* sp. and *Oscillatoria* sp. max biomass production noted were 428.66 mg L<sup>-1</sup> (in 20% leachate), 237.04 mg L<sup>-1</sup> (in 30% leachate) and 805.96 mg L<sup>-1</sup> (in 20% leachate), respectively. The corresponding nitrate removal was 77.4% for *Chlorella* sp. (in 20% leachate with initial nitrate concentration at 480.34 mg L<sup>-1</sup>), 66.34% for *Scenedesmus* sp. (in 20% leachate with initial nitrate level at 456.45 mg L<sup>-1</sup>). It was noted that microalgal growth directly impacted the nitrate removal rate. The study reported that *Oscillatoria* sp. has the fastest and highest biomass production and productivity, which makes the species useful for rapid and large-scale biomass production. Moreover, *Oscillatoria* sp. has the highest carbohydrate, lipid, and protein productivity among *Chlorella* sp. and *Scenedesmus* sp. As mixing conditions, DO, temperature, surface area to volume ratio and the inoculation of wild bacterial/algal culture can impact the real-life application of algal-based remediation methods, therefore, small-scale, controlled lab studies seldom provide results which can be representative of real conditions. Casazza et al. (2018) [32] studied the use of microalgae (*C. vulgaris*) to reduce the nitrogen content from leachate both before and after nitrification. Leachate was microfiltered to remove suspended solids. Ammonium removal of 38.8 mg L<sup>-1</sup> d<sup>-1</sup> was observed with the microalgae growing and aggregating lipids. Chang et al. (2018) [33] utilized a membrane photobioreactor to treat raw leachate using *C. vulgaris*. In this method, a pretreatment step is avoided by utilizing an ion-exchange membrane barrier between the microalgae and the leachate which allowed ammonium ions at such a concentration and rate that it never reached the concentration to cause toxicity but at enough concentration and rate that encouraged algal growth; also the permeate water was clearer and encouraged higher photosynthetic activity. The performance of the membrane bioreactor (MBR) was compared with a traditional bioreactor (TBR); in MBR the algae reached a max biomass concentration of 0.95 g L<sup>-1</sup> as compared to TBR where the corresponding value was 0.66 g L<sup>-1</sup>. Nitrogen reclamation was ~50% and phosphorus reclamation was ~70%. The lipid produced by MBR was of superior quality than TBR with a high cetane number of 60.96% and low linolenic acid content of 8.32%, making the process highly valuable from a biofuel production perspective. However, the study lacked in terms of discussion on the effluent quality of the treated leachate with respect to COD, BOD, pH etc., which is critical for scaling up and final disposal. Paskuliakova et al. (2018) [34] studied the growth and nutrient removal capacity of *Chlamydomonas* sp. strain SW15aRL in six different leachate samples obtained either from different landfill sites or sampled from different location and time within the same site with leachate concentration varying between 10–100%, such that total ammonia nitrogen remained within the range of 30–220 mg L<sup>-1</sup>. The strain showed growth in a variety of leachate conditions with a dependence factor rather on the overall composition profile of the leachate than just on the dilution factor. The leachate samples were phosphate supplemented as K<sub>2</sub>HPO<sub>4</sub> to achieve a molecular ratio of 16:1 N:P in the final volume. The algal strain showed variable growth in a variety of leachate composition depending on nutrient availability and heavy metals concentrations. The effect of dilution is shown not only to reduce ammonia toxicity but also to lead to the reduction of key minerals required at certain concentrations for algal growth leading to bioremediation. Ammonia nitrogen decreased between 70–100% with maximum biomass growth of 1.2 g L<sup>-1</sup> and a specific growth rate of 0.19 d<sup>-1</sup>. In another study, Paskuliakova et al. (2018) [35] utilized the same algal strain to study its growth and nitrogen removal capacity in leachate with concentration varying between 30–100%. The study

particularly explored the impact of the forms of phosphorus compound used as P supplements; the two compounds included in the study were  $K_2HPO_4$  and  $H_3PO_4$ . The study concluded the moderate difference between the obtained biomass of the algae at the end of the experiment with two different P compounds, however, a major difference was observed with respect to the maximum number of cells achieved ( $H_3PO_4$  showed higher response) but no significant difference between the growth rate. No major difference was seen for ammonia nitrogen removal with ~98% removal in 30–35 days. Maximum biomass growth reached up to  $1.2\text{ g L}^{-1}$  with lipid content of 10–18% of the dry weight of the strain. Elemental analysis was also performed to gain insights into how much nitrogen was incorporated into biomass and other aggregates obtained by centrifugation. The study noted that with  $K_2HPO_4$  higher loss of ammonia occurred due to volatilization. A similar nitrogen balance study was performed by Sniffen et al. (2018) [36] using an algae-based leachate remediation system in both lab-scale (250 mL reactor volume) and pilot-scale mode (100 L and 1000 L reactor volume). The algae-bacteria consortium was used with two hypotheses that all nitrogen is incorporated into the algal biomass and in the absence of biomass growth no nitrogen removal occurred in the liquid phase. In a 205-week study, it was reported that only 16.6% (34 weeks) of the time the nitrogen balance was achieved i.e., the total amount of nitrogen in liquid and solid phases at the beginning of the week was equal to the total nitrogen in liquid and solid phases at the end of the week; 25 out of these 34 weeks showed biomass growth whereas in 9 weeks biomass loss was observed. In another scenario, it was reported that 83.4% of the time (171 out 205 weeks), the amount of nitrogen in liquid and solid phases at the beginning of the week did not equal the total nitrogen in liquid and solid phases at the end of the week; in this scenario, 110 out 171 weeks showed positive biomass growth and only 61 weeks showed biomass loss. In the third scenario, it was noted that the consumption of nitrogen for biomass growth is not the only nitrogen conversion mechanism in the system. This occurred for 80 out of 205 weeks. The study found the existence of multiple pathways for nitrogen removal from leachate using microalgae and suggested more research to identify all the pathways. The understanding of these pathways can pave a robust plan for scale-up operations.

## 2.2. Pilot-Scale Studies

Phycoremediation techniques have not been much amenable to scale-up and have thrown inconsistent results when compared to the corresponding lab-scale studies. Therefore, to account for discrepancies between controlled studies and large-scale applications Sniffen et al. (2017) [37] conducted a comparative study at three scales—small-scale (0.25 L), medium (100 L), and large (1000 L). The bacterial-algal consortium obtained from a fishpond was utilized. The small-scale experiment was conducted in flasks as batch experiments, the medium and large-scale experiments were carried out as semi-batch processes within a greenhouse in an uncontrolled setting with working volume 60% of their total volume. Kolmogorov–Smirnov statistical tests were applied to the experimental data to determine if the ammonia removal, total nitrogen removal, and biomass growth rate at each scale were different. The study shows that there is a significant difference between all rate determined at the large-scale reactors as compared to that of small-scale reactors. It is concluded that small-scale experiment results may not be accurate as inputs to predict the performance of full-scale algal processes. The study also reported that both nitrogen removal and biomass growth increased with warmer temperatures. Velasquez et al. (2012) [38] studied the impact of mixing landfill leachate with the facultative lagoon wastewater treatment plant (FLWTP) waste influent. The influent contained low organic loading and high salinity. The microalgae present in the system were *Microcystis* sp., *Euglena* sp., *Scenedesmus* sp., *Chlorella* sp., *Diatomea* sp., and *Anacystis* sp. and they were evaluated for their BOD, COD, heavy metal removal performance along with their growth rates in varying leachate ratio (4%, 6%, and 10%) under saline conditions. The study reported no impact of mixing leachate with wastewater on BOD and COD removal rates, though both BOD and COD increased due to its addition. The study also suggested the inhibitive effects of salinity on algal density and chlorophyll concentration. Mustafa et al. (2012) [39] assessed the ability of high rate algal ponds (HRAPs) to meet the Malaysian

discharge standards when treated with a consortium of microalgae; the study compared a 5-species consortium including *C. vulgaris*, *Scenedesmus quadricauda*, *Euglena gracilis*, *Ankistrodesmus convolutus*, and *Chlorococcum oviforme* with a consortium of natural lake algae. The individual strain was also observed for their growth in the treated leachate (TL) whose leachate concentration varied as 0%, 25%, 50%, 75%, and 100%. The species *S. quadricauda*, *E. gracilis*, and *A. convolutus* showed growth in up to 50% TL; *C. vulgaris* grew in up to 25% TL, whereas no growth was observed for *C. oviforme* in the study. The consortium was then grown in a 40 L pilot-scale HRAP with 4% TL fed daily in a semi-continuous mode. The consortium was noted to be more tolerant than each species in the TL. Reduction in COD,  $\text{NH}_4\text{-N}$ , and  $\text{PO}_4\text{-P}$  was observed with a 98.73% reduction in  $\text{NH}_4\text{-N}$  at a high loading rate of 4% TL.

Martins et al. (2013) [40] studied a pilot-scale landfill leachate treatment system utilizing a wild consortium of bacteria and microalgae where a series of three ponds and a rock filter was operated for 111 weeks with a continuous flow rate of leachate as 200 L per day. The study reported three different operation phases (I: conventional operation, II: aeration and III: aeration/recirculation) where COD removal and ammonia removal varied between 35–82% and 75–99%, respectively. The study also estimated the nitrogen removal pathway and noted that 64–79% nitrogen was found in dead/inert algae settle, 12–27% in volatilized ammonia and 1–6% assimilated by the algae. *Chlamydomonas* was the most dominant genera in the pond. The same system was utilized by Costa et al. [41] for 43 weeks and 75% BOD and ammonia removal were reported with ammonium concentration remaining below  $6 \text{ mg L}^{-1}$  in the effluent.

A pilot-scale study with 114-litre Plexiglas tanks (working volume 57 L) in an open, outdoor setting, and a semi-batch mode with a retention time of 3 weeks was conducted by Sniffen et al. (2016) [42]. The growth media used were COMBO media (in tank 1; for first 9 weeks COMBO media and then tank 1 was switched to receive a leachate nutrient feed for the remaining period up to week 22) and raw landfill leachate diluted to 5–20% using tap water (in tank 2; fed with leachate throughout the experiment). At the end of each week, 19 L was withdrawn for testing and compensated by equal volume by untreated leachate/fresh growth media. The study utilized a mixture of algae–bacteria inoculum obtained from an outdoor pond. The impacts of varying environmental factors (temperature and sunlight) and leachate composition in terms of fluctuating ammonia concentrations, organic particulates, heavy metals, and other inhibitory constituents were studied on nutrient removal and biomass growth. It was reported that ammonia removal positively and linearly correlated with the initial ammonia concentration, with max removal of  $8.43 \text{ mg N-NH}_3 \text{ L}^{-1} \text{ day}^{-1}$  was observed for  $80 \text{ mg N L}^{-1}$ . Beyond  $80 \text{ mg N L}^{-1}$ , the ammonia removal rate declined rapidly (though no impact on biomass growth was noted); this inhibiting concentration was found to be 2–3 times greater than previously reported values. Overall, it was noted that the ammonia removal depended on initial ammonia concentration, initial biomass concentration, and the average number of daylight hours. The biomass growth trended positively with ammonia removal rate with max biomass density reaching  $480 \text{ mg biomass L}^{-1}$ . However, the amount of nitrogen lost by the media on a weekly basis could not be matched with the biomass assimilation. As the pH never exceeded beyond 8, therefore the only insignificant amount of nitrogen loss can be attributed to ammonia volatilization.

The available literature shows that there exist multiple challenges and limitations which must be resolved before algal-based methods can be utilized for leachate treatment on a large-scale. The results of the available studies are summarized in Table 4.

**Table 4.** Current challenges and limitations for large-scale development of algal-based methods.

Leachate Characteristic (Source/Age)	Algal Species (Group)	System & Operation Mode	Treatment Condition pH/Leachate Conc/Temp/Time/Luminescence	Initial NH <sub>4</sub> <sup>+</sup> (mg L <sup>-1</sup> )/% Removal	Initial P (mg L <sup>-1</sup> )/% Removal	Initial COD (mg L <sup>-1</sup> )/% Removal	Biomass Cell Density (g L <sup>-1</sup> )/ Specific Growth Rate (μ = day <sup>-1</sup> )	Ref
Filtered raw LL > 12 year	<i>Chlorella pyrenoidosa</i> (isolated from LL) (Green algae)	Lab scale, 500 mL flask, batch mode	7.8/10–100%/25–30 °C/12 days/2000–3000 lux	400 mg L <sup>-1</sup> 75% removal	1.5 mg L <sup>-1</sup> 70% removal	384 mg L <sup>-1</sup> 75% removal		[22]
Saline LL	Native consortium: <i>Microcystis</i> sp., <i>Merismopedia</i> sp., <i>Euglena</i> sp., <i>Scenedesmus</i> sp., <i>Chlorella</i> , <i>Diatomea</i> , and <i>Anacystis</i> sp.	Pilot scale, 480 L facultative lagoon reactors, continuous mode	8/4%, 6%, 10%/ambient nature/110 days/natural			711 mg L <sup>-1</sup> 35% removal		[38]
Treated LL	<i>Chlorella vulgaris</i> , <i>Scenedesmus quadricauda</i> , <i>Euglena gracilis</i> , <i>Ankistrodesmu scovolutus</i> and <i>Chlorococcum oviforme</i> <i>Scenedesmus</i> cf. <i>rubescens</i> ,	Pilot scale, 40 L HRAPs, semi-continuous mode, feed daily	7–9/1%, 2%, 4%/ambient natural, 27–30 °C/ 50, 100, 200 days/natural, 1007–1160	3–152 mg L <sup>-1</sup> 92–98.38%	0.2–8.2 mg L <sup>-1</sup> 45–86%	860–3040 mg L <sup>-1</sup> 71–91%	2–5 g L <sup>-1</sup>	[39]
Methanogenic-stage LL	<i>Chlorella</i> cf. <i>ellipsoidea</i> (Green algae)	Lab scale, 125 mL flask, batch mode	6.8–7/100%/25 °C/3–4 days/6522 lux	980 mg L <sup>-1</sup>	13.2 mg L <sup>-1</sup>	-	0.285, 1.33 g L <sup>-1</sup> 0.83, 0.67 day <sup>-1</sup>	[23]
Raw LL > 20 year	Consortia of microalgae and bacteria	Pilot-scale, 20.33 m <sup>3</sup> stabilization pond system, continuous mode 200 L day <sup>-1</sup>	9–10/100%/15–28 °C/111 weeks/nature	805–1510 mg L <sup>-1</sup> , 75–99%	-	1670–2474 mg L <sup>-1</sup> , 35–82%	-	[40]
Filtered LL	<i>Nanochloropsis gaditana</i> , <i>Pavlova lutheri</i> , <i>Tetraselmis chunii</i> , and <i>Chaetoceros muelleri</i> (Marine algae)	Lab-scale, 2.5 L–12.5 L cylindrical photo-bioreactors	26.1–34.6 °C	-	-	-	-	[24]
Raw LL > 20 year	Consortia of microalgae and bacteria	Pilot scale, 20.33 m <sup>3</sup> stabilization pond system, Continuous mode 250 L/day	9–9.6/100%/17–25 °C/ 43 weeks/ nature	845–1342 mg L <sup>-1</sup> , 70–82%		1313–1789 mg L <sup>-1</sup> , 42–48%	-	[41]
Filtered LL	<i>Chlorella pyrenoidosa</i> +bacteria (Green algae)	Lab scale, 500 mL flasks, batch mode	0–20%/ 25 °C/12 days/8000 lux	183(222) mg L <sup>-1</sup> , 95(90)%	3 mg L <sup>-1</sup> , 95%	60 mg L <sup>-1</sup> , negative	1.58 g L <sup>-1</sup> , 0.28 μ: day <sup>-1</sup>	[25]

Table 4. Cont.

Leachate Characteristic (Source/Age)	Algal Species (Group)	System & Operation Mode	Treatment Condition pH/Leachate Conc/Temp/Time/ Luminescence	Initial NH <sub>4</sub> <sup>+</sup> (mg L <sup>-1</sup> )/% Removal	Initial P (mg L <sup>-1</sup> )/% Removal	Initial COD (mg L <sup>-1</sup> )/% Removal	Biomass Cell Density (g L <sup>-1</sup> )/ Specific Growth Rate (μ = day <sup>-1</sup> )	Ref
LL from Self-making lysimeter	<i>Scenedesmus</i> sp. + <i>Paenibacillus</i> sp. (Bacteria) Green algae	Lab-scale, flasks, batch mode	30 °C/10 days	-	-	6000 mg L <sup>-1</sup> , 40%	-	[26]
Raw leachate, Permeate leachate	34 -> 5 -> <i>Chlamydomonas</i> sp. strain SW13aLS (Green algae)	-	25–50%/15 °C/ 60 days/ 1667 (22) lux	23 mg L <sup>-1</sup> , 82–100%	1–41 mg L <sup>-1</sup> , 22–98%	-	-	[27]
Raw leachate, Permeate leachate	4 -> <i>Chlamydomonas</i> sp. strain SW15aRL (Green algae)	Lab-scale, 250 mL flasks, batch mode	7–8.5/ 10%/15 °C/24 days/ 1667 (22) lux	100 mg L <sup>-1</sup> , 52–92%	1–6.5 mg L <sup>-1</sup>	-	-	[28]
3 pre-treated LL	<i>Chlorella vulgaris</i> CCAP 211/11B (Green algae)	Lab-scale, 1 L flasks, batch mode	~8/16–21 °C/12 days/ 2372–3113 (32–42) lux	18–75,150 mg L <sup>-1</sup> , 22–100%,0–27%	15–45 mg L <sup>-1</sup> , 38–100%	-	0.81–1.71 g L <sup>-1</sup> , 0.028–0.13 μ: day <sup>-1</sup>	[29]
Raw LL	Mixed algae–bacteria from an outdoor pond	Pilot-scale, 114 L Plexiglas tanks, semi-batch mode	5–20%/ 7–30 °C/22 weeks	5–90 mg L <sup>-1</sup>	-	-	0.48 g L <sup>-1</sup>	[42]
Autoclaved LL	<i>Chlorella</i> sp. (Green algae)	Lab-scale, 1000 mL flask, batch mode	9/ 10–100%/25–28 °C/ 13 days	286–2858 mg L <sup>-1</sup> , 60–90%	-	2392–23926 mg L <sup>-1</sup> , 4–60%	1.2 g L <sup>-1</sup>	[30]
Raw LL	Mixed algae from domestic wastewater	Lab-scale, 555 mL algae cathode microbial fuel cells	7–8.5, 5–40%, 30 °C, 5 days, 5000 lux	50–300 mg L <sup>-1</sup> , 80.6–98.7%	2.7–3.06 mg L <sup>-1</sup> , 0–80.28%	316–1557 mg L <sup>-1</sup> , 52–97%	-	[21]
Nitrified LL	<i>Chlorella</i> sp., <i>Scenedesmus</i> sp. (Green algae), <i>Oscillatoria</i> sp. (Cyanobacteria)	Lab-scale, 500 mL Erlenmeyer flask, batch mode	<8/10–30%/ 14 days	106–128 mg L <sup>-1</sup> , 34.4–84%	3.67–4.4 mg L <sup>-1</sup> , 100%	-	0.24–0.81 g L <sup>-1</sup> , 0.65–0.94 μ: day <sup>-1</sup>	[31]
Raw LL	Mixed algae/bacteria consortium from a pond	Small-scale, 0.25 L flask, batch mode Medium scale, 1000L aquarium tanks, semi-batch mode, feed weekly, large scale, 1000 L raceway ponds, feed weekly	6.5–9.5/25- nature 7.8–41.7 °C/ 7–364 days	-	-	-	2.1 g L <sup>-1</sup>	[37]

Table 4. Cont.

Leachate Characteristic (Source/Age)	Algal Species (Group)	System & Operation Mode	Treatment Condition pH/Leachate Conc/Temp/Time/Luminescence	Initial NH <sub>4</sub> <sup>+</sup> (mg L <sup>-1</sup> )/% Removal	Initial P (mg L <sup>-1</sup> )/% Removal	Initial COD (mg L <sup>-1</sup> )/% Removal	Biomass Cell Density (g L <sup>-1</sup> )/ Specific Growth Rate (μ = day <sup>-1</sup> )	Ref
Microfiltrated LL before and after nitrification from closed L	<i>Chlorella vulgaris</i> (Green algae)	-	-	38.8 mg L <sup>-1</sup> d <sup>-1</sup> NH <sub>4</sub> <sup>+</sup> removed	-	-	-	[32]
Raw LL	<i>Chlorella vulgaris</i> FACHB-31 (Green algae)	Lab-scale, 4L membrane photobioreactor, batch mode	100%/25 °C/ 8 days/9000 lux	136+632 mg L <sup>-1</sup> , 77.5-99.1%	15 mg L <sup>-1</sup> , 100%	342 mg L <sup>-1</sup>	0.66–0.96 g L <sup>-1</sup>	[33]
Six LL	<i>Chlamydomonas</i> sp. strain SW15aRL (Green algae)	Lab-scale, 250 mL flask, batch mode	10–100%/ 15 °C/40 days/ 957 lux	30–220 mg L <sup>-1</sup> , 70–100%	20–40 mg L <sup>-1</sup>	-	1.2 g L <sup>-1</sup> , 0.19 μ: day <sup>-1</sup>	[34]
Filtered LL	<i>Chlamydomonas</i> sp. Strain SW15aRL (Green algae)	Lab-scale, 250 mL flask, batch mode	30–60%	160 mg L <sup>-1</sup> , 83%	23 mg L <sup>-1</sup> , 91%	-	1.2 g L <sup>-1</sup>	[35]
LL	Wild mix algal/bacterial consortium	Lab-scale, 250 mL flask, closed batch mode, Pilot-scale, open 100 L aquarium tanks and 1000 L raceway ponds, semi-batch, feed weekly	Nature, 365 days	-	-	-	-	[36]

LL: Landfill leachate.

### 3. Challenges and Prospects of Algal-Based Leachate Treatment

Recent studies have shown the feasibility of algae-based treatment schemes for leachate remediation. However, there remain certain issues that must be answered by further research in the area for its large-scale implementation. First, developing the fundamental understanding of the algal-leachate system and the mechanistic role of different parameters that affect the system's performance is critical. The understanding would facilitate scaling up, improving the rate kinetics, optimization, and control of the performance parameters. Secondly, minimizing the environmental footprint of the process is critical for its large-scale implementation, as current practices reveal that leachate treatment by algae requires certain pre-treatment steps, supplementing phosphorus and nutrients and dilution by water. These steps, taken to ensure the growth of the algae, inevitably lead to material cost and water footprints. Lastly, to make the process sustainable, the paradigm of resource recovery must be incorporated into the treatment strategies, such that valuable products can be extracted along with the remediation and minimum waste generation in the process. In this section, we will discuss each aspect and its challenges and potential opportunities.

#### 3.1. Scaling—Up Challenges and Potential Future Approach

The available literature review shows that the mechanism of bioremediation and algal growth in the leachate medium remains poorly understood. Most studies identify certain process parameters such as pH, ammonia concentration, leachate concentration, temperature, biomass loading etc. and correlate the growth and bioremediation. However, there are studies too where these parameters either show poor correlation or contrary results [34,39]. The inconsistency shows that the mechanism of algal growth is still not properly understood. Part of the reason for the inconsistent results might be the lack of standardized experimental conditions in the studies, however, since the algal-leachate systems are relatively novel, more studies are needed to develop a standard protocol (e.g., similar mixing dynamics, light availability, leachate quality, and consortium) and mathematically explain and predict the performance of the treatment scheme. Such studies would ensure the development of large-scale treatment systems and optimization of their performance. Results from these studies can then be utilized to perform the predictive analyses such as life cycle assessments (LCAs) and techno-economic analyses (TEAs) to determine the long-term economic and environmental feasibility of the process. Few studies have investigated the system performance at a large-scale [36–39,42]. Moreover, it is seen that data from small-scale studies do not correlate well with the corresponding medium or large-scale studies [37]. Therefore, it is paramount to focus on developing a standardized protocol at a small-scale so that a good correlation is obtained for the corresponding medium/large-scale studies and thus improves the reliability of the LCAs and TEAs derived from the small-scale studies.

It is also noted that in certain cases the nutrients' impact or which of them plays limiting role is not properly understood [34]. Moreover, information on the complete nutrient requirement, their pathways/mechanism of assimilation for a species is also not fully known. Apart from nutrients requirements, their correct proportion in the media is also a critical factor to explore. Another challenge is to improve the rate kinetics of the process. While the biological processes are typically slower than the physicochemical approach, however, specific algal strains or consortiums can be explored that show faster growth, e.g., *Oscillatoria* sp. has shown a much faster growth rate and higher productivity than other species such as *Chlorella* and *Scenedesmus* in a recent study [31]. More such studies should be carried out for classifying the algal strains/consortium with respect to their growth rate and productivity under comparative, standardized conditions.

It has been noted in certain cases that the consortium is more tolerant of leachate media than individual species [39]. This means that a certain mix of individual species potentially plays a neutralizing role for each other in the consortium. Exploiting the symbiotic nature of different algal strains can be used to minimize the effect of metal toxicity to the algal consortium compared to a monoculture system. Therefore, studies estimating optimum speciation of the consortium is needed to tailor-made the inoculum needed for treating the leachate stream. Since each alga can be triggered by

one or more than one of the parameters, it is necessary to identify species that symbiotically mitigate toxicity and boost growth for other algal strains. Process innovation such as stepwise loading of leachate can be explored which would allow the adaptation of microalgae and at higher loadings, the most tolerant species can grow. This approach can be useful in the case of a bacterial-algal consortium where bacteria can break-down various nitrogen-containing compounds including ammonia into forms that can readily be utilized by the algae for adaptation and growth [25]. Symbiotically, microalgae provide O<sub>2</sub> to bacteria, which in turn receives CO<sub>2</sub> from bacterial respiration. The bacterial-algal approach would be particularly more useful when organic contaminants need to be degraded and mineralized. However, it is also to be kept in mind that bacterial presence limits the lipid productivity and impairs the bio-fuel value of the algal strain [25]. Therefore, some optimum mix of bacteria and algae should be estimated for achieving higher biomass growth and remediation without compromising the lipid quality.

In the scale-up scenario, handling, and disposal of sludge or dead biomass would also be a challenge. Harvesting techniques need to be developed, where dead biomass can be effectively removed from the growing biomass, as dead biomass will take up space—either in suspended or sedimented form—within the reactor, block the light availability and may promote the growth of other wild microorganisms by acting as a food source. These factors can potentially impact algae performance and, therefore need to be studied.

### 3.2. Environmental Footprints of the Process and Minimization Approach

The algal treatment of leachate mainly suffers from high toxicity, high turbidity and P deficiency of leachate media. The toxicity mainly emanates from high ammonia concentration at alkaline pH, and heavy metals concentration; high turbidity leads to poor light penetration and thus impacts the photosynthetic activities of the algae. To overcome these factors, most often the leachate media is diluted by wastewater or regular tap water. The literature survey shows that the typical dilution factor is often in the range of 10–50%, which means an additional 50–90 L of water is required to treat 10–50 L of leachate. This makes the process highly unsustainable and can limit the process' large-scale implementation, given the worsening water crisis globally. By dilution, the concentration of secondary nutrients such as Fe, Mg, and Zn, etc. which is essential for algal growth is also reduced [34]. Therefore, research should focus on identifying species that can thrive in the leachate media without any significant pretreatment, e.g., *Euglena* sp. becomes dominant when there is limited light penetration [38] or *Galdieria sulphuraria* thrives well under extreme acidic pH [43], making them suitable for leachate with high ammonium concentration and without any pH adjustment. The focus should be directed to such species which can grow in the leachate media similar to their natural environment, like algae from bog water can grow in dark conditions with high humic acid content [28]. To overcome high turbidity in leachate media, bioluminescent algal strains can be investigated for their leachate treatment capabilities either as standalone or in a consortium.

Another major limitation of using algae in the leachate media is of supplementing P externally. It has been widely reported that for healthy biomass growth, alga requires the N:P ratio to be above 16:1 (known as Redfield ratio). Different algae have different optimum N:P ratio needed for their growth. Most often, the leachate media is limited with respect to their P availability for complete nutrient removal and biomass growth. Therefore, it is essential to supplement P externally to achieve desirable biomass growth and nutrient removal. However, P is itself a non-renewable resource, whose availability is estimated to run out within the next 45–100 years [44]. This aspect of the algal–leachate system makes it unsustainable for the long term. Overcoming this challenge would be a major achievement in minimizing the environmental footprint of the process. One way of addressing the issue can be the utilization of waste streams such as agricultural run-offs, which are rich in phosphorus, as a diluent for leachate media. This would lead to combined treatment of multiple waste streams without resorting to additional water for dilution and P for nutrient supplements. More research into exploring bacterial-algal consortium where one species is specifically added to utilize P from its dead

biomass in the leachate media for the other species growth can be conducted to eliminate the need for adding P externally.

One such approach is to develop a synergistic algal treatment system to simultaneously treat both landfill leachate and anaerobic digester (AD) centrate from municipal wastewater treatment plants. High nutrient concentrations are found in the centrate in the form of ammonium (600–1200 ppm as ammoniacal N) and phosphate ions at concentrations up to 200 ppm [45–47]. Centrate is an ideal feedstock for nutrient recovery because it is highly concentrated and is thus low volume. The centrate accounts for 0.5% to 1% of a wastewater treatment plant (WWTP) influent flow but accumulates approximately 15% to 25% of the nutrient of the WWTP [48]. The anaerobic digester centrate stream can potentially help to achieve an optimum N:P ratio in the growth media, thus improves the nutrient recovery and biomass production.

The algal treatment of leachate may require a large land area. Some of the key factors determining the land area requirement are adequate light penetration and CO<sub>2</sub> absorption into the pond. The leachate is often dark brown in color. This affects light penetration negatively and thus challenges algal growth rate. Therefore, it is recommended that a shallow pond (0.6–1.5 m deep) be used for treating landfill leachate with microalgae [49]. To maintain such a shallow pond and given the rate of leachate generated daily (75,000 L day<sup>-1</sup>–350,000 L day<sup>-1</sup> [3]) a large area would be needed. To overcome this challenge, a continuously operated compact reactor design like MBR or MFC should be explored. Consideration must be paid to the residence time as large residence time would lead to depletion of CO<sub>2</sub> which is also critical for the algal growth [22]. To overcome this possibility of sequentially mixing raw and treated leachate should be explored which would replenish the nutrient supply and CO<sub>2</sub> concentrations.

### 3.3. Resource Recovery Opportunities from Algal–Leachate Treatment Systems

Algae have been considered as a 3rd generation biofuel for sometimes. In 2015, the market for 2nd (produced from non-food crops and waste) and 3rd generation (produced from algae, considered the most efficient source of biofuels) biofuels was valued at \$3574 million and is projected to increase to \$57,124 million by 2022 with a compounded annual growth rate (CAGR) of 48.9% from 2016–2022, with biodiesel accounting for the highest market share in 2015, and bioethanol is expected to witness the highest growth in the forecast period [50]. The growth is expected on account of the support offered by various government agencies for 3rd generation biofuels, mainly due to its potential to provide environmental services such as carbon sequestration and bioremediation. The quality of biofuels produced by algal strains depends upon the lipid content of the biomass, typically more than 20% is considered useful for biodiesel production [51–53]. It is established that under stressful condition, mainly under low nutrient concentration, the lipid concentration of the biomass increases [52,53]. However, stressful conditions like low nutrient levels also lead to lower biomass growth and thus impact overall lipid productivity. Therefore, more research is required in this area to arrive at an optimum level of nutrients that maximizes the lipid concentration in the biomass without affecting much of its productivity. The cost aspects need to be investigated in terms of the ratio of energy expended to energy yield from the biomass.

## 4. Conclusion

MSW generation, landfilling, and subsequent landfill discharge have shown an increase in recent years. If left untreated, serious environmental pollution may result. To address the issue, a robust, eco-friendly solution, with minimal environmental footprints, is needed. Microalgae have shown immense potential in treating wastewater. Their utilization for treating landfill leachate is relatively novel. Moreover, microalgae remediation methods show promise as an alternative source of 3rd generation biofuels. Recent market trends also suggest huge demands for biofuels produced by microalgae. Therefore, the microalgal approach holds both environmental and economic benefits and will be highly sustainable. However, very few research works exist that show the treatment of

leachate using microalgae. The literature survey reveals that most of them are small-scale lab studies. Moreover, the treatment approach needs to develop a fundamental understanding of the process and standardized protocols, such that large scale studies can be carried out which correlate well with the lab results. The focus should be directed to minimize the environmental footprints of the approach. This would require certain process innovation, identifying suitable algal strains, and their synergistic consortium and reactor design. The attention should also be given to improve the biofuel quality and yield of the biomass. Additionally, resource recovery options, such as food supplements, fertilizers, etc. need to be explored to fully utilize the algae for leachate treatment.

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